

Mapped and written guidance in relation to birds and onshore wind energy development in England

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RSPB Research Report No 35

**A report by the Royal Society for the Protection of Birds, as part of a
programme of work jointly funded by the RSPB and Natural England**

2009

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ISBN 1-905601-18-2

Executive Summary

Concerns over climate change have led to renewable energy targets for EU member states, with the UK being allocated a target of 15 % of energy consumption from renewable sources by 2020. This will require a large increase in renewable energy production, with wind energy currently representing the fastest growing renewable energy source. Climate change poses the single greatest long-term threat to birds and other wildlife, and the RSPB recognises the essential role renewable energy, including wind, plays in addressing this problem. However, poorly sited wind farms can also have negative effects on birds, leading to potential conflict where proposals coincide with areas of activity for species of conservation concern.

To help minimise this conflict, the RSPB and Natural England have jointly produced a GIS map and written guidance to aid the planning process for onshore wind energy development in England. The map is based on distributional data for twelve sensitive bird species, plus statutory SPAs (Special Protection Areas), and sites containing nationally important populations of breeding waders and seabirds, or wintering waders or wildfowl. Ten of the 12 species included on the map are listed on Annex I of the EU Birds Directive, with two additional species of conservation concern included, due to concern about potential impacts on their rapidly declining or highly localised populations. All species included have known or suspected (based on information on the species' behaviour or ecology) susceptibility to the effects of wind turbines on birds, either through collision mortality and/or disturbance displacement.

Each 1-km square in England was assigned one of three sensitivity ratings (high, medium or unknown) as follows:

- For each species, 'sensitivity criteria' were created using information on foraging ranges, collision risk, disturbance distances and other relevant features of behavioural and population ecology, from reviews of literature and the best available information.
- Distributional data were then buffered according to species (e.g. circles drawn around nest or roost locations) and sensitivity ratings of either high or medium were applied to these buffered areas based on these sensitivity criteria.
- Maps of buffered areas for each species were converted to 1-km square grids, by selecting the sensitivity rating of the centre of each 1-km square.
- The SPA network, selected IBAs (Important Bird Areas), SSSIs (Sites of Special Scientific Interest), and WeBS (Wetland Bird Survey) principal sites were also included to represent nationally important distributions of breeding waders and seabirds, or wintering waterfowl.
- These sites were amalgamated with the individual species' maps by choosing the highest sensitivity rating for each 1-km square.

The map is not comprehensive and it was not possible or practical to include some species or important sites on the map. Notably, wintering waterfowl feed extensively, and in some locations in significant numbers, on cropped land. This information could not be included due to a lack of readily available data at the national scale. For these sites, written guidance has been included, as well as reference to regional data sources, where these are available. Those species that were not included on the map but also considered to be of conservation concern and potentially susceptible to wind farms were: honey buzzard, merlin, peregrine falcon, red kite, golden plover and dunlin. Honey buzzard could not be included as data were not available at an appropriate spatial scale. Merlin and peregrine falcon were not mapped due to their relatively widespread distributions. Red kite was not mapped, as the species is

now widespread and numerous in England, and continuing to expand its range, meaning that a map of distribution would quickly become out of date. In addition, golden plover and dunlin were not mapped (outside protected sites or principal sites identified by WeBS counts) due to lack of comprehensive data. Written guidance has been included for all these species. A résumé of sensitivity criteria for mapped species, and written guidance for unmapped species, can be found on pages 18-29 of this report, with full literature reviews for each species in Appendices 2 and 3.

More detailed and longer-term studies are likely to be necessary in locations identified as 'high' or 'medium' sensitivity to assess potential impacts, and to inform appropriate mitigation or compensation measures. The map is not intended to depict 'no-go' areas for development or indicate any presumption that RSPB or Natural England will automatically object to proposals in or near to those areas identified as being highly sensitive. Furthermore, data deficiency and gaps in survey coverage preclude a distinction between 'low' and 'unknown' sensitivity squares. Many of these areas will not be within the range of the species of interest, or will not contain suitable habitat, but some may be sensitive. The map and guidance do not obviate the need for specialist, detailed assessment of specific wind energy proposals on a case-by-case basis.

It is anticipated that the map and guidance will be most useful to RSPB and Natural England staff, Local Authorities and wind energy developers in the early stage of the planning process, particularly in informing site selection and scheme design and in defining likely study and survey requirements. Early, pre-application consultation with the RSPB and Natural England, to discuss possible bird sensitivities, survey requirements and optimum design for individual proposals, is encouraged.

We also anticipate that the map and written guidance could inform future regional and sub-regional renewable energy capacity assessments and help ensure that the expansion of onshore wind energy is delivered sustainably.

As well as being a useful tool in its own right, the map and guidance will be a valuable component of Natural England's broader approach to assessing the degree to which the natural environment can accommodate onshore wind energy development.

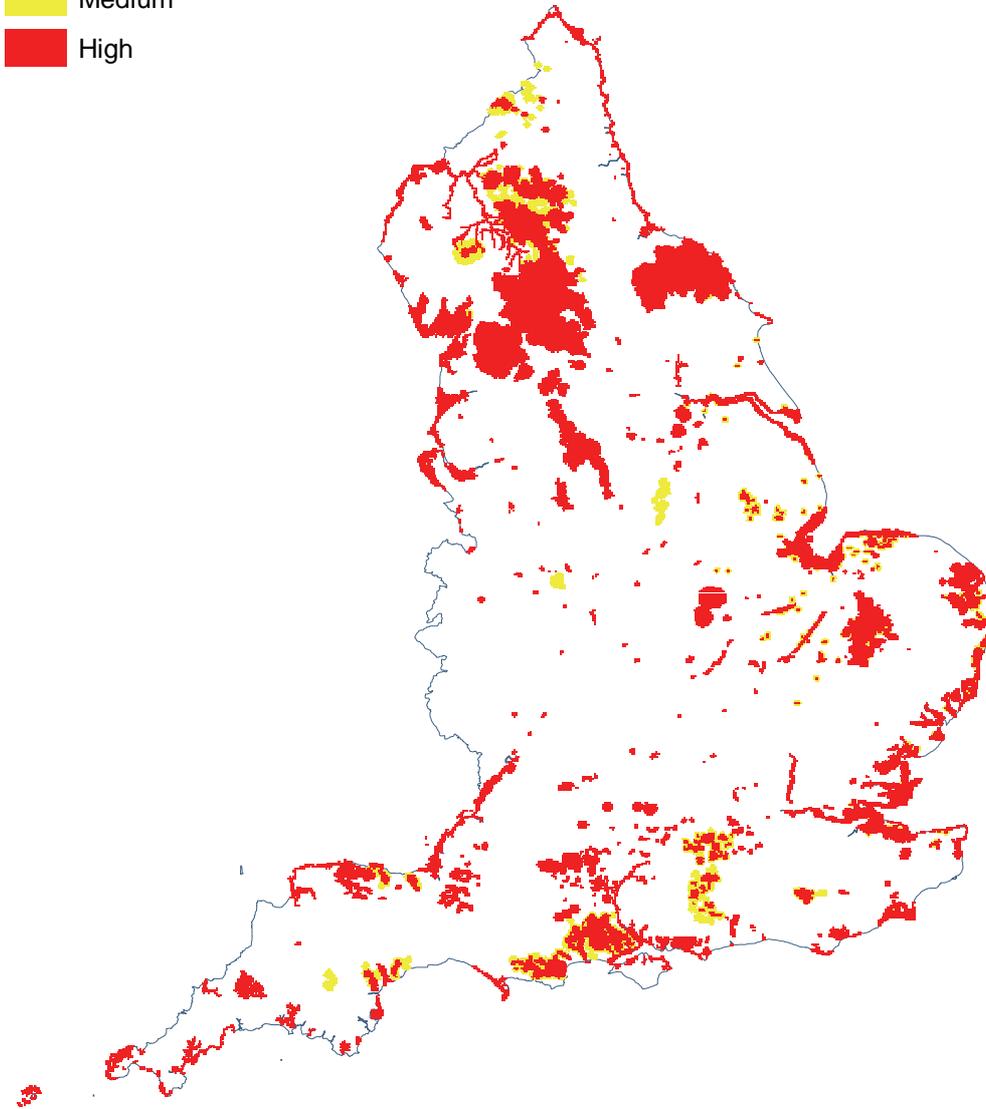
The map indicates a greater incidence of bird sensitivities in coastal and estuarine areas and upland areas in the north of England. Wind farms in these areas will require particularly careful assessment and planning to minimise adverse impacts on birds of conservation concern. It is hoped that this map and guidance will assist in this process. Overall, 16 % of the area on the map is classified as 'high sensitivity' (21 350 1-km squares), 2 % as 'medium' (2935 1-km squares) and 82 % as 'unknown sensitivity' (110 880 1-km squares). Although strictly unknown, the latter category is likely to contain large areas of low sensitivity.

The map was created using the best data and information currently available, but there are inevitably caveats that apply to its use:

- Data deficiency and gaps in survey coverage mean that the map cannot be comprehensive, and therefore preclude a distinction between 'low' and 'unknown' sensitivity squares. Many of these areas will not be within the range of the species of interest, or will not contain suitable habitat, but some may be sensitive. No liability is accepted for the presence or absence of species at particular sites contrary to that indicated on the map.
- It was not always possible to include information on the locations of immature and non-breeding (including wintering and migrating) birds. The SPA network has been used to represent distributions of wintering waterfowl, in combination with other sites (IBAs, SSSIs and WeBS principal sites) holding nationally important concentrations of wintering waders or wildfowl. For the twelve species mapped separately, data related primarily to breeding birds, although some information on wintering locations was included where available, this is incomplete.
- The sensitivity map was created by collating data that were collected for other purposes, and thus not tailored specifically to the map's requirements (which would be an enormous task).
- The map is not a substitute for Environmental Impact Assessment, but is intended as an indicative map of likely bird sensitivities, to help guide decision-makers in the early stages of the planning process, for example at the scoping stage to define study requirements.
- The map needs to be updated regularly to incorporate new data, and to reflect revisions to the sensitivity criteria in the light of new information, notably from research and experience at operational wind farms.

Sensitivity rating

- Unknown
- Medium
- High



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Figure 1. Map of sensitive bird areas in relation to onshore wind farms in England.
Based on the highest sensitivity rating, for any of the species or sites included, in each constituent 1-km square.

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Introduction

Climate change

Climate change poses a serious threat to humanity and is likely to worsen global problems such as drought, famine, floods, disease, regional insecurity and population displacements, and seriously hinder efforts to tackle poverty in poorer countries (IPCC, 2007). Energy security is also of increasing concern, with 20 % of the UK's gas supply coming from imports in 2007 (BERR, 2008). Climate change is considered the single greatest long-term threat to birds and other wildlife, with models of mid-range climate change scenarios predicting that approximately 15 % to 37 % of species globally will be 'committed to extinction' by 2050 (Thomas *et al.*, 2004). Climate change has been implicated in the large-scale breeding failure of seabirds in 2004 on the North Sea coast of Britain (Eaton *et al.*, 2005), and in declines in a range of bird populations across Europe (e.g. pied flycatchers *Ficedula hypoleuca*, Both *et al.*, 2006; black grouse *Tetrao tetrix*, Ludwig *et al.*, 2006 and ring ouzel *Turdus torquatus*, Beale *et al.*, 2006). Predictions of the potential future distributions of European breeding bird species by the end of the 21st century, based on suitable climate conditions found that, for the average species, the predicted future range is just 4/5 the size of the current range (Huntley *et al.*, 2007, Walmsley *et al.*, 2007, Green *et al.*, 2008). Average overlap between current and predicted future range is 40 %, for some species the two do not overlap at all, and ability of species to occupy this future suitable habitat could be constrained by poor dispersal ability, habitat fragmentation or physical barriers (Huntley *et al.*, 2007, Walmsley *et al.*, 2007, Green *et al.*, 2008).

Renewable energy

Tackling climate change will require reduced energy use and increased energy efficiency, as well as a switch to energy sources with lower greenhouse gas emissions such as renewable energy. Currently, approximately 5.0 % of the UK's electricity comes from renewable sources (2007 figure, BERR, 2008). However, the proposed EU Renewable Energy Directive has set a target of 20 % of energy consumption across member states to come from renewable sources by 2020, with the UK allocated a target of 15 % (EU, 2008). In order to help reach renewable energy targets, the UK Government introduced the Renewables Obligation in 2002, requiring that electricity suppliers supply a specified and increasing proportion of electricity from renewable sources (BERR web-site, 2008). Wind power currently makes a contribution of 8.8 % of renewables in input terms (i.e. in volume of fuel inputs of renewables to heat, electricity and transport, after conversion to an oil equivalent basis), but accounts for 27 % of electricity generated by renewables (2007 figures, BERR, 2008). However, onshore and offshore wind represent the cheapest, most technologically advanced and most rapidly growing forms of renewable energy, with wind growing by 25 % between 2006 and 2007 in terms of input into electricity generation (BERR, 2008). The UK has one of the largest wind resources in Europe (Risø National Laboratory, 1989) and currently has 179 operational wind farms onshore, and seven offshore (Table 1, BWEA, 2008).

Impacts of wind farms on birds

The RSPB recognises the essential role that renewable energy, including wind, plays in the attempt to tackle climate change, and supports appropriately located wind energy developments. Unfortunately, poorly sited wind farms can have negative effects on biodiversity (Langston and Pullan, 2003, Drewitt and Langston, 2006), making it necessary to

balance sustainable renewable energy development and nature conservation interests. Effects of wind farms on birds vary depending upon factors such as species / taxa, season, weather, habitat type and individual site characteristics such as topography (e.g. Langston and Pullan, 2003, Barrios and Rodríguez, 2004, Drewitt and Langston, 2006, 2008, de Lucas *et al.*, 2008, Smallwood and Thelander, 2008). As wind energy is still a relatively new technology, information about these effects is sparse, with few relevant published peer-reviewed papers on the subject (Whitfield and Coupar, 2008), although the knowledge base is increasing.

Wind farms can affect birds in four main ways: collision, disturbance displacement, barrier effects (sometimes considered a type of disturbance displacement) and direct habitat loss. Direct habitat loss is generally considered to be a relatively minor concern (Percival, 2000), although there will be exceptions, for example where overlap is with very rare species with small home ranges (Whitfield and Coupar, 2008). Collision risk and disturbance displacement are considered the two predominant effects. Cumulative impacts resulting from several wind farms in the same area or affecting the same species are of particular concern (Scottish Natural Heritage (SNH), 2005a).

Collision risk

Wind farms in the UK have not been associated with high collision rates to date, because on the whole they have been constructed in areas with little bird activity (Percival, 2005, Madders and Whitfield, 2006), and generally, collision rates recorded at many wind farms in the UK and elsewhere have been low (e.g. Erickson *et al.*, 2001, Percival, 2005, Drewitt and Langston, 2006, 2008). However, in some cases at poorly sited wind farms, collision rates have been high, for example minimum estimates of annual fatalities of 67 golden eagles, among over a thousand raptor collisions per annum, at Altamont Pass in California (Smallwood and Thelander, 2008) and 36 common kestrels and 30 griffon vultures at Tarifa in southern Spain (Barrios and Rodríguez, 2004). These figures are corrected for search efficiency and scavenger removal. Twenty white-tailed eagle fatalities were reported between 2005 and 2008 at a wind farm at Smøla in Norway (weekly corpse sample searches have been conducted since 2006, figures are not corrected for search efficiency and scavenger removal, T. Nygård, pers. comm.). It should be noted that wind farms in both Altamont Pass and Tarifa comprise a range of older wind turbine designs that do not feature in modern wind farms, although recent work by de Lucas *et al.* (2008) found no effect of turbine type on collision rates at two wind farms in Tarifa.

Different species groups vary in terms of susceptibility to collision, with raptors appearing to be particularly susceptible (e.g. NWCC, 2000, Langston and Pullan, 2003, Thelander *et al.*, 2003, Smallwood and Thelander, 2004, Anderson *et al.*, 2004, 2005). Whitfield (2007) suggests that much of this vulnerability appears to be due to reduced avoidance of wind turbines by raptors (e.g. Whitfield and Madders, 2006a, b). Some other groups also appear quite susceptible, for example, together with raptors, gulls accounted for most of the fatalities in Hötker *et al.*'s (2006) review of wind farm impact studies, and an estimated 165 collisions of several species of tern per year occurred at a 25-turbine wind farm at Zeebrugge, Belgium (Everaert and Stienen, 2006). Moorehead and Epstein (1985) suggested that large wetland birds, such as geese, might be especially susceptible to collision. However, Hötker *et al.* (2006) found that records of collisions in geese were relatively rare, and generally there have been few recorded collisions of geese at wind farms. A recent review of wind farm impact studies involving waders by Whitfield (2007) concluded that waders have relatively low susceptibility to collision, as they are often displaced from the wind farm area. Although there is clearly a relationship between the two factors, it should be noted that disturbance displacement and collision are not necessarily mutually exclusive, as individuals may

respond differently to wind farms, depending on factors such as age, breeding status and season. For example, 13 of the 20 white-tailed eagle fatalities at Smøla occurred between March and May, and avoidance of turbines is much lower at this time of year (T. Nygård, pers. comm.). Although collisions of passerines have been recorded (Hötker *et al.*, 2006, Smallwood and Thelander, 2008) these have generally not been in particularly large numbers (e.g. an estimated 57 passerine collisions annually at Altamont Pass Wind Resource Area in California, compared to over a thousand raptor collisions, Smallwood and Thelander, 2008). Devereux *et al.* (2008) suggest that farmland passerines are less likely to be collision victims than larger, less maneuverable species, although it should be noted that their small size means that under-reporting of collision fatalities due to lower detection rates or higher rates of scavenger removal is a potential problem.

Hötker *et al.* (2006) presents a review of collisions at 129 wind farms, mainly focusing on species relevant to Germany. However, it should be noted that this review is the result of collation of records from a variety of sources, as opposed to controlled corpse searches at wind farms, and different taxa may feature more or less prominently due to factors such as size.

The 'Band' Collision Risk Model (Band *et al.*, 2006) allows estimation of the number of annual collision fatalities of a species at a particular wind farm, and is widely used during Environmental Impact Assessment (EIA). However, the model is very dependent upon estimated turbine avoidance rates, which vary between different taxa / species, and are often poorly quantified (Chamberlain *et al.*, 2005, 2006). Collision models also rely on the assumption that collision rate is related to bird abundance, which recent work suggests is not necessarily the case (de Lucas *et al.*, 2008). Estimates of annual collision rates and avoidance rates should be treated with caution, and used as comparative rather than absolute measures (Chamberlain *et al.*, 2005, 2006, Whitfield, 2007).

The reasons why birds collide with wind farms are not clear, but it is often assumed to be due to them not seeing the turbines ('invisible blades' or 'motion smear' hypothesis, Hötker *et al.*, 2006). However, Whitfield (2007) considers that the limited success of measures to increase the visibility of turbines in preventing collisions (e.g. McIsaac, 2001, also see Hötker *et al.*, 2006 for review) suggests that an alternative hypothesis may be more likely; that birds do see the blades but are unable to avoid them under certain conditions, depending on factors such as wind or turbine location (Drewitt and Langston 2008, de Lucas *et al.*, 2008), or when distracted by certain behaviours. A study of raptor collisions at two wind farms in Tarifa, southern Spain, supports this theory, with griffon vultures colliding more when uplift wind conditions were poor, for example due to gentle slopes or weaker thermals, presumably due to lower manoeuvrability (Barrios and Rodríguez, 2004, de Lucas *et al.*, 2008). Bird mortality varied markedly with season, but was not highest in the season with highest bird abundance (de Lucas *et al.*, 2008). Rather, mortality was related to factors such as turbine height and elevation, with more fatalities being associated with higher turbines and elevations (de Lucas *et al.*, 2008). A better understanding of factors causing collision is essential in identifying locations where wind farm developments may lead to a higher collision risk.

Disturbance displacement

Wind farms also may affect birds via disturbance displacement (e.g. Drewitt and Langston, 2006, Hötker *et al.*, 2006), and as with collision, different species and groups vary in their susceptibility to this effect. Hötker *et al.* (2006) presents a review of disturbance displacement across 129 wind farms, mainly in Europe, focusing on species relevant to Germany. Observed effects of disturbance displacement were more common in the non-breeding season, with

waders and wildfowl being particularly susceptible (Hötker *et al.*, 2006). Breeding season displacement effects were only common in waders and gamebirds (Hötker *et al.*, 2006). A review by Whitfield (2007) also suggests that disturbance displacement around wind turbines is relatively common amongst waders, and particularly wintering birds, with there being fewer examples of disturbance displacement in breeding birds. This could be because breeding birds show greater site fidelity, or are more limited in their site choice. Effects could also be masked by a time lag as birds faithful to the sites return, but new birds may not settle in them. However, recent work by Pearce-Higgins *et al.* (2008) provides evidence of disturbance displacement in breeding golden plover up to distances of at least 200 m, and also in other breeding waders, at distances of 0 – 800 m, depending on the species (Pearce-Higgins, unpubl.). Geese appear particularly susceptible to human disturbance generally, and there are a number of examples of displacement from wind farms (e.g. reduced feeding densities close to turbines, extending up to 600 m, Kruckenberg and Jaene, 1999, for reviews see Langston and Pullan, 2003, Hötker *et al.*, 2006), although there is a wide range of recorded displacement distances, indicating no displacement in some situations. A study at a wind farm on a mixed grassland/farmland habitat in the USA reported displacement of passerines during the breeding season, however the study did not appear to control for differences in habitat types at different distances from turbines (Leddy *et al.*, 1999). Devereux *et al.* (2008) found no evidence for displacement of farmland birds in winter (with the exception of common pheasant) at two wind farms in East Anglia, although effects during the breeding season still require investigation.

Disturbance can affect species in a number of ways; either by direct loss of e.g. nesting, foraging, roosting or moulting habitat, or by affecting productivity, and potentially survival. Meta-analysis of 19 datasets suggested reduced abundances of birds at wind farms, particularly of wildfowl and waders (Stewart *et al.*, 2007), in accordance with results on disturbance displacement above. It is frequently suggested that habituation may occur at wind farm sites, reducing the impact of disturbance displacement over time.

However, contrary to this suggestion, analysis by Stewart *et al.* (2007) found that abundances at wind farm sites actually decreased over time, and a review of wind farm impact studies by Hötker *et al.* (2006) concluded that evidence for habituation was neither widespread, nor indicative of a strong effect.

Significance of effects

The importance of impacts of wind farms on birds depends upon whether they result in effects at the population level. The term 'population level' can be applied at different spatial scales, both national and regional effects may be considered important (e.g. for discussion see SNH, 2006). Quantification of population-level effects is rarely achieved, due to cost and practicalities (Morrison and Pollock, 1997, Sterner *et al.*, 2007), but potential population level effects (i.e. population declines) were associated with several of the examples above (Everaert and Stienen, 2006, Sterner *et al.*, 2007, Thelander and Smallwood, 2007). For species of conservation concern, even small increases in mortality may be significant either from individual wind farms or the cumulative effects of multiple wind farms. For proposals on or near designated sites, an assessment of the potential impact on the qualifying populations is required. For disturbance displacement, the significance of observed responses depends upon whether there are effects on survival or productivity. This can occur, for example, if birds are excluded from nesting or foraging habitat and equivalent, alternative suitable habitat is not available (effectively habitat loss), or if energy expenditure is increased above that which can be readily compensated for. An example of the latter is the case of little terns at an offshore wind farm at Scroby Sands in Norfolk, where construction of the wind farm resulted in a nearly three-fold increase in foraging flight distances (Perrow *et al.*, 2006).

Aims and objectives

Careful location of wind farms is key to minimising impacts on birds. Wind farms in the UK are frequently sited in areas not considered likely to have significant effects on bird populations (e.g. Percival, 2005, Fielding *et al.*, 2006, Bright *et al.*, 2008). However, care when locating wind farms will be increasingly important in maintaining this situation as wind energy expands to help meet ambitious renewable energy targets.

The sensitivity map is intended to help ease conflict between wind farm development and nature conservation by indicating the areas where there is a greater probability of detrimental effects on important bird populations, which might require more detailed and longer-term information to assess potential impacts, and potentially also appropriate mitigation or compensation. The map is intended to help guide decision makers, such as Local Authorities, and wind farm developers, in the early stages of the decision making process, and is not a substitute for site-specific Environmental Impact Assessment (EIA). Guidance for bird survey methods during EIA is provided in SNH (2005b, 2009a, b). Early consultation with the RSPB and Natural England, to discuss potential bird sensitivities and survey requirements for individual proposals, is recommended.

Methodology

The project has produced a map of ranked bird sensitivities to the development of onshore wind farms in England, at 1-km square resolution, together with written guidance on mitigation measures involving wind turbine locations in relation to sensitive nesting birds and birds outside the breeding season. The map is a GIS product, based on the distributions of bird species of high conservation priority in England, with the sensitivity rankings based mainly on published information relevant to their sensitivity to different aspects of wind farm development and operation.

Species list

Priority species for inclusion on the map were those listed on Annex I of the Birds Directive (EC, 1979), as representing birds of high conservation priority. A few additional species of conservation concern that display particular risk factors in relation to wind farms also were included. Bird species for which there was little evidence of adverse effects or that are of lower conservation priority were not included.

Special Protection Areas (SPAs) were included on the map as 'high sensitivity', as these sites are classified under Article 4 of the Birds Directive as "most suitable territories" to deliver the conservation of Annex I and regularly occurring migratory birds. In particular, these sites were used as a surrogate for the distributions of congregational species of wintering geese and other waterfowl, and colonial nesting seabirds, as designation criteria include populations exceeding 20 000 birds as well as individual species criteria (Stroud *et al.*, 2001). This was because, for many species, a high proportion of their populations occur within the SPAs (Stroud *et al.*, 2001). Species included on the map by means of the SPA network alone are presented in Table 2.

The SPA network was supplemented with data for Important Bird Areas (IBAs, Heath and Evans, 2000, although note updated boundaries were used, RSPB, unpubl.), Sites of Special Scientific Interest (SSSIs) and Wetland Bird Survey (WeBS) principal sites that held nationally

or internationally important concentrations of breeding waders, or wintering waterfowl (see Table 3 for details). IBAs for breeding seabirds were also included (see Table 3).

For species of goose and swan, SPA boundaries and estimates of coverage in Stroud *et al.* (2001) are largely based on roost locations (Table 4). However, these species often feed on cropped land in the vicinity of designated sites, and could be at risk of collision when flying between roosting and feeding sites, or of displacement from feeding sites. Few areas of cropped land have been included in the SPA network to date, and there are difficulties in mapping goose and swan feeding areas, which may be at considerable distance from roost sites and may change depending on cropping patterns. Likewise, for other wintering waterfowl, boundaries of coastal and estuarine SPAs often exclude important inland feeding and/or roosting areas. However, the Birds Directive requires Member States to protect the habitats of these species outside SPAs too, and, more specifically, the Habitats Directive requires assessment of the implications of plans or projects in view of site (SPA and/or SAC) conservation objectives, which can include off-site effects where these may reduce SPA populations (for example through damage or disturbance to 'functionally linked' feeding, roosting or flyway areas). There is no single national data source for waterfowl feeding areas on cropped land, with the Wetland Bird Survey (WeBS) focusing predominantly on coastal and estuarine areas (Austin *et al.*, 2008). However, some data are available regionally. Due to the absence of a comprehensive national data set, cropped habitats used by these species for feeding have not been included on the map. However, regional data sources, and guidelines with respect to search-areas for waterfowl feeding areas, and possible mitigation against displacement from these areas, have been provided in Appendix 1.

The SPA network alone is not sufficient to represent distributions of all species of concern, because designated sites tend to include small proportions of dispersed species. For example, only 15 % of the British breeding population of golden eagle occurs within SPAs (Stroud *et al.*, 2001). Therefore, distributions were included for a further 12 species of conservation concern (Table 5) for which the literature indicated sensitivity to collision risk, disturbance or changes in habitat, especially in relation to wind farms. The suite of sensitive species was chosen after reviewing published literature about sensitivity of bird species to wind farms (e.g. Langston and Pullan, 2003). Species were included if (a) data were available, (b) the species has limited range or occurs in restricted habitats or sites, and (c) there was some evidence or probability that wind farms can have an adverse effect. Estimated SPA coverage in Stroud *et al.* (2001) for bittern and stone-curlew was relatively high. However, this is known to have decreased since the time of the review (for example, an estimated 90 % of the British breeding population of bittern occurred within the SPA network at the time of the review, but this was estimated at 59 % by 2008 (S. Wotton, pers. comm.)), and so these species also were mapped separately.

Ten of the twelve species mapped separately are listed on Annex I of the EU Birds Directive. The two other species were included because, whilst not listed on Annex I, they are either very localised in distribution or undergoing rapid national population declines. Bean goose was included as they regularly occur in just two areas in England, one of which supports the only British population of the Tundra bean goose race *A. f. rossicus*, and is not an SPA, and geese are particularly susceptible to disturbance displacement from a variety of sources, including wind farms (e.g. Kruckenberg and Jaene, 1999). Black grouse was included as the UK population is small and rapidly declining (Sim *et al.*, unpubl.), it is a red-listed Bird of Conservation Concern (BoCC) (Gregory *et al.*, 2002) with low dispersive ability and is particularly susceptible to collision with man-made structures (e.g. Bevanger, 1995). Both of these species were allocated a sensitivity rating of 'medium' to reflect their national, rather

than international (i.e. non-Annex I) conservation status. IBAs that held nationally important populations of species included on the list were also included.

It was not possible to include honey buzzard on the sensitivity map, as data were not available at an appropriate spatial scale. Other Annex I raptor species considered for inclusion were merlin, peregrine falcon and red kite. However, it was decided that it would be more appropriate to provide written guidance for these species, rather than include them on the map. Golden plover and dunlin were also not included on the sensitivity map outside designated sites or principal sites identified by WeBS counts, due to lack of comprehensive data. Potential data sources for species not included on the map are listed in Table 6, and written guidance for these species provided in Appendix 2 and summarised in the results section.

Data

National surveys for many of the species included are undertaken at five/six or ten/12 year intervals as part of the Statutory Conservation Agency/RSPB Breeding Bird Scheme (SCARABBS, Baker *et al.*, 2006). In addition to these periodic surveys, records for many of the species included on the sensitivity map are submitted to the Rare Breeding Birds Panel annually (e.g. Holling and the RBBP, 2008). Dedicated annual monitoring programmes are undertaken for some of the scarcer species included (e.g. bittern, see Wotton *et al.*, 2008). The data sources used for each species are presented in Table 5. For most, survey coverage was reasonably comprehensive.

The most recent data available were used, but age of dataset varied between species. Data from surveys spanning a range of years was used for most species, in order to provide a better representation of breeding locations, i.e. because of sample surveys or because of the birds' use of multiple sites.

Most national surveys for the species in Table 5 occur during the breeding season, so the distributions mapped for these species related primarily to breeding birds. Survey units varied (Table 5), with most being nest locations or adults showing breeding activity, but calling males being used as an indication of breeding activity (Gibbons *et al.*, 1993) for secretive nesting species. For hen harrier and marsh harrier, locations of communal winter roosts were also included. Additionally, as mentioned above, the SPA network, together with data for IBAs, SSSIs (supporting nationally important populations of wintering waterfowl or breeding waders), and WeBS principal sites (supporting nationally or internationally important numbers of wintering waterfowl) were used to represent distributions of breeding waders and seabirds, or wintering waterfowl. No data on immature or migrating birds (except for migrating birds included in the SPA network) have been incorporated.

Creation of the map

The map was created in the Geographical Information System (GIS) MapInfo Professional version 6.0. Distributional data for each bird species were mapped on a separate layer. Most data were available at 100-m resolution, but some were available only at 1-km.

The distributional data were buffered according to species criteria (e.g. as circles drawn around nest locations, or lines at a set distance around a nature reserve used by wintering bean geese), and sensitivity ratings were applied to these buffered areas (Table 7). For two species, buffering was not applied as a site-based approach was taken. These were bittern, for which all reedbeds used since 1990 were included, and nightjar. As nightjar are relatively

numerous and widespread in England, the SPA network and other sites with nationally important populations (sites containing over 1 % of the British breeding population based on the 2004 national survey data) were used to represent sensitive locations for nightjar (see Table 5 and Appendix 3 for details).

Species' sensitivities and buffer distances were determined on the basis of information relating to collision risk, foraging ranges and/or information relating to disturbance displacement from wind turbines, taking into account other relevant features of behavioural or population ecology for each species. In the absence of this, information relating to disturbance displacement from other sources of disturbance was used. Disturbance distances need to be treated with caution, as individual responses to disturbance depend on a number of factors, such as the quality of the site, distance to other suitable sites, and investment made in a site (Gill *et al.*, 2001). Literature searches were conducted using species' names as keywords, as well as searches under 'wind farm', wind turbine', 'collision' and 'disturbance' in ISI Web of Knowledge, and web-sites of various research bodies. Information from peer-reviewed published literature was given greater weight than that from 'grey literature'. Finally, in the absence of adequate documented information, relevant species' experts were consulted. Once completed, the literature review, proposed buffers and sensitivity ratings for each species were circulated amongst species' experts for comment and revision.

Buffered areas, or sites (e.g. SPAs, reedbeds used by breeding bittern), were assigned a rating of 'high' or 'medium' sensitivity for each species (Table 7). Maps of buffered areas for each species were then converted to 1-km square grids, by selecting the sensitivity rating of the centre of each 1-km square. For irregularly shaped polygons representing a site boundary (i.e. SPAs and reedbeds for breeding bitterns) any 1-km squares intersecting with the site were selected to ensure capture of smaller sites.

Individual species' maps were combined to produce a composite map layer showing the highest sensitivity rating for each 1-km square for any species. This was the preferred method for the production of the sensitivity map, as the protective legislation applies equally to locations with single species and multi species importance. Thus, no weighting by species or number of species present was applied to the sensitivity ratings. Due to paucity of information available on species-specific effects, a three-level scale of sensitivity rating was used, as follows:

- 'High sensitivity'.
- 'Medium sensitivity'.
- 'Unknown sensitivity.'

It was not possible to distinguish between 'low' and 'unknown' sensitivity areas, due to data deficiency, and the use of written guidance for some species. For example, more detailed data may be available at the regional level relating to cropped feeding areas for wintering waterfowl. However, it is expected that in a large proportion of the 'unknown sensitivity' category squares there will be low ornithological sensitivities.

Sensitivity ratings have been used to indicate both likelihood of an effect via collision or disturbance displacement, and conservation status. For example, for golden eagle and marsh harrier, a concentric ring buffer approach has been taken, with an inner core area classified as 'high sensitivity' and an outer area as 'medium sensitivity'. This is intended to indicate estimated use of the area, and where collision and indirect habitat loss due to displacement is considered most likely. For bean goose and black grouse only a 'medium sensitivity' rating

was applied, in order to indicate these species' national rather than international (i.e. non-Annex I) conservation status.

Results

Sensitivity map

The sensitivity map indicates a greater incidence of bird sensitivities in coastal and estuarine areas and upland areas in the north of England (Figure 1). Overall, 16 % of the area on the map is classified as 'high sensitivity' (21 350 1-km squares), 2 % as 'medium' (2935 1-km squares) and 82 % as 'unknown sensitivity' (110 880 1-km squares).

Sensitivity criteria: mapped species

Summaries of sensitivity criteria are presented below; full reviews and rationale for the criteria applied for each species are provided in Appendix 3.

Bittern *Botaurus stellaris*

- Although increasing, the UK bittern population remains very small (76 booming males in 2008, Wotton *et al.*, 2008), and estimated SPA coverage has decreased since the SPA review (from 90 % of the breeding population, Stroud *et al.*, 2001, to 59 % of booming males in 2008, S. Wotton, pers. comm.).
- During the breeding season, males feed within their home ranges (White *et al.*, 2006), but females in the UK frequently travel from the nest site to forage, and can regularly make flights up to 2 km (unpublished data in Gilbert *et al.*, 2005). Females may be at risk of collision during such flights, particularly if they are over trees or houses (G. Gilbert, pers. comm.). Males may also defend several fragmented bits of territory that may be separated by tall trees or other obstructions (G. Gilbert, pers. comm.). Bittern seem to be quite prone to collisions with structures such as power lines and fences (G. Gilbert, pers. comm., White *et al.*, 2006).
- Cramp and Simmons (1977) suggested that the bittern is more shy of disturbance than most species of heron, but can become accustomed to human activities.
- All reedbeds used by breeding bitterns since monitoring began in 1990 were included as 'high sensitivity'. This was thought reasonable to encompass breeding and main feeding areas due to the small and localised, albeit increasing, population.

Bean goose *Anser fabalis*

- The British population of bean geese is small, ca 522 birds (RSPB, 2005, Maciver, 2006) and occurs in just three regularly used locations; the Yare Valley in Norfolk (Wildfowl and Wetlands Trust (WWT), 2006) and Slamannan Plateau in Central Scotland (Maciver, 2006), for Taiga bean geese *A. f. fabalis*, and around North Warren in Suffolk for Tundra bean geese *A. f. rossicus* (RSPB, 2005).
- Although Moorehead and Epstein (1985) identified large wetland birds, such as geese, as being especially susceptible to collisions with wind farms, there is little evidence to support this, with relatively few collisions by geese being reported (Hötker *et al.*, 2006).
- Several studies indicate disturbance displacement of geese by wind turbines (Langston and Pullan, 2003, Hötker *et al.*, 2006), with a maximum reliably detected distance of 600 m for European white-fronted geese (Kruckenberg and Jaene, 1999).
- Bean geese were included on the map, despite not being an Annex I species, due to their small population size, and as the only British population of Tundra bean geese occurs on an unprotected site. Traditional sites holding Tundra bean geese (North Warren and

Minsmere RSPB reserves) were included on the map and buffered by 600 m, based on disturbance distances, and this area classified as 'medium sensitivity', on account of the bean goose's non-Annex I status. Nearly all of the English population of Taiga bean geese occur within the Broadland SPA, which was included as 'high sensitivity', along with the rest of the SPA network.

Marsh harrier *Circus aeruginosus*

- The British marsh harrier population is currently estimated at 364 confirmed and possible breeding pairs (M. Eaton, pers. comm.).
- Marsh harriers typically fly low when foraging (Clarke, 1995) and are likely to be at highest risk of collision during aerial displays, which Cramp and Simmons (1980) suggested typically occur over the nesting territory and up to 1 km beyond. Other observations suggest display flights may occur further from the nest (e.g. 2 km, I. Higginson, pers. comm.; 3 km, J. Day, pers. comm.), and birds also may return from foraging areas by spiralling to a great height before descending to the nest (I. Carter, pers. comm.). Distances of foraging flights vary (e.g. maximum distances of 1.5 km to 3.1 km for males, and 1.4 km to 1.8 km for females in Holland and France (Schipper, 1977), 5-6 km and exceptionally up to 8 km (Glutz von Blotzheim *et al.*, 1971), and 12 km by a male in Cambridge (Clarke, 1995)).
- Birds usually arrive at roost sites singly in low flight (2-3 m, but up to 30 m), before repeatedly circling over the roost at a height of up to 30 - 40 m before settling (Oliver, 2005). Group circling also occurs (Oliver, 2005).
- Breeding locations from the 2005 national survey were plotted, and the area within 1 km of the nest classified as 'high sensitivity', and the additional area within 2 km of the nest as 'medium sensitivity', to represent the area where collision risk is considered highest.
- Roost locations were collated and those containing 1 % or more of the summed counts from all locations included on the map. The area within 1 km of these locations was classified as 'high sensitivity' to account for movement of roost location and the fact that aerial displays may occur above roost sites.

Hen harrier *Circus cyaneus*

- The 2004 national survey estimated the UK and Isle of Man hen harrier population at 806 territorial pairs, a 41 % increase since 1992 (Sim *et al.*, 2007). However, the English population decreased to just 11 territorial pairs during this time (Sim *et al.*, 2007).
- A review of eight wind farm impact studies found good evidence of displacement of hen harriers in only one study (Madders and Whitfield, 2006). However, a recent field study at 12 wind farms and paired control sites in Scotland and England found reduced hen harrier flight activity within 250 m of turbines (Pearce Higgins *et al.*, unpubl.), and preliminary results from studies in Argyll, Scotland and Northern Ireland indicate that local displacement of nesting attempts may occur within 200 - 300 m of turbines (Whitfield and Madders, 2006a).
- Hen harriers typically hunt low over the ground (Watson, 1977). Birds may be more at risk of collision with turbines during display flights, which occur mainly within about 500 m of the nest, but up to 1 km away, or during flights made as newly fledged birds (Madders, 2004). A review of nine wind farm impact studies involving hen harrier found collisions in just three of these, and these were in low numbers (Whitfield and Madders, 2006a).
- Given the extremely small size of the English population, and the fact that its size and distribution is dramatically constrained by persecution, breeding locations spanning a ten-year period (1997 to 2006) were plotted and a precautionary buffer of 2 km applied, as used on the sensitivity map for Scotland (Bright *et al.*, 2006, 2008). This is based on the fact

that foraging flights appear to occur most frequently within 1 km to 2 km of the nest, although they can extend further (e.g. Schipper, 1973, Picozzi, 1978, Martin, 1987, Arroyo *et al.*, 2005).

- Locations of roost sites from the hen harrier winter roost survey 2004 - 05 (the most recent readily available data) containing 1 % or more of the estimated British wintering population (750 birds, Stroud *et al.*, 2001) were buffered by 1 km and this area classified as 'high sensitivity'.

Montagu's harrier *Circus pygargus*

- There were an estimated 13 nesting attempts by Montagu's harriers in Britain in 2007 (M. Thomas, pers. comm.).
- Montagu's harrier in England nest predominantly in cereal crops, due to loss of natural habitat (Clarke, 1996), and nest location can vary between years due to changes in cropping pattern (Clarke, 1996).
- Foraging occurs over long distances (up to 10 km and 15 km from the nest site in males, Schipper, 1973, Clarke, 1996, P. Castle, pers. comm.), and although foraging flights are usually low over the ground, birds can return from forays by spiralling to a considerable height before gliding back to the nest (I. Carter, pers. comm.). Risk of collision may also occur during aerial displays, which occur mainly over the nesting territory, but occasionally further from the nest (up to 3 km, Cramp and Simmons, 1980, up to 4 km, P. Castle, pers. comm.).
- Breeding locations spanning a 10-year period (1997 to 2006) were plotted. Although high flying can occur throughout the foraging range, it was not thought practical to include buffers of foraging ranges, and it is considered likely there will be a higher density of high risk flights near to the nest. Initially, a two-tier approach was planned, with the area within 1 km of breeding locations being classified as 'high sensitivity' and the additional area within 3 km classified as 'medium sensitivity', on the basis of estimated likelihood of aerial displays. However, this was rejected in favour of a 3 km 'high sensitivity buffer' on the map in order to protect the locations of breeding Montagu's harrier.

Black Grouse *Tetrao tetrix*

- The 2005 national black grouse survey showed a decline of 22 % in the UK population since the 1995/6 national survey (Hancock *et al.*, 1999, Sim *et al.*, 2008), and 80 % since the previous population estimate in 1990 (Baines and Hudson, 1995). The English population of black grouse has become fragmented into just two subpopulations in north Northumberland and the North Pennines, and is estimated at 1029 males (Warren and Baines, 2008).
- Black grouse are particularly susceptible to collisions (see Drewitt and Langston, 2008 for review), for example with deer fences (e.g. Catt *et al.*, 1994), and power lines (Bevanger, 1995). Two collision fatalities have also been reported at a wind farm site in Argyll, Scotland (Y. Boles, pers. comm.) and two in Austria (T Dürr in Hötker *et al.*, 2006), these being with the wind turbine towers, rather than the blades (Y. Boles, pers. comm., H. Zeiler, pers. comm.).
- Disturbance of lekking birds has been identified as a problem at some isolated sites (Anon., 2003), although studies on the impact of disturbance on black grouse have had variable results (Miquet, 1986, Miquet, 1990, Zeitler, 2000, Baines and Richardson, 2007).
- Estimates of ranging distances from the lek for males and females vary slightly, but are usually between 1 and 2 km (e.g. Picozzi, 1986, Willebrand, 1988, Cayford, 1993, Warren and Baines, 2004) and management for black grouse is targeted at the area within 1.5 km of the lek (e.g. Anon., 1993). Black grouse leks were buffered by 1.5 km and this area classified as 'medium sensitivity', on account of the black grouse's non-Annex I status.

Stone-curlew *Burhinus oedicnemus*

- The British stone-curlew breeding population, although increasing, remains small and vulnerable, comprising an estimated 350 pairs in 2007 (R. Wynde, pers. comm.).
- It is not known whether there is a significant risk of collision for stone-curlew. Green (unpubl.) modelled potential risk of collision with wind turbines for breeding stone-curlews. Average annual stone-curlew mortality rate is about 0.2 (Green *et al.*, 1997). Green (unpubl.) suggests that, as population trends of species with such low adult mortality can be particularly sensitive to increases in mortality, even a low level of additional mortality would be undesirable, and wind turbines within the foraging ranges of breeding adults may present a problem.
- A study of responses to sources of disturbance in Wiltshire and Hampshire suggested that the species may be more sensitive to disturbance than other species of wader (Taylor *et al.*, 2007). Green *et al.* (2000) found reduced densities of stone-curlews on arable fields up to 3.6 km from the nearest road, with the effect remaining even where alternative habitat was scarce, suggesting disturbance displacement may be limiting population size (Green *et al.*, 2000).
- Radio-tracking breeding stone-curlews in southern England revealed that most activity occurred within 1 km of the nest or chicks (75 %, 83 % and 95 % of active locations during the pre-laying, incubation and chick-rearing periods respectively, Green *et al.*, 2000).
- Nest sites used for the last five years (2003-2007) were plotted and buffered by 1 km and this area classified as 'high sensitivity'. The timespan of data used for this population was decided following consultation with species' experts. Individual wind farm proposals within 3 km of breeding stone-curlew should be assessed for impacts, given the very small population and that foraging can extend up to this distance (Green *et al.*, 2000). The assessment needs to include any new roads associated with the wind farm.

Nightjar *Caprimulgus europaeus*

- The British nightjar population has increased in recent decades, with the 2004 national survey estimating a population of 4606 males, with an estimate for England of 4282 churring males (Conway *et al.*, 2007). However, only a minor recovery in range has occurred, and population declines and range contractions occurred in north-west Britain (Conway *et al.*, 2007).
- Risk of collision would occur mainly during display or migratory flights, and possibly some foraging flights. Recent work monitoring flight heights in Dorset found that all flights were below 20 m (Infinergy, 2008). However, the study site was an area of open habitat, higher flights would be most likely to occur over forestry, and observations of flights over trees at greater heights have been made (e.g. Cramp *et al.*, 1985, S. Wotton, pers. comm.). Thus, wind farms located adjacent to, or in clear-felled pockets within, forestry are most likely to pose a risk of collision for breeding nightjars.
- Human disturbance can affect nightjar densities and breeding success (Murison, 2002, Liley and Clarke, 2003, Woodfield and Langston, 2004, Langston *et al.*, 2007). No information on effects of wind farms on breeding nightjars was found, but disturbance displacement from the turbines themselves, or due to increased human activity or access at the site could affect nightjar populations.
- As the English population is relatively numerous and increasing, a less precautionary approach was taken here than for the Scottish sensitivity map (Bright *et al.*, 2006, 2008). SPAs designated for breeding nightjars were included, plus non-designated sites containing nationally important populations of nightjar in 2004 (see Table 5 and Appendix 3 for details). These were classified as 'medium sensitivity'.

Chough *Pyrrhocorax pyrrhocorax*

- The UK breeding population of choughs has increased in recent decades, with 399 probable and confirmed breeding pairs being located in 2002 (Johnstone *et al.*, 2007). Choughs became extinct in England in the 1970s (Stillman *et al.*, 1998), but a pair bred again in Cornwall from 2002 onwards (Johnstone *et al.*, 2007), with a second pair breeding from 2006 (RSPB, 2008).
- No information was found on likelihood of choughs colliding with wind turbines.
- Information relating to the effects of disturbance on choughs is variable (e.g. Owen, 1989, Rolando and Pattersen, 1993, Cramp and Perrins, 1994, Rolando *et al.*, 2003, Storch and Leidenberger, 2003, Laiolo, 2007).
- A precautionary approach was taken due to the extremely small and localised nature of the English breeding population. All nest sites used since choughs first bred in England in 2002 were buffered by 1 km, and this area classified as 'high sensitivity'. This is based on studies of foraging distances for breeding choughs (Holyoak, 1972, Bullock *et al.*, 1983, Bignal *et al.*, 1996, Cook *et al.*, 1999, Gray *et al.*, 2004, Whitehead *et al.*, 2006), and is in line with SNH guidelines for defining SPA boundaries.
- Roost sites were also included and buffered by 1 km, in part because they may become future nest sites.

Common crane *Grus grus*

- Cranes were formerly widespread in England (Boisseau and Yalden, 1998), but became extinct in about 1600 (British Ornithologists' Union, 1971). A breeding population established again in Norfolk in 1981 (Mathews and Macdonald, 2000). In 2005, there were five to seven pairs of crane at four sites in Britain, six of these pairs being in England (Holling and the RBBP, 2008).
- Large numbers of cranes are killed in collisions with power lines throughout Europe (e.g. European Crane Working Group, 2008). Evidence of effects of wind farms on cranes is sparse, but Moorehead and Epstein (1985) suggested that large wetland birds such as geese and cranes were likely to be particularly susceptible to collisions with wind farms.
- Displacement of cranes from disturbed areas has been reported (e.g. Franco *et al.*, 2000, Nowald, 2001, Leito *et al.*, 2005, 2006), and Meine and Archibald (1996) considered that cranes generally avoid human activity by at least several kilometres. However, nests can be built in relatively disturbed areas in some parts of the range (e.g. adjacent to roads in Germany; P. Newbery, pers. comm.). No information has been published on disturbance displacement of cranes from wind farms.
- Home range sizes vary from about 0.4 to 10 km² (Nowald, 1999, Mathews and Macdonald, 2000, Peske *et al.*, 2003, Röper and Hake, 2003, Leito *et al.*, 2006). These are equivalent to the areas of circles with radii 0.36 to 1.8 km.
- Due to the extremely low size of the English population, a precautionary approach was taken, with nest locations or protected areas containing cranes being buffered by 2 km and this area classified as 'high sensitivity'.

Osprey *Pandion haliaetus*

- The last recorded osprey nesting attempt in England prior to the reintroduction programme occurred in 1842 (Dennis, 1985). The reintroduction programme began at Rutland Water in 1996 (Rutland Osprey Project, 2008) and a pair naturally colonised the Lake District in the same year (Hawk and Owl Trust, 2008). Overall, an estimated 161 - 187 pairs bred in Britain in 2005, with two pairs nesting in England and one in Wales (Holling and the RBBP, 2008).

- Osprey foraging areas can be located quite far away from the nest site. For example, of 14 nests in Scotland in 1976, four were more than 3 km from the nearest fishing water (RSPB in Cramp and Simmons, 1980). Observations have been made of foraging flights up to 10 km or 20 km (Dunstan, 1973), and primary foraging grounds of eight males in North Carolina were about 14 km from the breeding grounds (Hagan, 1986).
- Aerial displays can occur at great heights over the nest site, and foraging flights involve plunging from heights of 5 - 70 m, although heights of 20 - 30 m are most common, from flight or from a perch (Cramp and Simmons, 1980). Pre-construction monitoring at a wind farm in Washington found that 50 % of flights by five osprey were at rotor-swept height (25 - 75 m, Erickson *et al.*, 2003).
- Accounts of effects of disturbance on ospreys vary, with some studies recording higher breeding success in less disturbed areas (Swenson, 1979, Levenson and Koplun, 1984), but other observations of birds breeding successfully despite nest visits, aerial surveys or proximity to roads and houses (Poole, 1981, 1989). These conflicting results may be due to habituation.
- Given the extremely low population size, it was decided to consult experts from the osprey projects to identify breeding locations and areas around these used regularly for foraging. Breeding locations elsewhere from a 10 year time period (1997 to 2006) were also included if more than one breeding attempt had occurred within 1 km during that period (to avoid including transient locations). These were buffered by 2 km, to account for the fact that aerial displays often occur within 1 km or 2 km of the nest (R. Thaxton, pers. comm.). Buffered areas were classified as 'high sensitivity'.

Golden eagle *Aquila chrysaetos*

- The 2003 national survey for golden eagles located 443 breeding pairs in Scotland (Eaton *et al.*, 2007). One pair also formerly bred in England, at Haweswater in Cumbria, from 1969, although just a single male is now present following the death of the female in 2004 (Holling and the RBBP, 2008).
- An estimated 67 golden eagle fatalities occur annually at the 5400 turbine Altamont Pass Wind Resource Area (APWRA) in California (Smallwood and Thelander, 2008). However, a review of wind farm impact studies, focusing mainly on Europe, found just one reported golden eagle collision (Hötker *et al.*, 2006), although collisions have been recorded for other species of eagle.
- Two studies in the USA found no evidence of disturbance displacement of golden eagles due to wind farms (Johnson *et al.*, 2000, Schmidt *et al.*, 2003), but a third study at the Altamont Pass wind farm in California found some evidence of displacement (Hunt *et al.*, 1995). Golden eagles appeared to change their foraging behaviour following construction of wind farm in Argyll, although construction of the wind farm coincided with some local land management changes, which confound the results (Walker *et al.*, 2005).
- In light of the fact that Haweswater was until recently the species' only breeding site in England, and considering the potential for recruitment of a new mate, the territory was included on the sensitivity map. The mean nest location over the past 10 years was taken as the territory centre, and the area within a buffer of 2.5 km around this classified as 'high sensitivity', with the area between this buffer and an outer buffer of 6 km classified as 'medium sensitivity'. This is based on the RIN model, which suggests that golden eagles spend approximately 50 % of their time within 2 - 3 km, and 97 % of their time within 6 km, of the territory centre (McGrady *et al.*, 1997, 2002). The same approach was taken for golden eagles nesting in Scotland, where mapped estimates of foraging ranges extend into England.

Written guidance for unmapped species

As mentioned in the methods, it was not possible to include honey buzzard on the map as data were not available at an appropriate spatial scale. Written guidance is presented, based on the review, in Appendix 2, and the summary review below.

Three other Annex I raptor species were not mapped, merlin and peregrine falcon due to their relatively widespread populations, and red kite because it is increasing in numbers and range in England, such that a map of distribution would be quickly out of date. Although population-level impacts on these species in the UK were considered less likely, particular cases could still give rise to an adverse impact locally or regionally, and written guidance is provided in Appendix 2, with summaries below.

Golden plover and dunlin were not included on the map, outside protected sites or principal sites based on WeBS counts, but written guidance, based on literature reviews for these species, are presented in Appendix 2, with summaries below.

It was not possible to include important feeding areas on cropped land for wintering waterfowl on the map. Guidance relating to these sites, and reference to regional data sources, is provided in Appendix 1, with summaries below.

Honey buzzard *Pernis apivorus*

- The first national survey of honey buzzards in 2000 found a population of 33 confirmed breeding pairs and 36 probable/possible breeding pairs; most of these were in England (Ogilvie, 2003).
- Honey buzzards are generally considered to be quite tolerant to human disturbance, with birds breeding successfully close to sources of human disturbance (Roberts *et al.*, 1999).
- Home ranges are often large, with a study of one to two breeding pairs in Nottinghamshire in 1971 - 1979 estimating that about 70 % of feeding activity occurred within 3 km, and most of the rest within 5 km, of the nest (Irons, 1980). Other flights at distances ranging from 3.5 km (in Germany and France, Münch, 1955, Thiollay, 1967), 'up to 5 km' (Brown, 1967), 'up to 7 - 8 km' (Roberts *et al.*, 1999), as far as 8 - 10 km (C R Tubbs, cited in Cramp and Simmons, 1980) have been reported. In southern England, birds of both sexes are known to forage up to 10 - 12 km away from the nest site (Hampshire Raptor Group, pers. comm., Wiltshire Raptor Group, pers. comm.).
- Display and migratory flights may be at considerable heights (up to several hundred metres, Cramp and Simmons, 1980). There is little published information on flight heights during foraging, but observations of birds returning to the nest with prey in Hampshire and Wiltshire suggest that at least some of these take place several hundred metres above the ground (Hampshire Raptor Group, pers. comm., Wiltshire Raptor Group, pers. comm.). However, collision risk is considered to be of most concern during aerial displays, which are common over the home range (Cramp and Simmons, 1980).
- Due to the scarcity of this species, and in the absence of published information on foraging flight heights, if it had been mapped a relatively precautionary approach would have been taken, with the area within 3 km of the nest classified as 'high sensitivity', and the additional area within 5 km as 'medium sensitivity'.

Merlin *Falco columbarius*

- The UK merlin population was estimated at 1300 pairs following a survey in 1993 and 1994, and had increased or remained stable in all areas where comparison with previous surveys was possible (Rebecca and Bainbridge, 1998). Population estimates from the 2008 national survey are not yet available.

- The UK SPA suite currently holds 426 pairs, or an estimated 33 % of the British breeding population, at 14 sites, four of which are in England, and a further SPA in Dorset is designated for non-breeding merlin, which holds 15 birds or an estimated 1 % of the wintering population (Stroud *et al.*, 2001).
- A review of European wind farm impact studies found one recorded merlin fatality as a result of collision with a wind farm in Germany (Hötter *et al.*, 2006), although it should be noted this study is the result of collated records of reported collisions, rather than controlled searches at wind farm sites.
- Little has been published on the effects of human disturbance on merlin (for review see Ruddock and Whitfield, 2007). Although there is some evidence for effects of disturbance on breeding success, habituation is considered likely, and urban nesting is regularly recorded in the USA and Canada (Ruddock and Whitfield, 2007). In Britain, a 200 – 400 m buffer around nest sites is recommended for forestry workers, although it is unclear on what this is based (Currie and Elliot, 1997). A survey of expert opinion by Ruddock and Whitfield (2007) found a wide range of suggested disturbance distances for merlin, ranging from < 10 m to 300 – 500 m. The upper limit from this survey corresponds roughly to the buffer zones proposed by Currie and Elliot (1997) and in the USA (see Ruddock and Whitfield, 2007 for review).
- Due to their relatively widespread population, merlin were not included on the sensitivity map. This approach should be assessed in light of the results of the recent national survey. Relocation of individual turbines within 200 – 500 m of a merlin nest is recommended, depending on individual site characteristics, with the upper end of the range applicable where turbines are in the line of sight.

Peregrine falcon *Falco peregrinus*

- The 2002 national survey estimated the UK peregrine falcon population at 1426 breeding pairs, or 1514 occupied territories, with an estimated 602 occupied territories in England (Banks *et al.*, in prep.). There have been population increases in most regions in England, as well as considerable range expansion (Banks *et al.*, in prep).
- The UK SPA suite currently holds 109 breeding pairs of peregrine, or 9 % of the British breeding population, in ten sites, two of which are in England (Stroud *et al.*, 2001).
- A review of European literature found two recorded peregrine fatalities as a result of collision with wind turbines, both of these in Belgium (Hötter *et al.*, 2006). However, peregrine flights are likely to occur at rotor height during display and during hunting pursuits so there is potential risk of collision.
- The peregrine appears relatively tolerant to disturbance, although this varies according to factors such as accessibility of the nest site (Ratcliffe, 1980), and habituation to sources of disturbance is considered likely (see Ruddock and Whitfield, 2007 for review of disturbance effects on peregrine falcon). Most USA states have protective buffers for peregrine of 150 – 800 m around active peregrine falcon nests. In Britain, disturbance-free zones for forestry workers of 400 - 600 m (Petty, 1998), and 600 – 1000 m (Currie and Elliot, 1997) have been recommended around occupied nests. Ruddock and Whitfield's (2007) survey of expert opinion suggested that peregrine falcons may be sensitive to disturbance within 500 - 750 m. This is in line with most of the cited literature and previous guidance.
- It was decided not to include peregrine falcon on the sensitivity map due to their relatively widespread and increasing population. Relocation of turbine positions within 400 – 800 m of nest sites is recommended, depending on individual site characteristics, with the upper end of the range applicable where turbines are in the line of sight
- This differs from the more precautionary approach taken for the Scottish sensitivity map (Bright *et al.*, 2006, 2008), where peregrine falcon was included on the basis that

populations in north and west Scotland are decreasing (e.g. 30 % declines since 1991 in the Highlands, Banks *et al.*, in prep.).

Red kite *Milvus milvus*

- The red kite became extinct in Britain by the late 19th century, except for a small population in mid Wales (Carter *et al.*, 1998). A re-introduction programme began in 1989, with release sites in England, Scotland and, more recently Northern Ireland. The English population rapidly expanded in numbers and range, with over 500 pairs estimated in 2008, and 1350 pairs in the UK overall (RSPB, unpubl.).
- The red kite is a qualifying species for just one SPA, which is in Wales and held 15 pairs of red kites, or 9 % of the British breeding population, at the time of the SPA review (Stroud *et al.*, 2001).
- Red kite were one of the most frequent collision fatalities in a review of (predominantly European) wind farm impact studies, with forty red kite fatalities being recorded at wind farms in Germany since 1989 (Hötker *et al.*, 2006). These figures have since been updated, with a total of 91 collisions recorded in the German wind farm collisions database from 2001 to 2007 (T. Dürr, unpubl.). Only a few red kite collision fatalities have been reported at British wind farms, despite some areas of co-occurrence (Percival, 2000, Whitfield and Madders, 2006b, Natural Research Limited, 2008).
- Red kites appear to be quite tolerant of disturbance (Carter, 2001, Ruddock and Whitfield, 2007), although potential negative effects of disturbance on breeding success have been suggested in some cases (Davis and Newton, 1981, Carter *et al.*, 1998, Carter, 2001, Seoane *et al.*, 2003).
- In Britain, disturbance free zones of 300 – 600 m (Currie and Elliot, 1997) and 400 - 600 m (Petty, 1998) around red kite nests have been suggested for forestry workers, and a survey of expert opinion by Ruddock and Whitfield (2007) suggested disturbance tolerance ranges of 10 – 300 m for red kite.
- As the English red kite population is rapidly increasing and expanding its range, it was not thought appropriate to map this species, because the map would be rapidly out of date. However, relocation of turbines within 300 m - 600 m of nest sites, depending on factors such as local breeding density and trends may be necessary, with the upper end of the range applicable where turbines are in the line of sight. Wind farm proposals close to communal winter roosts may cause concern, and a map showing core areas around release locations, which will hold many of the important roost sites, is presented in Appendix 2. Local assessment will be necessary to determine appropriate measures.

Golden plover *Pluvialis apricaria*

- The British breeding range of golden plover has shown substantial contractions, leading to extinctions in some areas (Parr, 1992). The UK golden plover population is estimated at 38 400 – 59 400 pairs (O'Brien in Thorup, 2006).
- The UK SPA suite contains about 26 % of the British breeding population of golden plover in seven SPAs, three of which are in England, and an estimated 22 % of the British wintering population in 22 SPAs, 16 of which are in England.
- Hötker *et al.* (2006) found evidence of disturbance displacement in 72 % of wind farm impact studies reviewed for non-breeding golden (n = 29), with minimum displacement distances ranging from 50 m to 850 m (median = 135, n = 22). Whitfield's (2007) review suggested that the main potential effect of wind farms on waders was disturbance displacement, particularly during the non-breeding season. Due to the possibility that turbine height and disturbance displacement distance may be related (Hötker *et al.*, 2006), Whitfield (2007) suggested that the maximum observed displacement of 850 m should be

used during EIAs involving effects of taller modern turbines on non-breeding golden plover.

- A recent study at 11 wind farm sites and paired control sites found evidence for displacement of breeding golden plover up to distances of 200 m (Pearce-Higgins *et al.*, 2008). Golden plover breeding densities were also lower than expected at wind farm sites (Pearce-Higgins *et al.*, 2008).
- Collision would be most likely during display or foraging flights. However, there are few recorded cases of golden plover collisions with wind turbines (Hötker *et al.*, 2006), and Whitfield (2007) considered that collision risk for waders was generally low.
- Feeding sites used at night differ from those used during the day, both in the breeding (Pearce-Higgins and Yalden, 2003) and non-breeding seasons (Gillings *et al.*, 2005). Both diurnal and nocturnal foraging areas need to be identified during Environmental Impact Assessment.
- The likelihood of a local or national population-level effect, or an effect on the designated population, should be assessed using information on displacement distances, and availability of alternative suitable habitat.

Dunlin *Calidris alpina*

- The UK dunlin population is estimated at 9150 – 9900 pairs (Baker *et al.*, 2006, based on Reed, 1985, although this estimate requires revision), which represents 85 % of the biogeographical population. The British breeding population is of the *schinzii* subspecies (Brown and Grice, 2005), which is listed on Annex I of the EU Birds Directive (EC, 1979). The RSPB's Repeat Upland Bird Survey found widespread population declines of dunlin between surveys in 1980 – 1991 and 2000 – 2002, suggestive of a population decline of at least 50 % over the last 25 years (Sim *et al.*, 2005).
- Around 74 % of the British breeding population of dunlin occur within eight SPAs, two of which are in England (Stroud *et al.*, 2001).
- During the early breeding season, males perform display flights, which are often at turbine blade height (10 – 50 m, Holmes, 1966, Cramp and Simmons, 1983), leading to possible risk of collision. Hötker *et al.*'s (2006) review found no records of collision fatalities (although note dunlin collisions could be overlooked due to their small size and cryptic plumage).
- Previous studies suggested that dunlin were relatively tolerant of disturbance (Thompson and Thompson, 1985, Yalden and Yalden, 1989). However, Finney *et al.* (2004) found that following resurfacing of the Pennine way, which significantly reduced the number of people straying from the path, use of areas within 200 m of the path increased by 50 %. Median distance between the footpath and nests also declined from 175 m to 97 m (Finney *et al.*, 2004). Waders generally appear to be particularly susceptible to disturbance displacement from wind turbines (Hötker *et al.*, 2006, Stewart *et al.*, 2007, Whitfield, 2007).
- The likelihood of a local or national population-level effect, or an effect on the designated population, should be assessed using information on displacement distances, and availability of alternative suitable habitat.

Wintering waterfowl

As outlined in the introduction, it was not possible to map important inland feeding areas, on cropped land, for waterfowl due to lack of a readily available national data source. However, some data sources are available regionally identifying important goose and swan feeding areas outside SPAs, and these are listed in Appendix 1. Written guidance is also provided relating to search areas for feeding areas for waders and geese in the vicinity of coastal and estuarine SPAs, and possible mitigation measures (Appendix 1). Relatively few collisions of geese or swans with wind farms have been recorded (Hötker *et al.*, 2006).

- Geese are particularly sensitive to disturbance from a range of sources. Several studies indicate disturbance displacement from feeding areas close to wind farms (e.g. up to 600 m, Kruckenberg and Jaene, 1999, also see reviews in Langston and Pullan, 2003, Hötter *et al.*, 2006), although there is a wide range of recorded displacement distances, indicating no displacement in some situations.
- Hötter *et al.* (2006) recorded a median minimum distance to the nearest wind turbine of 300 m for geese (mean = 373 m, S. D. = 226 m, n = 13 studies) and 125 m for swans (mean = 150 m, S. D. = 139 m, n = 8 studies).
- Similarly, relatively few collisions have been reported of waders with wind turbines (Hötter *et al.*, 2006, Whitfield, 2007), but waders appear to be particularly susceptible to disturbance displacement from wind turbines, in the non-breeding season (Hötter *et al.*, 2006, Whitfield, 2007), and also during the breeding season (Hötter *et al.*, 2006, Pearce-Higgins *et al.*, 2008, unpubl.).
- On the basis of observed displacement distances, Hötter *et al.* (2006) suggested that important roosting areas for waders and wildfowl should be kept free of wind farms, with a buffer distance of at least 400 m for waterfowl generally, and at least 500 m for goose roosts, being recommended. These distances are of a similar order of magnitude to the maximum reliably observed displacement distance for feeding geese, i.e. 600 m. On a precautionary basis, nearest turbine distances of 400 – 600 m should be maintained from important feeding and/or roosting areas for waterfowl, subject to local topography, line of sight, and existing levels of disturbance/activity. However, see earlier note that a precautionary distance of 850 m may be appropriate for wintering golden plover.
- A review by Vickery and Gill (1999) concluded that most species of goose regularly feed within 10 km of the roost, but prefer sites within 5 km. The review also provides guidance on management of refuges for geese. This guidance provides a valuable tool for planning measures to mitigate against loss of feeding areas for geese due to wind farm development.
- Milsom *et al.* (1998) conducted analyses to inform management recommendations for the creation of grassland feeding areas for waders, which would also prove valuable in planning measures to mitigate against loss of feeding habitat for waders due to wind farm development. The review focused on selection of grass fields used by lapwing and golden plover, which winter on farmland in lowland Britain, but also investigated requirements of estuarine waders which exploit coastal grassland as an alternative, usually supplementary, feeding habitat in winter. For estuarine waders, fields are usually used for feeding to compensate for inadequate intertidal feeding time or food depletion, making fields close to estuaries essential to reduce energetic costs of birds that are already under considerable pressure, notably during cold winter weather. Waders in the two study areas were found to use most fields infrequently, or for only part of each winter, with a very small number of fields being used throughout the winter by a large number of birds. For intertidal species, fields situated within 0.5 km of the coast tended to be used more than those further away.
- There are difficulties with mapping important areas for golden plover and lapwing in winter (see golden plover review), as they may change year-to-year with cropping regime, and diurnal and nocturnal feeding locations also differ (Pearce-Higgins and Yalden, 2003, Gillings *et al.*, 2005). Cropped areas important for wintering golden plover were therefore not included on the map, except for by inclusion of two IBAs proposed because they contain nationally important wintering populations.

Discussion

Applications of the locational guidance

Locational guidance such as that presented here is valuable in the light of the rapid increase in the number of wind farms proposed in England, which is likely to increase further to meet government targets for renewable energy. The map has two main uses. Firstly, it indicates at a national level the regions where bird sensitivities in relation to wind farm development are most likely to be encountered. In England, it seems that many coastal and estuarine areas as well as upland areas in the north of England are particularly sensitive. These are often areas that also have a particularly good wind resource. However, there is also considerable wind resource outside these areas (Figure 2) and local assessment will refine the relative sensitivities at a fine spatial resolution within the mapped areas; the map is not intended to represent 'no go' areas. Spatial planning will be a particularly useful tool to minimise any deleterious impacts of wind farms on the high conservation interest of these areas. Early consultation with the RSPB and Natural England is strongly recommended, to identify likely sensitivities to wind farm development prior to site selection, and, at the scoping stage, to determine suitable pre-application monitoring for important species likely to be present.

Secondly, the map can be used at a regional level, to indicate where wind farm development is less likely to conflict with bird conservation, thereby facilitating the planning process. Current and potential wind farm developments frequently occur outside the most sensitive areas (Figures 1 and 3), and the sensitivity map should prove a useful tool to help maintain this situation as the scale of wind farm development in England increases.

The map is intended for use by decision makers such as Local Authorities, as well as developers during the early stages of the planning process. The map is indicative, rather than absolute, and is not intended to replace site-specific EIA. 'Opportunities' or 'constraints' mapping exercises, incorporating the bird sensitivity map alongside other factors pertinent to wind farm development such as wind speed, technical feasibility and cost could be used by planners and developers to identify preferred areas for wind farm development within a region. A previous example of this was the incorporation of a similar bird sensitivity map for Scotland (Bright *et al.*, 2006, 2008) in the Highland Renewable Energy Strategy (Aquatera, 2006). SNH has produced locational guidance for wind farms in Scotland, incorporating a number of different 'natural heritage sensitivities' (SNH, 2005c), and this is currently being updated to incorporate the Scottish bird sensitivity map.

Significance of effects

When considering impacts of wind farms on birds, the key factor is to assess whether these are likely to result in effects on population size or range. These 'population level' effects can be considered at different spatial scales; both national and regional effects may be important, and for designated sites any adverse effect on the site's conservation objectives should be avoided. Consideration of 'cumulative' (e.g. effects of multiple wind farms, see SNH, 2005a for guidance) and 'in combination' (e.g. effects additional to those from other forms of development) is also necessary. Effects of wind farms at individual sites may become significant across multiple sites even though they may be negligible at the level of the individual wind farm.

Limitations and caveats

The map was created using the best data and information currently available, but cannot be comprehensive, and there are inevitably caveats that apply to its use. Unfortunately, data

deficiency and gaps in survey coverage precluded a distinction between 'low' and 'unknown' sensitivity squares. Many of these areas will not be within the range of the species of interest, or will not contain suitable habitat for them, but some sensitivities may come to light during an EIA and would need to be dealt with accordingly, applying the same criteria as presented here.

Some species were not included on the map because of problems of data access/spatial scale (honey buzzard); data availability (e.g. golden plover and dunlin; inland feeding areas for waterfowl); or rapidly expanding distributions, which might lead either to the map quickly becoming out of date (e.g. red kite) or to inappropriately large areas of high sensitivity on the map as a result of applying buffers to the whole distribution (e.g. merlin, peregrine falcon). For these species, written guidance has been presented instead. This approach will require review as new data on distribution and abundance become available, e.g. from national surveys of merlin in 2008. For red kite, a large number of collision fatalities have been recorded in Germany (Hötter *et al.*, 2006), and so a full literature review was conducted for this species.

For the twelve species mapped separately, data related primarily to breeding birds, although some information on wintering locations (e.g. marsh harrier and hen harrier roost locations) was included. The use of SPAs as surrogates for some species and species groups provided an expedient way of identifying the most important areas for those species that tend to congregate, i.e. colonial breeders and wintering aggregations. However, the proportions included of such species varied, so the addition of IBAs, selected SSSIs and principal sites from WeBS counts was necessary to cover additional sites of national importance.

An important omission of the map is that it was not possible to map key feeding or roosting areas for waterfowl on cropped land, which are generally not included within SPAs. There are some difficulties with mapping these areas, which may change between years with changing cropping patterns and may be at considerable distance from SPAs in some instances. For this reason, a standard buffer around a site was not considered appropriate as it might incorporate large areas of unsuitable and unused habitat. Cropped areas are not generally covered by WeBS counts, which focus predominantly on coastal and estuarine sites, with some inland wetlands (Austin *et al.*, 2008). However, some regional data exist e.g. from local bird recorders, and these have been referred to in Appendix 1, alongside written guidance in terms of identifying search areas for key cropped feeding habitat, and possible measures to mitigate for its loss. The Cropped Habitats Information Project (CHIP; UK SPA Scientific Working Group, 2002) went some way to addressing this problem by identifying SPAs for which important cropped feeding areas occur outside the site boundary for a range of species. A project to collate geographical data on these areas at the national level would be extremely valuable. The sensitivity map will require updates as new data become available.

An additional source of variation between species was the information available on which to base the buffer distances and sensitivity ratings. As there is little information concerning the effects of wind farms on some species, a precautionary approach based on behavioural ecology was taken in some cases. For some species, this information, e.g. disturbance distances and home range sizes, was sparse, or lacking. However, for others, such as golden eagle, radio-telemetry studies have been used to create models predicting the percentage time spent within different distances of territory centres (McGrady *et al.*, 1997, 2002) providing a useful assessment of likely sensitivity. The development of habitat-based models may aid interpolation of species distributions beyond the range covered by sample surveys.

More detailed research is necessary to elucidate the impacts of wind farms on birds, and our assessments of sensitivity should be revised in light of this. In particular, reported collisions may be misleading unless derived from systematic studies with the application of suitable correction factors (Smallwood, 2007). Similarly, recorded displacement distances are of limited value without contextual information to aid interpretation of differing values both within and between bird species. Studies such as that by Pearce Higgins (unpubl.) are valuable in addressing this problem, as is guidance to improve the quality and consistency of post-construction monitoring studies (SNH, 2009a, b).

Locational guidance in other UK countries

Scotland was considered the priority for sensitivity mapping, as it has the highest number of onshore schemes currently being considered for planning approval in the UK, with England having the second highest number of schemes (Table 1, BWEA, 2008). Whilst similar in approach, there are differences between the sensitivity map guidance produced for England and Scotland, mainly arising from differences in bird species composition and population status. The Welsh Assembly Government was the first UK government to provide strategic locational guidance for wind farm developments, having identified seven 'Strategic Search Areas' for wind farms (Welsh Assembly Government, 2005). RSPB Wales provided ornithological data for use in refining these search areas. More recently, the Northern Ireland Department of the Environment (DOE) produced 'Wind Energy Development in Northern Ireland's Landscapes - Draft Supplementary Planning Guidance (SPG)' to accompany draft Planning Policy Statement 18 (www.planningni.gov.uk) which provides some locational guidance for wind farms. RSPB Northern Ireland commented on, and provided ornithological data to inform, this guidance.

Conclusion

Delivery of sustainable expansion of renewable energy is essential to reduce the scale of climate change. Careful location of renewable energy developments, including wind farms, is key to minimising effects on nature conservation interests. It is hoped that strategic guidance, such as that presented here, will provide assistance in this, and facilitate responsible development of onshore wind farms. Increasing demands for wind energy development globally to combat greenhouse gas emissions (IPCC, 2007), make strategic planning for renewable energy an important requirement and sensitivity mapping is a useful tool to assist this process.

Acknowledgements

The authors would like to thank the RSPB and Natural England for funding this project. The following people provided useful comments on an earlier draft of this report: Richard Archer, Graham Austin, Richard Bradbury, Harriet Dennison, Allan Drewitt, Rob Lucking, Martin Kerby, Tim Mackrill, Peter Newbery, Phil Sheldrake, Innes Sim, Dave Thompson, Colin Wilkinson, Simon Wotton and Tim Youngs.

The following groups provided data used in the project: BirdLife International, the British Trust for Ornithology, Forestry Commission, the Game and Wildlife Conservancy Trust, the Hawk and Owl Trust, the hen harrier winter roost counters, the Joint Nature Conservation Committee, Natural England, the Rare Breeding Birds Panel, the Wildfowl and Wetlands Trust. The following individuals were of great help during data supply: Graham Austin, Andrew Dobson, Mark Holling, Andy Musgrove, Phil Warren and Chris Wernham.

We are grateful to the following people, who provided useful input to the sensitivity criteria: John Barrett, Gavin Bloomfield, Dejan Bordjan, Ian Carter, Paul Castle, Ian Court, Tim Cowan, John Day, Andrew Dobson, Andrew Dodd, Nick Droy, Gillian Gilbert, Simon Gillings, Jason Godfrey, Murray Grant, Renny Henderson, Ian Higginson, Fiona Hunter, Kate Jennings, Martin Kerby, Jeff Knott, Michael Meadows, Tim Melling, Jonathan Morley, Claire Mucklow, Peter Newbery, James Pearce-Higgins, Richard Pow, Peter Robertson, Ian Robinson, John Sharpe, Phil Sheldrake, Mike Shurmer, Norman Sills, Innes Sim, Paul St Pierre, Tim Strudwick, Stewart Taylor, Richard Thaxton, Mark Thomas, Dave Thurlow, Kirsty Turner, Phil Warren, Colin Wilkinson, Pete Wilson, Simon Wotton, Robin Wynde and Tim Youngs.

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Tables

Table 1. Number of onshore wind farm developments in the UK (Source: British Wind Energy Association (BWEA), 2008)

	Operational	Stage in planning process		In planning
		Under construction	Consented	
England	75	11	50	82
Northern Ireland	21	4	12	50
Scotland	58	14	54	104
Wales	25	5	10	28
Total	179	34	126	264

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Table 2. Species included on the sensitivity map for which SPA coverage was used to represent distribution¹.

Species	Population size, England	Coverage (Stroud <i>et al.</i> , 2001)
Little egret <i>Egretta garzetta</i>	391 – 443 breeding pairs in 2005 (although 10 - 12 of these occurred in Wales, Holling and the RBBP, 2008).	45 % of the British wintering population within three SPAs, all in England.
Common tern <i>Sterna hirundo</i>	4850 pairs (33 % of UK breeding population, Mitchell <i>et al.</i> , 2004)	46 % of the British breeding population, 11 of the 18 British SPAs are in England.
Arctic tern <i>Sterna paradisaea</i>	3610 pairs (6 % of UK breeding population, Mitchell <i>et al.</i> , 2004)	38 % of the British breeding population, two of the 13 British SPAs are in England.
Sandwich tern <i>Sterna sandivensis</i>	9018 Apparently Occupied Nests (63 % of UK breeding population, Mitchell <i>et al.</i> , 2004)	72 % of the British breeding population, nine of the 13 British SPAs are in England).
Roseate tern <i>Sterna dougallii</i>	36 Apparently Occupied Nests (5 % of UK breeding population, Mitchell <i>et al.</i> , 2004)	88 % of the British breeding population, four of the six British SPAs are in England.
Little tern <i>Sterna albifrons</i>	1541 Apparently Occupied Nests (72 % of UK breeding population, Mitchell <i>et al.</i> , 2004)	67 % of the British breeding population, 23 of the 27 British SPAs are in England.

¹The SPA network, together with sites listed in Table 3, was also used to represent distribution of breeding and wintering waterfowl, and colonially nesting seabirds.

Table 3. Other sites included on the sensitivity map to represent distributions of breeding waders or seabirds or wintering waterfowl.

Site	Selection criteria
Important Bird Areas	Designated for breeding waders or seabirds, wintering waterfowl, or one of the species listed in Table 5.
SSSIs	Breeding Waders and Wet Meadows survey sites: nationally important sites for breeding waders/SSSI feature Sites designated for aggregations of non-breeding birds
WeBS principal sites	Principal site with five year mean of > 20 000 waterfowl, or nationally or internationally important population of any waterfowl species.

Table 4. Goose and swan species included on the sensitivity map for which SPAs were used to represent distribution of roost sites¹

* denotes species not listed on Annex 1.

Species	Coverage
Bewick's swan <i>Cygnus columbianus</i>	99 % of the British wintering population, all 13 British SPAs are in England.
Whooper swan <i>Cygnus cygnus</i>	44 % of the British wintering population, seven of the 20 British SPAs are in England.
*Pink-footed goose <i>Anser brachyrhynchus</i>	82 % of the British wintering population, eight of the 24 British SPAs are in England.
*European white-fronted goose <i>Anser albifrons albifrons</i>	76 % of the British wintering population, all of the eight British SPAs are in England.
*Greylag goose <i>Anser anser</i>	57 % of the British wintering population, two of the 22 British SPAs are in England.
Svalbard barnacle goose <i>Branta leucopsis</i>	79 % of the British wintering population, one of the two British SPAs is the Upper Solway Flats and Marshes, spanning England and Scotland.
*Dark-bellied Brent goose <i>Branta bernicla bernicla</i>	94 % of the British wintering population, all of the 19 British SPAs are in England.
*Svalbard light-bellied Brent goose <i>Branta bernicla rhota</i>	75.9 % of the British wintering population occur within the one British SPA at Lindisfarne, England.

¹Important functionally linked cropped feeding areas need to be identified on a site-by-site basis (see Appendix 1).

Table 5. Species included on the sensitivity map separately

* denotes species not listed on Annex 1.

Species	Population size, England	Data Sources
Bittern <i>Botaurus stellaris</i>	76 booming males (Wotton <i>et al.</i> , 2008)	Annual surveys by RSPB/NE
*Bean goose <i>Anser fabalis</i>		
a) Tundra bean goose <i>A. f. rossicus</i>	53 birds at North Warren (RSPB, 2005)	North Warren and Minsmere RSPB reserve counts
b) Taiga bean goose <i>A. f. fabalis</i>	169 birds at Yare Valley (WWT, 2006)	SPA network
Marsh harrier <i>Circus aeruginosus</i>	364 confirmed and possible breeding pairs	2005 national survey (J. Day, pers. comm.)
	Winter population size not known	Collation of regional winter roost data
Hen harrier <i>Circus cyaneus</i>	11 territorial pairs (Sim <i>et al.</i> , 2007)	Possible/probable/confirmed breeding locations submitted to RBBP 1997 - 2006
	Winter population in England estimated at 300 individuals (Clarke in Lack, 1986, N.B. This differs from figure of 750 birds quoted in results section, which was for Britain)	Hen harrier winter roost survey data 2004 - 2005

Species	Population size, England	Data Sources
Montagu's harrier <i>Circus pygargus</i>	13 breeding pairs (M. Thomas, pers. comm.)	Possible/probable/confirmed breeding locations submitted to RBBP 1997 - 2006
*Black grouse <i>Tetrao tetrix</i>	1029 displaying males (Warren and Baines, 2008)	1-km squares containing lek sites during the GWCT 2006 survey of English Black Grouse
Stone-curlew <i>Burhinus oedicephalus</i>	350 pairs in 2007 (R. Wynde, pers. comm.)	Nest locations from RSPB annual monitoring programme 2003 - 2007
Nightjar <i>Caprimulgus europaeus</i>	Estimated 4282 churring males	2004 national survey (Conway <i>et al.</i> , 2007) Clusters of over 45 churring males (i.e. at least 1 % of the British breeding population) within 2 km of each other.
Chough <i>Pyrrhocorax pyrrhocorax</i>	Two breeding pairs (RSPB, 2008)	Nest and roost locations from RSPB annual monitoring data since 2002
Crane <i>Grus grus</i>	Five confirmed and one probable breeding pair (Holling and the RBBP, 2008)	Nest locations and protected sites containing crane
Osprey <i>Pandion haliaetus</i>	Two breeding pairs (Holling and the RBBP, 2008)	Important nesting and foraging areas, identified by species experts Possible/probable/confirmed breeding locations submitted to RBBP 1997 - 2006
Golden eagle <i>Aquila chrysaetos</i>	One male (Holling and the RBBP, 2008)	Possible/probable/confirmed breeding locations submitted to RBBP 1997 - 2006

Table 6. Species not included on the sensitivity map for which written guidance has been provided¹

* denotes species not listed on Annex 1.

Species	Population size, England	Data Sources
Honey buzzard <i>Pernis apivorus</i>	33 confirmed breeding pairs and 36 probable/possible breeding pairs found by 2000 national survey; most of which were in England (Ogilvie, 2003).	2000 national survey data. Most breeding locations submitted to RBBP are not at fine enough resolution for inclusion.
Merlin <i>Falco columbarius</i>	The UK merlin population was estimated at 1300 pairs following a survey in 1993 and 1994 (Rebecca and Bainbridge, 1998). Figure from 2008 national survey not yet available.	Data from 2008 national survey available shortly.
Peregrine falcon <i>Falco peregrinus</i>	601 occupied territories in 2002 (Banks <i>et al.</i> , in prep.).	2002 national survey data (Banks <i>et al.</i> , in prep.).
Red kite <i>Milvus milvus</i>	Estimated at over 500 pairs in 2008 (RSPB, unpubl.).	RSPB/NE national monitoring programme.
Golden plover <i>Pluvialis apricaria</i>	The UK breeding population is estimated at 38 400 – 59 400 pairs (O'Brien in Thorup, 2006).	Repeat Upland Bird Survey data (Sim <i>et al.</i> , 2005). Surveys of SPAs.
	Wintering population estimated at 580 000 golden plover (Jackson <i>et al.</i> , 2006).	Winter golden plover survey (BTO, see Gillings <i>et al.</i> , 2006).
Dunlin <i>Calidris alpina</i> (<i>schinzii</i> race)	The UK breeding dunlin population is estimated at 9150 – 9900 pairs (Baker <i>et al.</i> , 2006, based on Reed, 1985, although note this estimate requires revision).	Repeat Upland Bird Survey data (Sim <i>et al.</i> , 2005). Surveys of SPAs.

¹See results section and Appendix 2.

Table 7. Summary of sensitivity criteria

Species	Data	Buffer zone	Sensitivity rating
Bittern	Reedbeds used by breeding bittern since 1990.	None	High
Bean goose: Tundra	North Warren and Minsmere RSPB reserves (main traditional sites).	600 m	Medium
Bean goose: Taiga	Broadland SPA.	None	High
Marsh harrier	Possible/probable/confirmed breeding locations from 2005 national survey.	1 km	High
	“	2 km	Medium
Hen harrier	Roost locations from regional recorders containing over 1 % of the total summed count.	1 km	High
	Possible/probable/confirmed breeding locations 1997 - 2006.	2 km	High
Montagu's harrier	Roost locations containing over 1 % of the estimated British wintering population (750 birds, Stroud <i>et al.</i> , 2001).	1 km	High
	Possible/probable/confirmed breeding locations 1997 – 2006.	3 km	High
Black grouse	Lek locations from the 2006 English survey (1-km square resolution).	1.5 km	Medium
Stone-curlew	Nest locations 2003 - 2007.	1 km	High
Nightjar	SPAs and other nationally important sites (defined in Appendix 3).	None	Medium
Chough	Nest locations and roost locations since 2002.	1 km	High
Crane	Nest locations and SPAs containing cranes.	2 km around SPAs/nest locations	High
Osprey	Nest locations and important foraging areas at main breeding sites, identified by species experts.	-	High
	Possible/probable/confirmed breeding locations 1997 - 2006, where more than one breeding attempt had occurred within 1 km during that period.	2 km	High
Golden eagle	Possible/probable/confirmed breeding locations 1997 - 2006	2.5 km	High
	Possible/probable/confirmed breeding locations 1997 - 2006	6 km	Medium

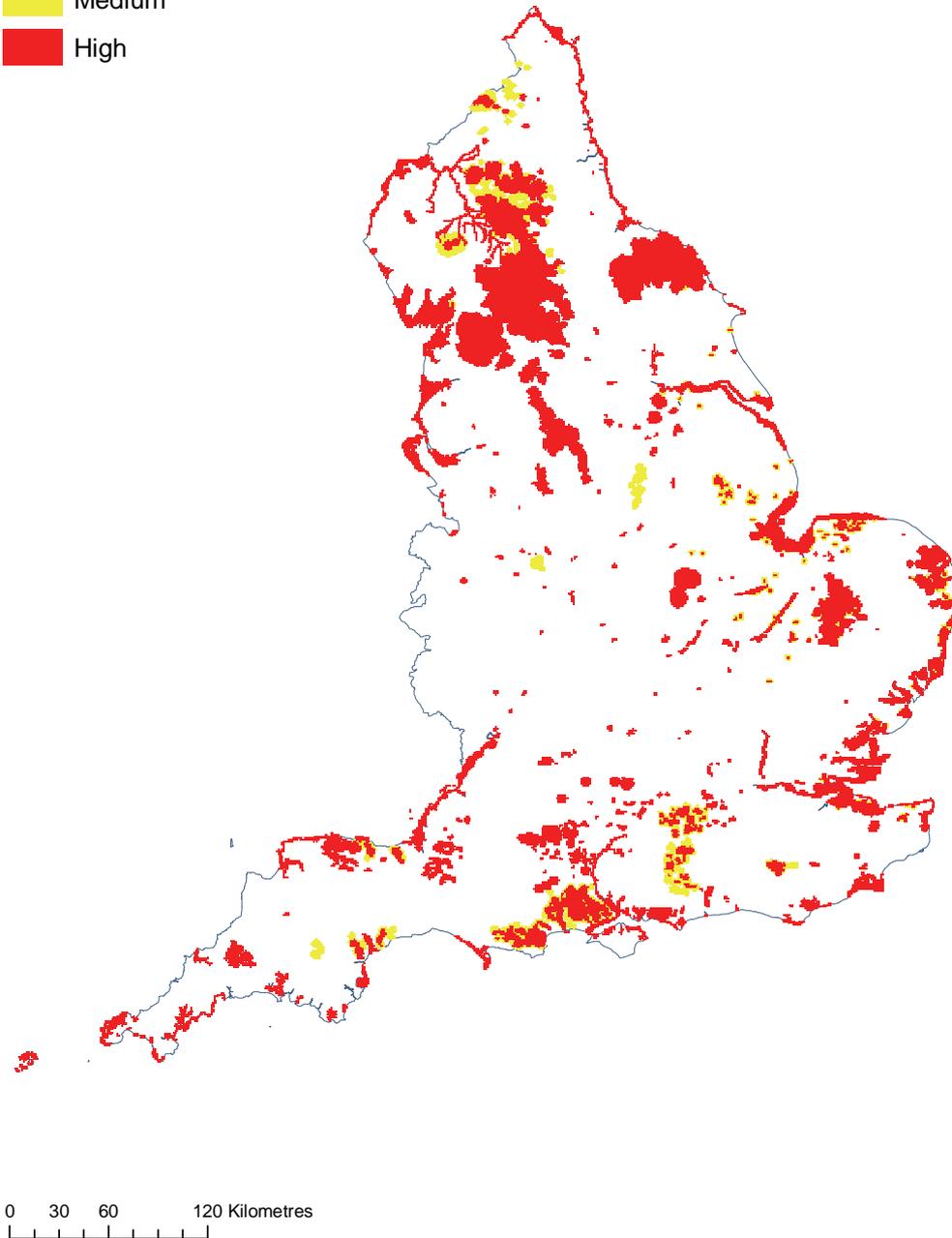
Figures

Sensitivity rating

Unknown

Medium

High



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Figure 1. Map of sensitive bird areas in relation to onshore wind farms in England.

Based on the highest sensitivity rating, for any of the species or sites included, in each constituent 1-km square.

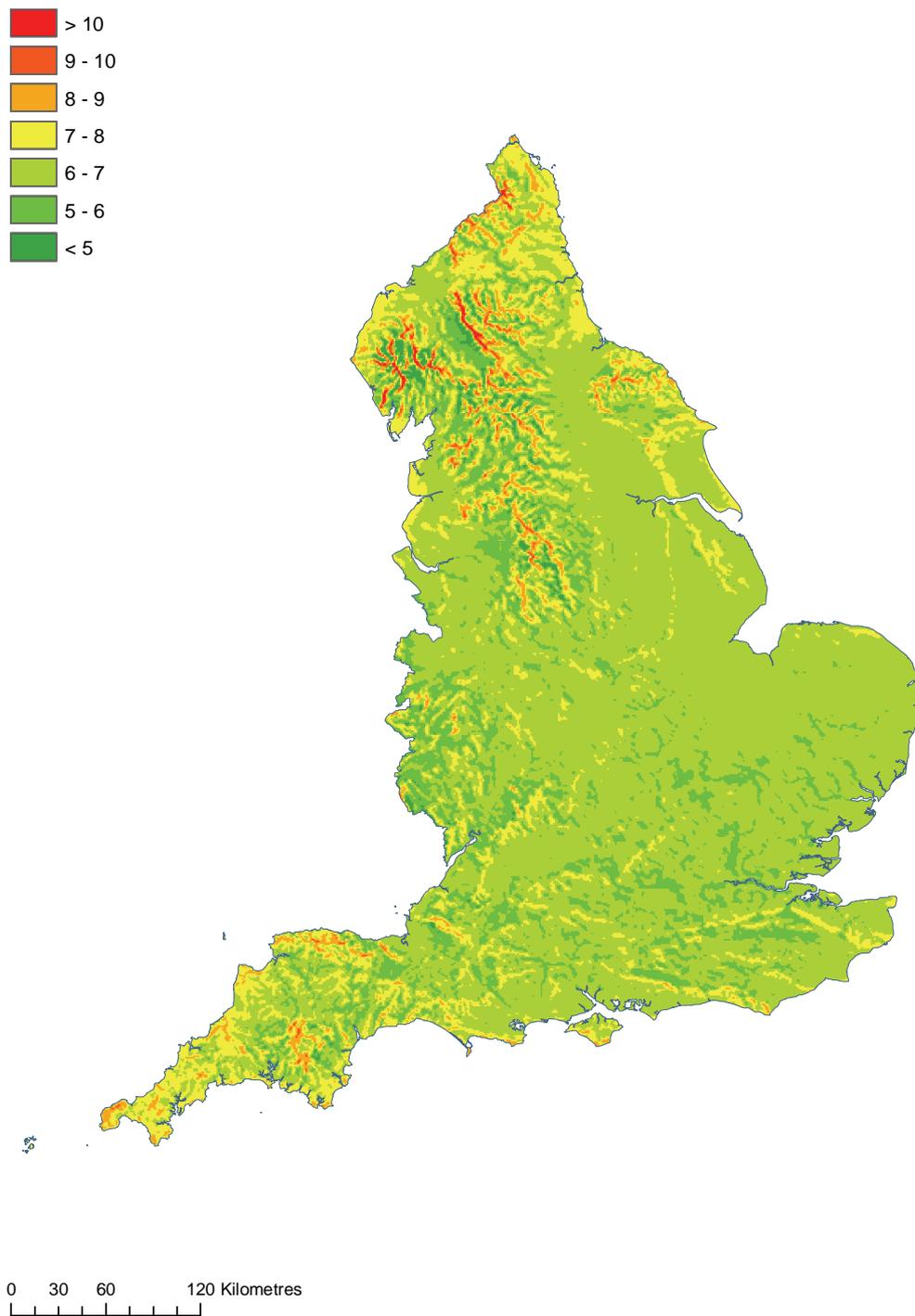


Figure 2. Annual mean wind speed (m/s) by 1-km square at 45 metres above ground level (UK Department of Business, Enterprise and Regulatory Reform (BERR), 2008).

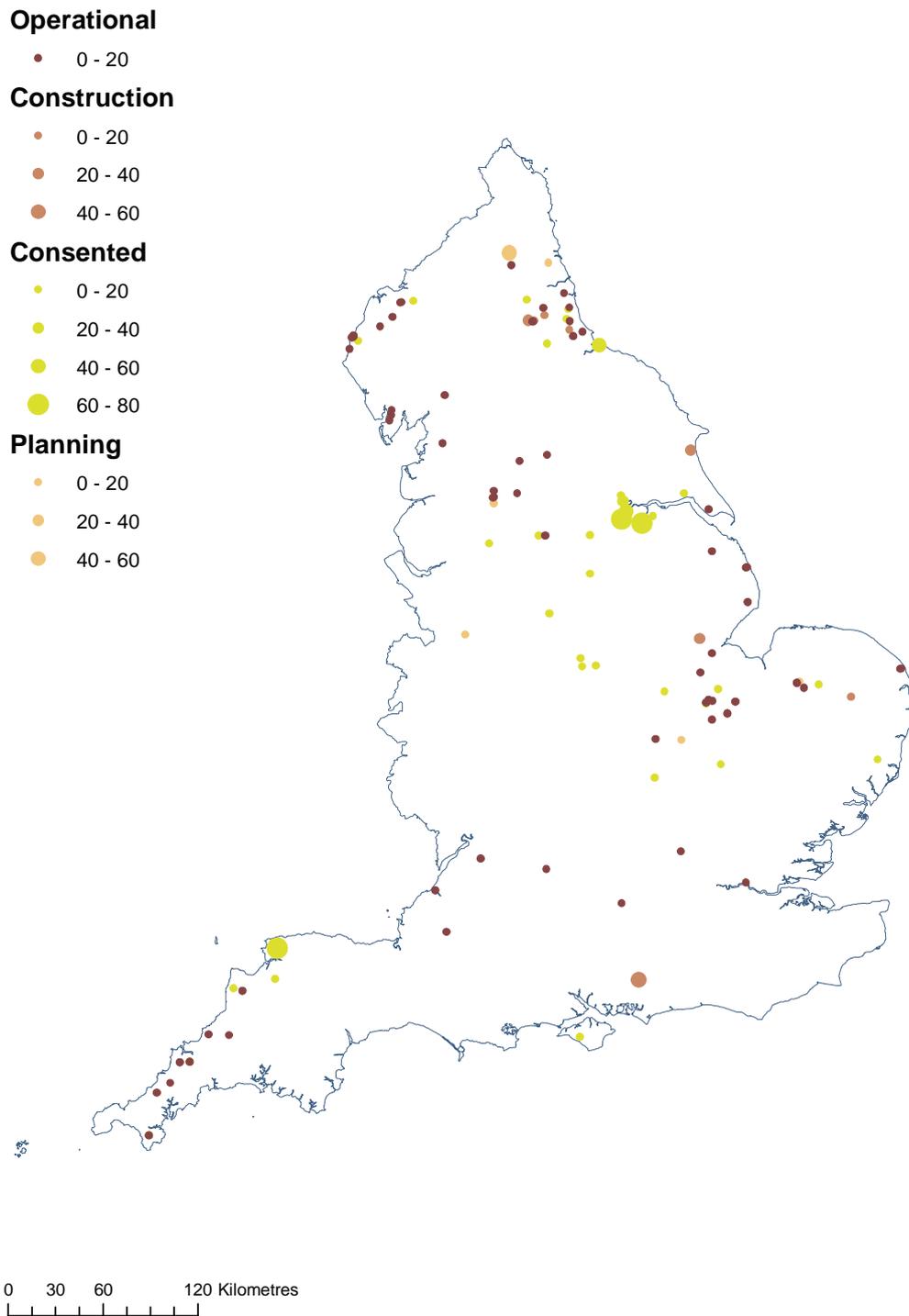


Figure 3. Wind farm developments at different stages in the application process (BWEA, 2008).

Size of circle indicates energy output (MW).

Appendix 1: Cropped feeding areas outside SPAs

SPAs designated for species of goose and swan are largely based on roost locations (Tables A1.1, A1.2, A1.3). However, these species often feed on cropped land in the vicinity of the designated sites, and so could be at risk of collision when flying between roosting and feeding sites, or of displacement from feeding sites. Likewise, for other wintering waterfowl, boundaries of coastal and estuarine SPAs often exclude important inland feeding and/or roosting areas. Few areas of cropped habitat have been included in SPAs to date, and there are difficulties in mapping these areas, which may be at considerable distance from roost sites and may change depending on cropping patterns. However, the Birds Directive requires Member States to protect the habitats of these species outside SPAs, and, more specifically, the Habitats Directive requires assessment of the implications of plans or projects in view of site (SPA and/or SAC) conservation objectives, which can include off-site effects where these may reduce SPA populations (for example through damage or disturbance to 'functionally linked' feeding, roosting or flyway areas). The Cropped Habitats Information Project (CHIP; UK SPA Scientific Working Group, 2002) lists SPAs for each species where important cropped feeding areas occur outwith the SPA boundary, and provides details on habitat types used and degree of site fidelity to particular cropped habitats. However, it does not provide spatially referenced information that could be mapped.

Geese and swans

There is no single national data source for goose and swan feeding areas on cropped land, and, although some good regional datasets are available, data at the national scale are patchy. WeBS counts are focused on estuarine and coastal populations, with only cropped land in the immediate vicinity of these core sites being surveyed (Austin *et al.*, 2008). For this reason, cropped feeding areas for geese and swans have not been included on the sensitivity map, but instead tables are provided here referring to available supplementary data (Table A1.1), along with guidance for identifying search areas for important goose feeding locations (Table A1.4), and for possible mitigation against effects of wind farms in these areas.

Geese and swans may be affected by wind farms due to collision or disturbance displacement. Moorehead and Epstein (1985) identified geese and swans as potentially vulnerable to collision risk due to their large size and low manoeuvrability. However, relatively few collisions of geese or swans with wind farms have been recorded (Hötker *et al.*, 2006, although note that this review is the result of collation of records from a variety of sources, as opposed to controlled corpse searches at wind farms). Geese are particularly sensitive to disturbance from a range of sources, and several studies indicate disturbance displacement from feeding areas close to wind farms (e.g. up to 600 m, Kruckenberg and Jaene, 1999, also see reviews in Langston and Pullan, 2003, Hötker *et al.*, 2006), although there is a wide range of recorded displacement distances, indicating no displacement in some situations. A median minimum distance to the nearest wind turbine of 300 m was recorded for geese (mean = 373 m, S. D. = 226 m, n = 13 studies) and of 125 m for swans (mean = 150 m, S. D. = 139 m, n = 8 studies) by Hötker *et al.* (2006). Similarly, relatively few collisions have been reported of waders with wind turbines (Hötker *et al.*, 2006, Whitfield, 2007), but waders appear to be particularly susceptible to disturbance displacement from wind turbines, particularly in the non-breeding season (Hötker *et al.*, 2006, Whitfield, 2007), but also during the breeding season (Hötker *et al.*, 2006, Pearce-Higgins *et al.*, 2008, unpubl.). On the basis of this, Hötker *et al.* (2006) suggest that important roosting areas for waders and wildfowl should be kept free of wind farms, with a buffer distance of at least 400 m for waterfowl generally, and at least 500 m for goose roosts, being recommended (note these buffers are disturbance free distances for roost sites, rather than applying to feeding areas). These

distances are of a similar order of magnitude to the maximum reliably observed displacement distance for feeding geese, i.e. 600 m. On a precautionary basis, nearest turbine distances of 400 – 600 m should be maintained from important feeding and/or roosting areas for waterfowl, subject to local topography, line of sight, and existing levels of disturbance/activity. Whitfield (2007) suggested a larger precautionary distance of 850 m for wintering golden plover.

Some important cropped feeding areas for wintering waders and wildfowl have been included in the IBA network, which has been included on the sensitivity map (e.g. Tables A1.1, A1.2) Some other data sources e.g. from Local Bird Recorders, have been collated regionally, for example for sensitivity mapping exercises. These have not been mapped, as coverage was considered too patchy, but are listed in tables A1.1, A1.2 and A1.3 where available.

Table A1.1. SPAs with geese listed as a qualifying feature, and sources of information on functionally related feeding areas.

SPA	Species	Country	Data source
Alde–Ore Estuary	Russian white-footed goose	England	
Benfleet and Southend Marshes	Dark-bellied brent goose	England	
Blackwater Estuary	Dark-bellied brent goose	England	
Breydon Water	Russian white-footed goose	England	
Broadlands SPA	Bean goose	England	
	Pink-footed goose	England	
	Russian white-footed goose	England	
Chesil Beach and The Fleet	Dark-bellied brent goose	England	
Chichester and Langstone Harbours	Dark-bellied brent goose	England	Hampshire Brent Goose Strategy Group (2000)
Colne Estuary	Dark-bellied brent goose	England	
Crouch and Roach Estuaries	Dark-bellied brent goose	England	
Dengie	Dark-bellied brent goose	England	
Exe Estuary	Dark-bellied brent goose	England	
Foulness	Dark-bellied brent goose	England	
Hamford Water	Dark-bellied brent goose	England	
Holburn Lake and Moss	Icelandic greylag goose	England	
Humber Estuary SPA	Dark-bellied brent goose	England	Mander <i>et al.</i> (in prep.)
Lindisfarne	Pink-footed goose	England	
	Icelandic greylag goose	England	
	Svalbard light-bellied brent goose	England	
Martin Mere	Pink-footed goose	England	Youngs and Shackleton (2008)
Medway Estuary and Marshes	Dark-bellied brent goose	England	
Morecambe Bay	Pink-footed goose	England	Youngs and Shackleton (2008)
North Norfolk Coast	Pink-footed goose	England	IBA network
	Russian white-footed goose	England	IBA network
	Dark-bellied brent goose	England	IBA network
Poole Harbour	Dark-bellied brent goose	England	
Portsmouth Harbour	Dark-bellied brent goose	England	Hampshire Brent Goose Strategy Group (2000)
Ribble and Alt Estuaries	Pink-footed goose	England	IBA network; Youngs and Shackleton (2008)
Severn Estuary	Russian white-footed goose	Wales/ England	
Solent and Southampton Water	Dark-bellied brent goose	England	Hampshire Brent Goose Strategy Group (2000)

Stour and Orwell Estuaries	Dark-bellied brent goose	England	
Thames Estuary and Marshes	Russian white-footed goose	England	
The Swale	Russian white-footed goose	England	
The Wash	Dark-bellied Brent goose	England	
	Pink-footed goose	England	IBA network; Lucking (2004)
	Russian white-footed goose	England	IBA network
Upper Solway Flats and Marshes	Dark-bellied brent goose	England	IBA network; Lucking (2004)
	Pink-footed goose	Scotland/ England	Youngs and White (2008)
	Svalbard barnacle ¹	Scotland/ England	See footnote

¹Svalbard barnacle goose distribution was not mapped by Youngs and White (2008) as the species occupy fields usually only one or two fields inland from the Upper Solway Flats and Marshes SPA (within the Solway Coast AONB) and spend most time on the estuary itself (please note that the study only related to the English side of the Solway estuary).

Table A1.2. SPAs with Bewick’s swan listed as a qualifying feature, and sources of information on functionally related feeding areas.

SPA	Data source
Arun Valley	IBA network
Avon Valley	
Breydon Water	IBA network
Broadland	
Dungeness to Pett Level 1	IBA network
Lower Derwent Valley	
Martin Mere	Youngs and Shackleton (2008)
Nene Washes	Lucking (2004)
Ouse Washes	Lucking (2004)
Ribble and Alt Estuaries (Phase 2)	Youngs and Shackleton (2008)
Severn Estuary	
Somerset Levels and Moors	
Walmore Common	

Table A1.3. SPAs with whooper swan listed as a qualifying feature, and sources of information on functionally related feeding areas.

SPA	Data source
Broadland	
Lindisfarne	
Martin Mere	Youngs and Shackleton (2008)
Ouse Washes	Lucking (2004)
Ribble and Alt Estuaries	Youngs and Shackleton (2008)
The Wash	Lucking (2004)
Upper Solway Flats and Marshes	Youngs and White (2008)

Search areas for goose feeding areas and mitigation against the effects of wind farms

Vickery and Gill (1999) reviewed the potential for managing grassland for wild geese in Britain in order to create 'refuges' and attract species considered agricultural pests away from other farmland areas. This review includes information on distances between feeding and roosting sites for the six species of wild geese that winter in Britain, and concludes that most species regularly feed within 10 km of the roost, but prefer sites within 5 km (see Table A1.4 for species-specific information). Thus, wind farm proposals within 10 km, and particularly those within 5 km, of estuarine sites or SPAs, that include geese as qualifying species, should assess site use by feeding geese and undertake Vantage Point watches that include pre-dawn/dawn and dusk/post-dusk observations to ascertain flight activity by geese. The review also provides guidance on management of refuges for geese, in terms of the area of land required to support different numbers of geese (by species), as well as information on habitat management. These guidelines provide a valuable tool for planning measures to mitigate against loss of feeding areas for geese due to wind farm development.

Table A1.4. Distance from important roost sites (e.g. SPAs designated for geese) within which EIA searches should be made for important feeding areas likely to be functionally linked to the SPA.

Extracted from table of preferred management options for grassland feeding areas in Vickery and Gill (1999). Excludes light-bellied brent geese, which currently rarely feed on grassland habitats in Britain but are likely to have similar requirements to dark-bellied brent and native greylag geese (Vickery and Gill, 1999).

Goose species	Distance from roost site
Pink-footed goose	Optimal 2 – 5 km maximum 10 km ^a
Greylag goose (Icelandic)	Optimal 2 – 5 km maximum 10 km
White-fronted goose	Optimal < 4 km maximum 9 km
Brent goose	1.5 km inland ^b 4-5 km along coast
Barnacle goose	Optimal < 5 km maximum 7 km
Bean goose Scotland	< 800 m ^c maximum 9 km
Bean goose Norfolk	4 km

^a Although it has been suggested this should be increased to 12 km, as the core foraging site for this species on the North Norfolk Coast extends up to this distance from the roost site (R. Lucking, pers. comm.).

^b Distance inland from coastal roost, feeding areas can be located 4 - 5 km along the coast.

^c Figure is mean distance of used fields from moorland roosts for the Scottish population but this did not influence field selection (Smith *et al.*, 1995).

Waders

Milsom *et al.* (1998) examined the importance of sward, landscape factors and human disturbance when designing grassland feeding areas for waders to inform the re-creation of permanent grassland habitats under agri-environment schemes. These outline management recommendations, in terms of mowing frequency, sward heights, field sizes, boundary features and distance from the coast, for creating feeding habitats for waders, that would also prove valuable in planning measures to mitigate against loss of feeding habitat for waders due to wind farm development. The review focused on selection of grass fields used by lapwing and golden plover, which winter on farmland in lowland Britain, but also investigated requirements of estuarine waders which exploit coastal grassland as an alternative, usually supplementary, feeding habitat in winter, particularly curlew and the Icelandic race of black-tailed godwit. For estuarine waders, fields are usually used for feeding to compensate for inadequate intertidal feeding time or food depletion, making fields close to estuaries essential to reduce energetic costs of birds that are already under considerable pressure, notably during cold winter weather. Waders in both study areas were found to use most fields infrequently, or for only part of each winter, with a very small number of fields being used throughout the winter by a large number of birds. Models showed that for curlew, fields less than 500 m from the sea would be used most, and those over 2.5 km from the sea the least. For oystercatchers, fields nearest the sea were also used most frequently. When choosing fields as feeding areas for waders, larger fields will generally be used more frequently and by greater numbers of birds than smaller fields, and fields enclosed by tall hedges, trees or other barriers will be used less, although this can be mitigated against by selecting broader fields. Fields away from sources of frequent disturbance, such as roads, are preferred. Distance to sea is also important, particularly for intertidal species, with fields situated within 0.5 km of the sea tending to be more attractive than those further away. Management should ensure a mean sward height of less than 10 cm during the winter. Maintenance of relatively high groundwater levels is also important to maintain a moist feeding substrate (e.g. Eglington *et al.*, 2008).

Most waders feed in coastal or estuarine areas in the winter, with the exception of golden plover and lapwing, which are widely distributed on lowland farmland in winter (Cramp and Simmons, 1983, Stroud *et al.*, 2004). There are difficulties with mapping important areas for these species (see golden plover review), as they may change year-to-year with cropping regime, and diurnal and nocturnal feeding locations also differ (Gillings *et al.*, 2005). Cropped areas important for wintering golden plover were therefore not included on the map, except for by inclusion of two IBAs proposed as they contain nationally important wintering populations.

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Appendix 2: Written guidance for species not included on the map

1. Peregrine falcon

The UK peregrine falcon *Falco peregrinus* population reached a low point of about 360 breeding pairs in 1963 (Ratcliffe, 1972), but has subsequently shown dramatic recovery in terms of population size and range (Banks *et al.*, in prep.). The 2002 national survey estimated the UK peregrine falcon population at 1426 breeding pairs, or 1514 occupied territories, with an estimated 602 occupied territories in England (Banks *et al.*, in prep.). Regional trends have varied, but populations in most parts of England, southern Wales and southern Scotland have increased, and there has also been considerable range expansion in England (Banks *et al.*, in prep.). The peregrine falcon is an Annex I species (EC, 1979) and is on the amber list of conservation concern in the UK (Gregory *et al.*, 2002). The UK SPA suite currently holds 109 breeding pairs of peregrine, or 9 % of the British breeding population, in ten sites, two of which are in England (Stroud *et al.*, 2001). Some birds, particularly females and juveniles, move away from upland breeding areas in autumn.

A review of European literature found two recorded peregrine falcon fatalities as a result of collision with wind turbines, both of these in Belgium (Hötcker *et al.*, 2006). However, it should be noted that this review is the result of collation of records from a variety of sources, as opposed to controlled corpse searches at wind farms.

The following text is largely based on a review of disturbance by Ruddock and Whitfield (2007), which was of great help in producing this guidance. The peregrine appears relatively tolerant to disturbance, although this varies according to factors such as accessibility of nest site (Ratcliffe, 1980), and habituation to sources of disturbance is also considered likely (Ruddock and Whitfield, 2007). As the population recovers, urban-nesting is increasing, with 4.4 % of nests located during the 2002 national survey being on urban structures (N. Dixon pers. comm., Banks *et al.*, 2004 cited in Ruddock and Whitfield, 2007). However, disturbance was cited as the cause of failure of 19 % of breeding attempts in Cumbria (Horne and Fielding, 2002), although this probably included both deliberate and accidental disturbance.

Behavioural observation of peregrine falcons have been conducted without eliciting a noticeable change in behaviour at distances of > 100 m (Carlier and Gallo, 1994), 8 m (using a hide, Rosenfield *et al.*, 1995), 200 – 400 m (Jenkins, 2000), 300 m (Palmer *et al.*, 2000, 2003), 70 – 200 m (Ruddock, 2006) and 400 m (Wildlife Commission, 2006). Waterskiing was tolerated at just 50 m away with no noticeable effects, but anglers stopping proved more disruptive (Olsen and Olsen, 1980); this difference is probably related to perceived threat. Successful breeding has occurred in studies involving nest visits at three-day intervals (Bradley and Oliphant, 1991, Olsen and Tucker, 2003). However, rock climbing can suppress breeding success and occupancy (Ratcliffe, 1972, Snow, 1972, Mitchell, 1979, Mearns and Newton, 1988, Lanier and Joseph, 1989). Flights by jet aircraft 150 m above nests affected activity budgets and nest attendance, but food-provisioning rates were unaffected (Palmer *et al.*, 2003). In a separate study, birds were tolerant of aircraft noise from 85 – 141 dB, and although flights under 980 m elicited a flight response in some birds, successful breeding still occurred (Eillis *et al.*, 1991). A study of the effect of blasting regimes at mining quarries on a similar species, the prairie falcon, involving noise levels up to 140 dB, found that some blasts resulted in flights and cessation of incubation and brooding, but usually for a short period (average = 1.4

minutes). No effects of blasts in the distance range 560 m - 1000 m were observed (Holthuijzen *et al.*, 1990). Note, in Britain, quarrying regimes usually involve much less frequent blasts (M. Ruddock, pers. obs. in Ruddock and Whitfield, 2007), but this information is provided for information on noise disturbance. Holmes *et al.* (1993) found that in winter, prairie falcons flushed at 17 - 180 m in response to humans, and 44 - 85 m in response to vehicles.

Most USA states have protective buffers for peregrine of 150 – 800 m around active nests although some extend as far as 4800 m (see review in Ruddock and Whitfield, 2007). The USA Fish and Wildlife Service guidance recommends assessment of disturbances occurring within 400 m (non-breeding season) and 800 m (breeding season) of peregrine nest sites (US Fish and Wildlife Service (USFWS), 1982). Other reviews recommend restriction of activities (e.g. hiking) within 400 – 800 m of nest cliffs (Eillis, 1982), exclusion of forestry activities up to 1600 m (Ellis, 1982), exclusion of human activity within 800 m (Richardson and Miller, 1997), restriction of disturbance and development within 200 - 800 m (Olsen and Olsen, 1980) and an exclusion zone for rock climbers of 200 m (Brambilla *et al.*, 2004). In Poland, a protection zone of 200 m is enforced, increasing to 500 m between January and July (<http://free.ngo.pl/eagle/ochrona.htm>). In Britain, disturbance-free zones for forestry workers of 400 - 600 m (Petty, 1998), and 600 – 1000 m (Currie and Elliot, 1997) have been recommended around occupied peregrine nests.

Ruddock and Whitfield's (2007) survey of expert opinion suggested that peregrines may be sensitive to disturbance within 500 - 750 m of the nest. This is in line with most of the cited literature and previous guidance, suggesting that buffer-free zones of 400 – 800 m might be appropriate, depending on site characteristics and source of disturbance.

Peregrine have not been included on the sensitivity map due to their relatively widespread population, which is increasing in range and numbers in England. Relocation of turbines within 400 – 800 m of nest sites is recommended, depending on individual site characteristics, with the upper end of the range applicable where turbines are in the line of sight. This differs from the more precautionary approach taken for the Scottish sensitivity map (Bright *et al.*, 2006, 2008), where peregrine falcon were included on the basis that populations in north and west Scotland are decreasing (e.g. 30 % declines since 1991 in the Highlands, Banks *et al.*, in prep.).

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2. Honey buzzard

Introduction

The honey buzzard *Pernis apivorus* has a large global range, breeding throughout the temperate and boreal regions of Europe and eastwards into boreal Asia (Cramp and Simmons, 1980). It is a long-distance migrant, over-wintering in west, central and southern Africa (Cramp and Simmons, 1980, Hagemeyer and Blair, 1997). Europe accounts for over 75 % of the honey buzzard's breeding range (BirdLife International, 2004), although the distribution is quite patchy across most of this range (Stroud *et al.*, 2001). The breeding population is generally stable globally and in Europe, although there were declines in some countries such as Sweden and Finland between 1990 and 2000 (BirdLife International, 2004).

The size of the UK population of honey buzzards has previously not been well documented as information on breeding birds was frequently withheld due to fears over persecution and egg collecting (Stroud *et al.*, 2001). However, the first national survey of honey buzzards was conducted in 2000 and found a population of 33 confirmed breeding pairs and 36 probable/possible breeding pairs (Ogilvie, 2003). By country, this constituted 24 confirmed and 21 probable/possible breeding pairs in England, five confirmed and five probable/possible breeding pairs in Wales and four confirmed plus ten probable/possible breeding pairs in Scotland (Ogilvie, 2003). The distribution of the honey buzzard in the UK is thus widespread but very fragmented, with the species usually occurring as isolated pairs or small groups, but within at least six counties in Britain (Gibbons *et al.*, 1993). The species does not occur regularly in Northern Ireland (Stroud *et al.*, 2001).

There is little reliable historic information on honey buzzard breeding numbers, and there has been considerable debate over honey buzzard population trends. The UK is at the edge of the honey buzzard's European breeding range, and the species is considered likely to have always been a rare and scattered breeder here (Stroud *et al.*, 2001). The species may have been extinct between 1900 and 1910 due to illegal killing, since recovering due to increased protection (DETR/JNCC Raptor Working Group, 2000). A review of population trends of breeding birds in the UK since 1800 suggested that the honey buzzard was in moderate decline from 1800 to 1939, with a subsequent slight increase between 1940 and 1969 (Gibbons *et al.*, 1996). This contrasts to Cramp and Simmons (1980), who considered that the honey buzzard was never numerous, but was perhaps more widespread in the 19th century. Comparisons cannot really be made between accounts of the honey buzzard in the 1968 - 72 and 1988 - 91 breeding bird atlases, because the information on which they were both based was incomplete (Gibbons *et al.*, 1993).

Roberts *et al.* (1999) suggested that, although information is sparse, the increase from 'probably less than 10 pairs' cited in Brown (1976) to 'possibly 20 or more pairs in 1988' (Spencer *et al.*, 1990), to 'up to 30 pairs summering but not necessarily breeding' (Batten *et al.*, 1990) reflects a genuine increase in numbers in recent decades. Roberts *et al.* (1999) attribute this increase to the colonisation of upland forests in western and northern Britain, and the increased breeding success now that plantations have entered a second rotation. Extensive clear-felling and presence of variable-age timber has also created new habitat for the honey buzzard (Roberts *et al.*, 1999). However, more recently, Combridge *et al.* (2003) disputes this, and suggests that when considered in full the statements of Brown (1976) and Spencer *et al.* (1990) do not in fact provide evidence of a genuine increase in recent decades. Numbers in the New Forest have fluctuated over the past fifty years, with Clark and Eyre (1993) reporting up

to four pairs during 1954 - 60; up to nine pairs during 1961-80; and two or three during 1981 - 92. Wiseman (2004) added that 'The period 1993 - 2002 saw a recovery to pre - 1981 population levels, although in 2003 numbers roughly halved and were then similar to those during 1981 - 92'.

The honey buzzard is an Annex I species (EC, 1979) and is on the amber list of conservation concern (Gregory *et al.*, 2002). The SPA review (Stroud *et al.*, 2001) estimated that two pairs of honey buzzards, or 13 % of the British breeding population (based on an estimate of 16 pairs made by the DETR/JNCC Working Group, 2000), occur within the New Forest Special Protection Area, although see note above about population fluctuations in this area.

Breeding ecology

Breeding habitat

Roberts *et al.* (1999) identified three main honey buzzard breeding habitats; mixed deciduous forests in the lowlands of southern England; central hill country with mixed farmland/woodland; and upland, even-aged coniferous plantations. Nests are in large forest trees, especially broad-leaved (such as beech and oak) or coniferous (especially pines), but also other species (Kostrzewa, 1998), with Douglas fir often being favoured in southern England (Hampshire Raptor Group, pers. comm., Wiltshire Raptor Group, pers. comm.).

Breeding system

There are few published studies of honey buzzards in Britain. Irons (1980) studied one to two pairs nesting in Nottinghamshire during 1971 - 1979. In nine years, only four young were known to have fledged. Irons (1980) suggested that this low productivity may have been due to poor weather conditions leading to reduced availability of the honey buzzard's principal prey, wasps. However, Roberts *et al.* (1999) dispute this, as they found high breeding success among honey buzzards in upland Britain, which were nesting at higher altitudes with colder, wetter weather. A generally higher breeding success was found in Roberts *et al.*'s (1999) study of 52 nests in England and Wales between 1989 and 1997, with a mean 1.66 young fledged per attempt. In Wiltshire, frogs and fledgling birds form an important part of the diet in years with low wasp numbers (Wiltshire Ornithological Society (WOS), 2007).

Birds usually arrive at the breeding grounds from mid-May (mean = May 18th, Irons, 1980) and form a monogamous bond of at least seasonal duration. A clutch of two eggs is laid (mean = 1.97, range = 1 - 3, n = 73, Kostrzewa, 1998), with first egg dates in Roberts *et al.*'s (1999) review ranging from 21st May to 8th June (median = 29th May, n = 23). Replacement clutches are rare (Cramp and Simmons, 1980, Roberts *et al.*, 1999). Estimates of the incubation period vary, from around 32 days (Irons, 1980) to 40 - 44 days (Kostrzewa, 1998). The female broods the young almost continually for 7 - 10 days, and stays nearby for a further 7 days (Kostrzewa, 1998), with the male feeding young directly and occasionally helping to brood them (Irons, 1980). Young may still be found on the nest at 35 days, and both parents tend young for at least 2 - 3 weeks after fledging (Irons, 1980).

Home ranges and site fidelity

Site fidelity

Honey buzzards return to the same nesting area each year, within which they have several nests that are used in rotation (Tubbs, 1974, Brown, 1976, Roberts *et al.*, 1999). A study of 52 nests in England and Wales from 1989 to 1997 found that some nests were reused the following year, with one being used for four years in succession, and seven others used in two successive years (Roberts *et al.*, 1999). Four nests, not reused in the following year, were reused in subsequent years (Roberts *et al.*, 1999). However, incidents of pairs not reusing a

nest for periods up to 17 years have also been reported (R Khan in Roberts *et al.*, 1999). A study in Germany found that new nests were built in 74 % of cases (Kostrzewa, 1998).

Of 18 nesting attempts in England and Wales, average distance moved between years in the New Forest was 2 km (maximum = 4.5 km), but just 250 m in other regions (maximum = 750 m). Re-nesting is considered extremely rare, but one pair in the uplands re-nested 500 m away following a failed attempt (Roberts *et al.*, 1999).

Home range

Honey buzzards often have very large home ranges, which are probably related to wasp density (Gibbons *et al.*, 1993), and frequently overlap with the range of one or more other pairs (Kostrzewa, 1998). Within the home range, the nesting territory is not always well-defined when local density is low (C Tubbs in Cramp and Simmons, 1980). Most pairs are hostile within a wide radius of the nest; with distances of up to 1500 m reported where conspecific intruders are involved, and 300 - 500 m with other species of diurnal birds of prey (Thiollay, 1967).

Cramp and Simmons (1980) state that estimates of hunting range sizes in Europe are mostly around 10 km², but can be up to 36 km² (equivalent to the areas of circles with radii 1.8 km and 3.4 km respectively). Pairs may occupy home ranges of up to 40 km² (equivalent to the area of a circle with radius 3.6 km) in the course of the breeding season, but with much of this area visited only occasionally, and the feeding areas of two or more pairs commonly overlapping (Cramp and Simmons, 1980). Kostrzewa (1998) reviewed data from more recent European studies and found home range sizes of 250 - 1800 ha (0.25 - 1.8 km², equivalent to the areas of circles with radii 0.3 km - 0.8 km) for males and 250-1500 ha (0.25 - 1.5 km², equivalent to the areas of circles with radii 0.3 km - 0.7 km) for females.

Roberts *et al.* (1999) reported that adults may travel up to distances of 7 - 8 km to look for bee or wasp nests. These long distance flights are generally made by males, with females foraging closer to the nest (Roberts *et al.*, 1999). In Germany and France, birds forage up to 3.5 km from the nest (e.g. Münch, 1955, Thiollay, 1967), but elsewhere distances of up to 5 km have been reported (Brown, 1976). A study of one to two breeding pairs in the Nottinghamshire in 1971-1979 (Irons, 1980) estimated that about 70 % of feeding activity occurred within about 3 km of the nest, with most of the rest occurring within 5 km, although it is not clear what these estimates were based on. A study of a small British population in the 1960s and 1970s found that most foraging appeared to occur within an area of 4 - 6 km² around the nest during incubation, but during the nestling period they travelled as far as 8 - 10 km (C R Tubbs, cited in Cramp and Simmons, 1980). In southern England, birds of both sexes are known to forage up to 10 - 12 km away from the nest site (Hampshire Raptor Group, pers. comm., Wiltshire Raptor Group, pers. comm.). In August 2008, two adult honey buzzards in Wales were fitted with radio-transmitters as part of a five-year study; this is likely to provide valuable information about home range sizes and foraging activity in Britain (Just Ecology, 2008).

The observed variation in home range size is probably at least partly attributable to the distance of suitable feeding grounds from the nest, for example pairs breeding in woods isolated from agricultural land or heathland may need to travel long distances (Cramp and Simmons, 1980).

Nearest-neighbour distances

Nests are usually considerable distances apart (Dementiev and Gladkov, 1951), e.g. a review of 52 nests in England and Wales between 1989-1997 found a minimum distance of 6 km

between active nests (two incidences), with two further instances of pairs nesting 8 km apart (Roberts *et al.*, 1999). A small British population studied in the 1960s and 1970s found nearest-neighbour distances were never less than 3 - 5 km (C R Tubbs, cited in Cramp and Simmons, 1980). Large nearest-neighbour distances have also been reported elsewhere in Europe (Kostrzewa, 1998) although on rare occasions, occupied nests can be just a few hundred metres apart (Glutz von Blotzheim *et al.*, 1971). High breeding densities are likely to be correlated with wasp availability (Mebs and Link, 1969).

Collision risk

Cramp and Simmons (1980) describe the honey buzzard as 'noticeably agile amongst trees'. The honey buzzard is less aerial than most diurnal birds of prey, but during display and on migration uses airspace up to several hundred metres, exploiting thermals (Cramp and Simmons, 1980). There is little published information on flight heights during foraging, thus the likelihood of risk of collision during foraging flights is uncertain. Observations of birds returning to the nest with prey in Hampshire and Wiltshire suggest that at least some of these foraging flights take place several hundred metres above the ground (Hampshire Raptor Group, pers. comm., Wiltshire Raptor Group, pers. comm.). However, collision risk is considered to be of most concern during display flights such as high circling and sky dancing.

High circling is the most common display flight in the honey buzzard, being common over the home range or territory, where a single bird or a pair perform frequent ascents by soaring to great heights in tight spirals (Cramp and Simmons, 1980). The main other display flight is sky dancing (also termed 'wing clapping' or 'butterfly flight'). Cramp and Simmons (1980) describe sky dancing as a series of dives and climbs, with wings held vertically upwards and shaken loosely at the top of the cycle, following on from high circling by the male, or both birds, and lasting up to 30 minutes. Irons (1980) also observed sky dancing displays in the Nottinghamshire lasting for up to 15 minutes, especially early after arrival. It is not clear how common sky dancing is, with Cramp and Simmons (1980) stating that prolonged periods of mutual high circling (and sky dancing) occur over the nest site and feeding grounds particularly in the middle and later parts of the nesting period, being most frequent from mid-July, whilst Roberts *et al.* (1999) observed sky dancing only infrequently. Non-breeding pairs display more persistently than breeding pairs (Hampshire Raptor Group, pers. comm., Wiltshire Raptor Group, pers. comm.).

Disturbance

The species was previously believed to be very sensitive to human disturbance (Cramp and Simmons, 1980, Gibbons *et al.*, 1993). However, Roberts *et al.* (1999) suggest that this is not the case, with no incidences of desertion or predation being recorded during their study of 52 nests between 1989 and 1997, and breeding success of visited pairs remaining high in subsequent years. Of 48 nesting attempts, 24 (50 %) were in trees adjacent to rides, paths or clearings, and 37 (77 %) within 20 m of them (Roberts *et al.*, 1999). The furthest distance between an access route and nest tree was 150 m (Roberts *et al.*, 1999). Of 12 nests immediately adjacent to used paths, only one was believed to have failed due to human disturbance (Roberts *et al.*, 1999). Rob Bjijsma, based on visits to hundreds of honey buzzard nests, considers the honey buzzard to be more tolerant of human activity than any other raptor species (cited in Roberts *et al.*, 1999). Kostrzewa (1987), in a study of the honey buzzard, common buzzard and goshawk, also suggested that the honey buzzard is fairly tolerant of disturbance.

Sensitivity criteria

The majority of records submitted to the Rare Breeding Birds Panel were not at sufficiently fine resolution to be included on the sensitivity map, and were at the 10 km square or site

level. Therefore, it was not possible to include this species, but the literature review has been included for information and to provide a basis for future additions to the sensitivity map were these data to become available.

Due to the scarcity of this species, and in the absence of published information on foraging flight heights, if it had been mapped a relatively precautionary approach would have been taken, with the area within 3 km of nests classified as 'high sensitivity', and that within 5 km as 'medium sensitivity'.

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3. Merlin

The UK merlin *Falco columbarius* population was estimated at 1300 pairs following a survey in 1993 and 1994, and had increased or remained stable in all areas where comparison with previous surveys was possible (Rebecca and Bainbridge, 1998). Although the British breeding population is thought to have declined from the 1950s until the early 1990s, previous surveys have not been sufficiently complete to allow for population trends to be assessed (Gibbons *et al.*, 1993). A national survey of merlin has been conducted in 2008, but population estimates are not yet available from this survey. The merlin is an Annex I species (EC, 1979) and is on the amber list of conservation concern in the UK (Gregory *et al.*, 2002). The UK SPA suite currently holds 426 pairs, or an estimated 33 % of the British breeding population, in 14 sites, four of which are in England (Stroud *et al.*, 2001). In winter, the breeding population is joined by most of Iceland's breeding population (Stroud *et al.*, 2001). The British breeding population leave upland areas and move to inland lowland and coastal areas, where they are widely scattered at low densities (Stroud *et al.*, 2001). Roost sites can be communal and are often shared with hen harrier (Stroud *et al.*, 2001). The wintering population is estimated at 1300 individuals (Stroud *et al.*, 1990), and one SPA is designated for non-breeding merlin (Dorset Heathlands SPA in England) and this holds 15 birds, or an estimated 1 % of the wintering population.

A review of European wind farm impact studies found one recorded merlin fatality as a result of collision with a wind farm in Germany (Hötcker *et al.*, 2006). However, it should be noted that this review is the result of collation of records from a variety of sources, as opposed to controlled corpse searches at wind farms.

The following text is largely based on a review of disturbance by Ruddock and Whitfield (2007), which was of great help in producing this review. Little has been published on the effects of human disturbance on merlin (Ruddock and Whitfield, 2007). Breeding merlins can be displaced by recreational facilities such as camping and picnic areas (James *et al.*, 1989), and higher reproductive success has been recorded in rural populations (Sodhi *et al.*, 1992). Flushing distances of 17 – 80 m in response to pedestrians and 44 – 85 m in response to vehicles have been recorded, with over 90 % of birds flushing in response to pedestrians, but just 38 % in response to vehicles (Holmes *et al.*, 1993). This is likely to be associated with perceived threat. As with all studies of disturbance, interpreting results can be difficult, for example tree-nesters can perceive disturbance at greater distances than birds nesting in long ground vegetation, but respond at shorter distances (e.g. Watson and Pierce, 1998, although see González *et al.*, 2006). Females that had experienced predation of eggs or young reacted less to nest inspections, showing that reduced response is not necessarily a sign of habituation (complicated by the fact that the lower defence behaviour could have led to the original predation incident, Wiklund, 1990). Birds may be particularly prone to desertion just prior to egg laying (Wiklund pers. comm. in Ruddock and Whitfield, 2007). However, habituation is considered likely, and urban nesting is regularly recorded in the USA and Canada (Becker and Ball, 1983, Haney and White, 1999).

US forestry guidelines suggest a minimum no-cut buffer of 91 m around nest sites. A 400 m buffer is recommended for Richardson's merlin *F. C. richardsonii* (Becker, 1984), and it is recommended that human activities should not occur within 366 m of active nests (Becker and Ball, 1983). In Britain, a 200 – 400 m buffer around nest sites is recommended for forestry workers, although it is not clear on what this is based (Currie and Elliot, 1997). A survey of expert opinion by Ruddock and Whitfield (2007) found a wide range of suggested disturbance distances for nesting merlins, ranging from < 10 m to 300 – 500 m. This upper

survey limit corresponds roughly to that proposed by Currie and Elliot (1997) and proposed buffers in the USA.

Due to their relatively widespread population, merlin were not included on the sensitivity map, but instead written guidance has been provided. This approach should be assessed in light of the results of the 2008 national survey. Relocation of individual turbines within 200 – 500 m of a merlin nest is recommended, depending on individual site characteristics, with the upper end of the range applicable where turbines are in the line of sight.

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4. Red kite

Introduction

The breeding range of the red kite *Milvus milvus* is almost entirely restricted to Europe, with approximately 90 % of all red kites breeding in France, Germany and Spain (Gensbol, 1986, Evans and Pienowski, 1991, cited in Evans *et al.*, 1994). Other small populations are found in North Africa, and possibly also the Cape Verde Islands and part of the Middle East (Carter, 2001). A steady decline occurred across most of the range from the 19th century due to persecution (Carter, 2001) and the global population is now very fragmented and restricted to relatively small areas of the former range (Evans *et al.*, 1994). The red kite was once an abundant resident breeder throughout most of Britain, but had become extinct except for a small population in mid Wales by the late 19th century due to persecution (Carter *et al.*, 1998). The Welsh population reached a low of just three pairs in 1905, but has slowly increased, with the population being estimated at over 750 pairs in 2008 (RSPB, unpubl.).

In 1989, a red kite re-introduction programme began, initially with two release sites; one in the Chiltern Hills in England, and one on the Black Isle in Scotland (O'Toole *et al.*, 2000). Between 1989 and 1994, 93 young were released at each site, with successful breeding first occurring in 1992 (O'Toole *et al.*, 2000). A second English release site was established in central England in 1995, with 70 birds being released between 1995 and 1998, and a release site established in central Scotland in 1996, with 103 red kites being released between 1996 and 2000 (Wotton *et al.*, 2002, D. Cameron, pers. comm.). A third English release site in Yorkshire has been established, with 42 birds released in 1999 and 2000, and two more release projects began in Scotland, one in the South of Scotland in 2001 (Carter, 2001), and one in Aberdeen in 2007 (P. Newbery, pers. comm.). The first red kite reintroductions to Northern Ireland began in 2008, with 27 kites being released (A. Lonergan, pers. comm.).

There has been a rapid increase in most of the introduced populations, but range expansion was initially slow, with only a few pairs being found over 50 km from the nearest core area (Carter *et al.*, 1998). The first co-ordinated national survey in Britain was undertaken in 2000 and yielded 430 breeding pairs (Wotton *et al.*, 2002). Since then, the UK population has increased to an estimated minimum of 1350 breeding pairs; 500 or more in England, 750 or more in Wales, and 122 in Scotland (RSPB, unpubl.). Populations in south and central England now number over 350 and 100 breeding pairs, respectively, and a further 67 breeding pairs occur in Yorkshire, and 23 in north east England (2008 figures, RSPB, unpubl.).

At the beginning of the re-introduction programme, the red kite was one of only three globally threatened species occurring in the UK (Collar and Andrew, 1988). Following increases in parts of central and northern Europe, however, this classification has been reduced to near threatened (BirdLife International, 2005).

The red kite is an Annex I species (EC, 1979) and is on the amber list of conservation concern in the UK (Gregory *et al.*, 2002). The red kite is a qualifying species for just one Special Protection Area, which is in Wales and held 15 pairs of red kites, or 9 % of the British breeding population at the time of the SPA review (Stroud *et al.*, 2001).

Breeding system

Nests are usually located in woodland patches surrounded by open countryside, often in areas of well-spaced trees close to a woodland edge (Carter, 2001). The nest itself is built in the fork of a tree, nest heights in Wales of 4 - 30 m have been recorded, although most were between 12 and 15 m (Cross and Davis, 1998), whereas in the Chilterns most nests were at heights of over 15 m (Carter, 2001).

Red kites are usually monogamous (Cramp and Simmons, 1980), although polygamy has been recorded (Van Kleef and Bustamante, 1999). Eggs are laid from late March and clutch sizes in Wales range from one to three, with a mean of 2.13 (Cramp and Simmons, 1980), although in England clutch sizes of four are occasionally observed (T. Melling, pers. comm.). The red kite has a single brood but replacement laying occurs occasionally (Carter *et al.*, 1998). Incubation lasts between 31 and 38 days, and is by the female (Cramp and Simmons, 1980). Both parents are involved in care of young; for the first two weeks all food is brought to the nest by the male whilst the female broods the young, later the female helps with provisioning. Young usually fledge from 48 - 50 days, although this can be up to 60 - 70 days, and then spend a further 15 - 20 days near the nest being fed by the parents (Cramp and Simmons, 1980).

Home ranges and site fidelity

Site fidelity

As with many large raptors, the red kite shows a high degree of natal philopatry and despite the tendency for a proportion of young birds to disperse, the majority return to breed in the area where they are fledged or released (Newton *et al.*, 1994, Evans *et al.*, 1999). Seventy nestlings were tagged in Wales between 1975 and 1979, and it was found that individuals moved up to 22 km between birthplace and breeding place, but having bred, most stayed in the same locality from year to year (Newton *et al.*, 1989). Cross and Davis (1998) found that 139 birds marked as chicks were found breeding an average distance of 12.5 km from their natal site.

Carter (2001) states that breeding pairs tend to remain faithful to a particular territory year after year, although they do sometimes have an alternative breeding site, usually within 10 km and often much closer. Most birds in Wales have between one and five alternative nest sites within a territory (Cramp and Simmons, 1980). Of the seventy nestlings tagged in Wales between 1975 and 1979, there were 49 incidences of birds staying on the same nesting territory in subsequent years, and only four of birds changing territories, in which case the new territories were 5, 3, 2 and 26 km from the old territories (Newton *et al.*, 1989). One territory in Wales was used continuously for 17 years, and some Welsh sites, known to be occupied over 100 years ago, are still used today (Walters Davis *et al.*, 1973). Fidelity to the nest site itself has been related to breeding success in Wales; Walters Davis *et al.* (1973) found 80 % of nests that had fledged at least one chick were used the following year (n = 65), compared to just 40 % of nests that had failed (n = 68, territories including only one nest site were excluded from the analysis).

Nest sites of birds in England and Scotland vary in how fixed they have been; Evans *et al.* (1999) reported pairs shifting to new nests within a territory or to new territories between years, with distance moved by pairs between nesting attempts ranging from 0 to 55 km, but usually being less than 6 km. In England, between 1992 and 1995, 56 % of successful pairs returned to the same nest to breed the following year (Evans *et al.*, 1999). In Scotland in the same period, Evans *et al.* (1999) reported that only one successful breeding pair out of 15 returned to the same nest it had used previously. However, it is now considered that red kites in the north of Scotland are extremely site faithful, with almost all successful breeding pairs re-using the same nest the following year (B. Etheridge, pers. comm.).

Densities

In the south of England, breeding kites have been found nesting within 100 m of each other (Evans *et al.*, 1999). In 2000, the south England breeding area supported approximately 0.25

pairs per km², although densities could be locally higher (seven pairs breeding within an area of about 4 km², Carter, 1997), and this may have increased as the population increases (Carter, 2001). In Wales, typical densities are far lower, with Cramp and Simmons (1980) stating that most nests are about 3 - 5 km apart, although minimum nearest-neighbour distances of 200 m have been recorded (Carter, 2001). Breeding distribution of the north Scottish population has expanded from six to ten 10 km squares, with the mean breeding density increasing from 1.3 pairs per 10 km square in 1994 to 3.3 in 2000 (Wotton *et al.*, 2002).

Elsewhere in Europe, densities of 0.3 - 0.5 pairs per km² (north east Germany: Nicolai, 1997) and 0.03 pairs per km² (Spain: Viñuela *et al.*, 1999) have been recorded. However, some very high local densities have been recorded (e.g. 136 pairs in the 13 km² Hakel Forest in north east Germany, Nicolai, 1997) as a result of groupings of nests in isolated blocks of suitable breeding habitat.

Home range

Red kites do not defend exclusive foraging ranges, with the defended area usually being restricted to a few hundred metres around the nest site (Carter, 2001). Size of the home/foraging range can vary markedly with local food availability (Carter, 2001). In high quality habitat in England, sightings of wing-tagged birds show that most food is collected within 3 km of the nest, and that birds are rarely seen further than 5 km from the nest. Females were very rarely observed further than 1 km from the nest site (Carter, 2001). In Wales, Walters Davis *et al.* (1973) reported that individuals normally ranged 2 to 3 km from the nest site, although in some cases individuals were recorded 15 km from the nest. A mean area per breeding territory of 15 km² has been estimated from RSPB data (equivalent to a circle with radius 2 km; Ward, 1996). In Germany, Ortlieb (1989) found that foraging flights up to 10 km were typical, and in southern Spain, foraging trips were found up to 20 km from the nest, although this was probably an extreme (Veiga and Hiraldo, 1990). Direct observations of red kites in central Germany found maximum foraging flights of 4.5 km, with most flights being within 2.5 km of the nest, and home range area was estimated at about 7.5 km² (Porstendörfer, 1997).

Roosting

Red kites in England tend to form communal winter roosts within the core breeding areas (Carter *et al.*, 1998). Some of these sites are used over many years, with others being more temporary (Cramp and Simmons, 1980). A traditional roost site in central England that has been used since the start of the release programme remains the most important site in the area, with over 50 birds regularly attending in winter (Carter, 2001). In the south of England, where the population exceeds 500 birds, there are a number of alternative roost sites, between which birds can move, but the original roost site still attracts over 100 birds (Carter, 2001).

Numbers of birds at winter roost sites build up from late August to September, and peak between November and February (Carter, 2001). Communal roosts of young, non-breeding birds also occur throughout the summer, but these are less traditional in their location than winter roosts (Carter, 2001). In England, adults from nest sites within a few kilometres of the roost often attend pre-roost gatherings, but it is not known whether they subsequently roost communally or at the breeding site (Carter, 2001). Some pairs remain on their breeding territories throughout the winter, only attending roosts infrequently (Carter, 2001). In Wales and northern Scotland, most breeding adults tend to join the roosts as food is less easy to come by on breeding territories in winter, although they may sporadically return to their breeding sites (Carter, 2001).

Radio-tracking of first-year kites in England over several winters found that the majority of foraging took place within 4 km of the roost site, with winter home ranges of six individuals ranging from 19 to 32 km² (mean = 23 km², Carter and Grice, 2000). Winter home ranges of six adults roosting on their nest sites in Spain indicated that resident birds used relatively small home ranges, ranging from 6.5 to 36 km² (median = 29 km², Heredia *et al.*, 1991). Core home ranges, i.e. areas within the home range used more frequently than by chance, were even smaller, ranging from 2.7 km² to 11.5 km², with a median of 3.6 km². Resident birds were found an average of 3.9 km from their roost during the day, with migrant birds in the same area having larger home ranges and being found further from the roost (Heredia *et al.*, 1991).

Disturbance

Hötker *et al.* (2006) reviewed seven wind farm impact studies and found a negative effect of wind farms on abundance on red kites in the non-breeding season (i.e. a decrease in density following construction of a wind farm, or in comparison with a control site) in four cases, but no negative effect in three cases, and so no significant evidence of an effect overall. Not enough studies were available to investigate effects on abundance during the breeding season.

Ruddock and Whitfield (2007)'s review of disturbance distances has been drawn on to a large extent in the following review. Red kites appear quite tolerant of disturbance, with Carter (2001) suggesting that red kites are probably more closely associated with human activities in the countryside than any other species of raptor in Britain, and sometimes nest close to busy roads, farmsteads and public footpaths. A positive association with waste sites such as rubbish dumps, as well as dedicated feeding stations close to habitation, has been noted (Carter, 2001, L. O'Toole, pers. comm., B. Etheridge, pers. comm. in Ruddock and Whitfield, 2007). Studies have found no response elicited by shooting disturbance approximately 400 m from red kites in Scotland (L. O'Toole, pers. comm. in Ruddock and Whitfield, 2007), or by observing birds at a distance of 200 – 300 m (Mougeot, 2000). Some red kites in Scotland have chosen to nest on farms and in large gardens close to people, with a pair in 2005 nesting only 40 m from an occupied farmhouse and successfully rearing two young (B. Etheridge pers. comm.). Carter (2001) reported an incidence of a nest 150 m from a dwelling, which failed at the egg stage, but other successful nests above footpaths and within gardens have been observed (Carter, 2001, L. O' Toole, pers. comm.).

Despite this apparent relative tolerance, disturbance and habitat alteration have been cited as a potential cause of failed breeding in red kites (Davis and Newton, 1981, Carter, 2001, Carter *et al.*, 1998, Seoane *et al.*, 2003) and negative relationships between breeding density and productivity with anthropogenic structures noted in black kite *Milvus migrans* (Sergio *et al.*, 2003). However, it is possible this was due to changes/differences in habitat associated with the anthropogenic structures, rather than the disturbance itself (Ruddock and Whitfield, 2007). Black kites have been found to select urban areas when foraging, but cliff-nesting birds showed avoidance of paths (280 +/- 34 m), roads (466 +/- 38 m), buildings (523 +/- 44 m) and villages (1002 +/- 123 m) when selecting nest sites (Sergio *et al.*, 2003). The same was not found for tree-nesting black kites (Sergio *et al.*, 2003). Davis and Newton (1981) attributed a minimum of 9 % of all recorded nest failures to disturbance near the nest through agricultural or other human activity. This was also the proportion of nest failures due to disturbance in Wales, when 11 nests probably failed due to incidental disturbance (Carter *et al.*, 1998). A small number of nests in England and Scotland are also thought to have failed as a result of disturbance (Carter *et al.*, 1998). However, it is not clear whether these studies included deliberate disturbance (i.e. persecution). It is considered likely that red kite habituate to sources of disturbance (Ruddock and Whitfield, 2007).

In Britain, disturbance free zones of 300 – 600 m (Currie and Elliot, 1997) and 400 - 600 m (Petty, 1998) around red kite nests have been suggested for forestry workers. Ruddock and Whitfield's (2007) survey of expert opinion suggested tolerance ranges of 10 – 300 m for red kite. Ruddock and Whitfield (2007) suggest that such buffers should be applied pragmatically, as some pairs appear to be exposed to regular human activity without any effect, with other less habituated pairs perhaps being less tolerant. The review did not consider disturbance distances for communal roost sites (Ruddock and Whitfield, 2007).

Collision Risk

Flight behaviour

Red kite courtship displays usually involve slow, high circling above the nesting wood by one or both of the breeding pair, and considerable heights may be reached during these displays (Cramp and Simmons, 1980, Carter, 2001).

Foraging flights are by soaring and circling over open ground, and frequently also by gliding low (Cramp and Simmons, 1980). Thus, high circling is also used when foraging, and birds can circle up to several hundred metres (Carter, 2001). Blanco (1982) found that kites in Spain spent about 65 % of their flying time each day over 20 m. Prey may be caught in the ground or the air (Carter, 2001), smaller carcasses are located by gliding just a few metres over the ground, with occasional brief hovering flights to locate prey. Pursuits of live prey can also involve gliding and flapping flights, and food piracy is common (Carter, 2001).

Collision

Raptors, and particularly red kite, were amongst the most frequent collision fatalities in a review of (predominantly European) wind farm impact studies by Hötker *et al.* (2006). Forty red kite fatalities were recorded at wind farms in Germany (where around half of the world's red kite population breeds) since 1989, with a further two occurring in the UK and one in Sweden, most of these collisions occurred during the breeding season (Hötker *et al.*, 2006). These figures have since been updated, with a total of 91 collisions recorded in the German wind farm collisions database from 2001 to 2007 (T. Dürr, unpubl.).

Impacts of wind farms on red kites have been studied by radio-tracking birds at a 28-turbine wind farm at Braes of Doune in Scotland (Natural Research Limited, 2008). Overall, 69 birds were fitted with radio-tags, and 64 of these have been monitored (Natural Research Limited, 2008). Despite a high level of use of the wind farm site, none of these tagged birds have been collision victims to date, although one non-radio-tagged adult collided in July 2007, nine months after the first turbine became operational, and four months after completion of the wind farm (Natural Research Limited, 2008). No kite casualties having been found at a wind farm site in Wales during a post-construction study (Percival, 2000), although casually discovered victims have been found at Welsh wind farms (Whitfield and Madders, 2006).

In light of this uncertainty as to whether the red kite is particularly susceptible to collisions, Whitfield and Madders (2006) conducted some analysis to derive an estimated turbine avoidance rate by red kite. In the absence of suitable data to calculate the avoidance rate, this was based on comparisons with other raptors to assess relative susceptibility to collision, and whether red kite avoidance rates were likely to be similar to those for other raptors, which are often cited as lying between 98 % and 100 % (Whitfield and Madders, 2006, Whitfield and Band, in prep.). Data on flight activity and collision fatalities for several raptors from a dataset gathered at several wind farms in Navarra, northern Spain were used (Lekuona and Ursúa, 2006). Regressions of number of birds seen on number of dead birds found, and

number of birds seen flying close to rotor blades on fatalities were used to assess relative vulnerability to collision, and red kite were found not to be unusually susceptible to collisions amongst raptors (unlike griffon vultures and common kestrels). The authors concluded that likely avoidance would probably be within the 98 % to 100 % range, and that an estimate of 99 % was appropriate until empirically derived estimates are available. Hopefully further post-construction monitoring at wind farms coinciding with red kites, such as the work in Germany and Braes of Doune, will help to clarify the circumstances leading to collisions in red kites.

Large birds of prey also may be susceptible to risk of collision with power lines. There is currently very little information on the problem in the UK and it is not known whether power lines pose a serious threat to raptors at the population level. Carter *et al.* (1998) suggest that collision with power lines is likely to be at most a local problem. Janss (2000) studied bird mortality with power lines in south west Spain, by surveying for corpses underneath power lines. Forty five red kite corpses were found, but all were classified as having died of electrocution rather than collision. Carter (2001) suggests that red kite can be killed by the standard three wire electricity pylons that are widespread in Britain, and that at least four deaths in Scotland and three in England have been due to this. A further ten red kites have been found under power lines in Britain of unknown causes, and Carter (2001) suggests that often it is assumed that cause of death is collision when it may in fact be electrocution (see also Haas *et al.*, 2005). Evans *et al.* (1994) found that 13 % of all recorded deaths were probably due to collision with electricity wires and high-speed vehicles (persecution accounting for 47 % and just 13 % of recorded mortality being from natural causes). Davis and Newton (1981) give similar figures for the period 1951 - 1980; of 32 red kites reported dead, 31 % died due to human persecution, 9 % due to collision with electric wires or vehicles, and 19 % from natural causes. However, as Carter (2001) points out, it is not specified whether these deaths may have been due to electrocution.

Sensitivity criteria

This review on red kite has been included, as they were identified as one of the most frequent collision victims in Hötter *et al.*'s (2006) review. However, only a few collisions have been recorded outside Germany, despite construction of wind farms in areas with high numbers of red kite. Information on disturbance in red kite is variable, but appears to indicate they can be relatively tolerant of disturbance.

Red kite were not included on the English sensitivity map, because their rapidly increasing population size and range expansion would mean that a mapped distribution would be quickly out of date. This differs from the approach taken for the Scottish sensitivity map (Bright *et al.*, 2006, 2008), which was more precautionary, as the Scottish range is still much more restricted, and the rate of population growth slower. Relocation of individual turbines within 300 m of a red kite nest should be considered, depending on individual site characteristics. Wind farms sited in areas with particularly large concentrations of red kites, for example communal roost sites, or in areas where new populations are establishing, may cause concern. A map to show release locations and core areas used around these, within which most of the important roost sites will be located, has been included, by buffering release locations by 10 km, to provide guidance for likely search areas within which impacts of wind farms on red kites may occur (Figure A2.1). Early consultation with local NE and RSPB staff is recommended to identify situations in which red kite is likely to be of particular concern.

Acknowledgments

These sensitivity criteria are based on a previous version written for a Scottish sensitivity map (Bright *et al.*, 2006, 2008), updated in light of new information and to place them in the context of English population sizes and trends. They were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thanks to Ian Carter, Martin Kerby, Jeff Knott, Tim Melling, Peter Newbery and Colin Wilkinson who provided helpful comments on an earlier draft of these sensitivity criteria.

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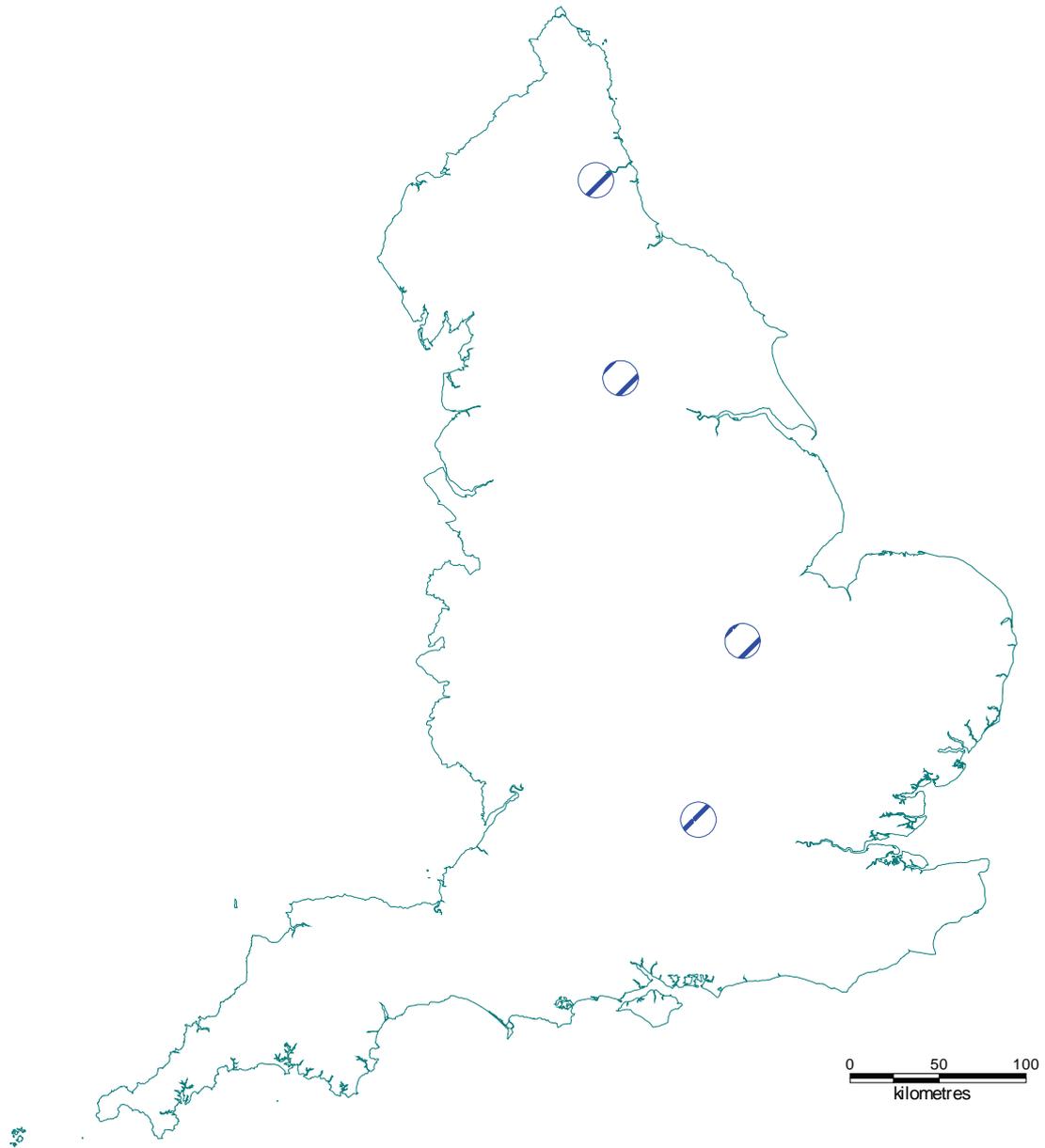


Figure A2.1. Red kite release locations in England, buffered by 10 km

5. Golden plover

Introduction

The global distribution of golden plover is restricted to boreal regions of the western Palaearctic, with only a small extension further east (Stroud *et al.*, 2001). In Europe, breeding occurs throughout Iceland, Scandinavia and the Baltic States, northern Russia and in northern/upland parts of Britain and Ireland, which form the southern edge of the species' range (Stroud *et al.*, 2001). The European population is estimated at between 440 000 and 785 000 breeding pairs (Hagemeyer and Blair, 1997). The core population in northern Europe was relatively stable between 1970 and 1990, but the smaller southern populations have been in decline since the 19th century (Tucker and Heath, 1994). It was previously thought that there was an *apricaria* subspecies, restricted to Britain (22 600 pairs), Ireland (400 pairs), Denmark (6 - 9 pairs) and Germany (19 pairs) and in decline (Stroud *et al.*, 2004). However, there is debate as to whether this is a genuine subspecies or simply a cline of variation across the range (Byrkjedal and Thompson, 1998).

In Britain, the species is distributed widely throughout upland areas, with concentrations in northern and western Scotland and the north and south Pennines, and smaller outlying groups breeding in Wales and south west England (Ratcliffe, 1976, Gibbons *et al.*, 1993). Two-thirds of the British and Irish breeding population occur in Scotland (Stroud *et al.*, 2001). A review of population trends of breeding birds in the UK since 1800 reported that the breeding population of golden plover was in decline between 1800 and 1970, and probably also between 1970 and 1995 (Gibbons *et al.*, 1996). Numbers in Britain were estimated at 29 400 pairs during 1968 - 1972 (Sharrock, 1976), and at 22 600 pairs during the 1980s (Baker *et al.*, 2006, based on Reed, 1985 and Stroud *et al.*, 1987). The most recent population estimate is for the UK, and is 38 400 – 59 400 pairs (O'Brien in Thorup, 2006). This is derived from modelled extrapolations, and suggests that the previous population estimates were under-estimates, rather than representing a genuine population increase. Even the estimate by O'Brien (in Thorup, 2006) may be an underestimate, being largely based on an extrapolation from two-visit Brown and Shepherd surveys (see Pearce-Higgins and Yalden, 2005). Substantial range contractions in Britain and Ireland have been attributed to afforestation and agricultural intensification (Thom, 1986, Boobyer, 1992, Lovegrove *et al.*, 1994, Byrkjedal and Thompson, 1998, Fuller and Gough, 1999), reduction of moorland burning and game-keeping (Gibbons *et al.*, 1993, Tharme *et al.*, 2001). These factors have led to extinctions in some areas (Parr, 1992).

The RSPB's Repeat Upland Bird Surveys, which involved the resurveying in 2000 and 2002 of nine study areas in the British uplands (totalling 1348 km²) first surveyed between 1980 and 1991, found mixed results for golden plover (Sim *et al.*, 2005). Numbers in one study area had undergone a significant decline, whilst in two others significant increases had occurred (Sim *et al.*, 2005). There is, however, evidence of long-term national scale range decrease for golden plover, of around 7 %, between 1968 - 72 and 1988 - 91 (Sharrock, 1976, Gibbons *et al.*, 1993).

The golden plover is an Annex I species (EC, 1979) and is on the green list of conservation concern (Gregory *et al.*, 2002). Around 26 % of the British breeding population of golden plover occur within seven SPAs, three of which are in England, and an estimated 22 % of the British wintering population occurs within 22 SPAs, 16 of which are in England (Stroud *et al.*, 2001).

Breeding system

Golden plover breed on heather moorland, blanket bog, acidic grasslands and montane summits, favouring areas with short vegetation (Pearce-Higgins and Grant, 2006). Adjacent

pastures are important for feeding adults, with birds commuting up to 10 km from the nest to feed on earthworms and leatherjackets (Pearce-Higgins and Yalden, 2003). They select fields with short swards and high invertebrate availability, and may feed in different areas during the night to those used during the day (Whittingham *et al.*, 2000, Pearce-Higgins and Yalden, 2003). Chicks appear to forage on a range of invertebrates, but leatherjackets and crane flies (larval and adult Tipulidae) appear most important (Whittingham *et al.*, 2000, Pearce-Higgins and Yalden, 2004), and they select the most invertebrate rich habitats. They can range up to 2 km or more with a mean home range in one study of 41 ha (Byrkjedal and Thompson, 1998, Pearce-Higgins and Yalden, 2004). Adults are site-faithful and recruits are faithful to their natal area (Parr, 1980).

Breeding densities generally vary from 2 - 7 pairs per km², with exceptional densities of 16 pairs per km² having been recorded (Ratcliffe, 1976). Densities in Great Britain are some of the highest within the species' range (Byrkjedal and Thompson, 1998).

In northern England, first eggs are laid at the end of March, but mainly from April 10th to May 7th (Cramp and Simmons, 1983). A clutch of four, sometimes three, and rarely two or five is laid (mean = 3.85, n = 126, northern England, Cramp and Simmons, 1983). There is one brood, although replacement clutches may be laid following the loss of the first clutch (Cramp and Simmons, 1983). Incubation is conducted by both sexes, and lasts 28 - 31 days (Cramp and Simmons, 1983, Pearce-Higgins and Yalden, 2003). Young are precocial and nidifugous. They are cared for by one parent, the female often abandons once chicks reach a certain age, or parents may take it in turn to care for chicks, whilst the other is away feeding on pasture (J. Pearce-Higgins, pers. comm.). Fledging occurs after 37 days (Pearce-Higgins and Yalden, 2002).

Wintering golden plover

During the winter, the small British breeding populations of golden plover and lapwing are joined by large numbers of immigrants from Iceland and continental Europe (Byrkjedal and Thompson, 1998, Wernham *et al.*, 2002). Unlike most wader species, golden plover and lapwing are mostly found on farmland during winter months (Cramp and Simmons, 1983, Stroud *et al.*, 2004). Historically, the species showed preference for grassland habitats, but there has been a recent change in distribution from the mixed farmland areas of central England (Lack, 1986) to the arable east (Gillings *et al.*, 2006). This is likely to be due to changes in climate, with milder weather allowing birds to winter closer to their breeding sites, although changes in terrestrial habitat, and increased intertidal feeding are alternative explanations (Gillings *et al.*, 2006).

The size of the wintering population is difficult to ascertain, due to the aggregated nature of the species in winter. Surveys of wintering golden plover on farmland have not been conducted until recent years. The Wetland Bird Survey (WeBS) is the main source of long-term data on wintering golden plover, providing counts at virtually all estuaries since the 1970s (Gillings *et al.*, 2006). Counts often include marshland, saltmarsh and fields adjacent to the coast in which waders often roost, but will miss the majority of agricultural sites used (Gillings *et al.*, 2006). For example, it was estimated that fewer than one quarter of Britain's golden plover and lapwing were thought to winter on coastal sites in the 1990s (Cayford and Waters, 1996). Inland wetlands are also surveyed, but counts of waders at these have only been conducted since 1991 (Gillings *et al.*, 2006). Recently, Winter Farmland Bird Survey (WFBS) data, which surveyed a stratified sample of random squares on lowland farmland between 1999 and 2003, has provided some further data (Gillings *et al.*, 2006).

The Atlas of Wintering Birds in Britain and Ireland ('Winter Atlas') provided an estimate of 250 000 wintering golden plover (Lack, 1986). More recently, a Wintering Plover Survey, comprising casual records of flocks of over 100 plovers (single species or mixed flocks), has attempted to overcome the problem of lack of data on plover wintering on farmland, and has been used to investigate changes in distribution. A separate survey of stratified random squares, the Dispersed Waterbirds Survey, provides the most recent population estimate of 580 000 wintering golden plover (Jackson *et al.*, 2006). These datasets have been used to investigate long-term trends, which show a decrease in numbers in the 1970s, followed by a short period of stability in the early 1980s, followed by a recent increase (Gillings *et al.*, 2006). The recent increase is due entirely to increases in the east of England, with numbers in all other regions declining (Gillings *et al.*, 2006).

Golden plover forage both during the day and night, and recent work comparing nocturnal and diurnal foraging reveals differences in behaviour and sites utilised (Gillings *et al.*, 2005). Nocturnal feeding is conducted in smaller, more widely dispersed flocks, and diurnal counts do not predict nocturnal feeding numbers (Gillings *et al.*, 2005). This can lead to problems, given that diurnal surveys are used to identify areas for site protection (e.g. Stroud *et al.*, 2001). For example, Gillings *et al.* (2005) found that in one year, the study area was used for nocturnal feeding, but the closest diurnal feeding areas were 6 km away. Diurnal observations on consecutive days confirmed that movements of ≤ 15 km occurred regularly (Gillings *et al.*, 2005). It seems that in golden plover, nocturnal feeding is conducted in preference to, rather than to supplement, diurnal feeding, given that nocturnal intake rates are up to 50 % higher (due to a greater reliance on catching large earthworms at night, Gillings and Sutherland, 2007).

This, together with movement of wintering flocks between years due to changes in cropping, can make a site-based approach to conservation difficult for wintering golden plover (Gillings *et al.*, 2007), and could lead to problems identifying important areas for wintering golden plover or areas of potential sensitivity in relation to wind farms.

Disturbance

Recreational disturbance

Reported distances at which golden plover react to human disturbance range from 50 m to 400 m (Ratcliffe, 1976, Thompson and Thompson, 1985, Yalden and Yalden, 1989, Yalden and Yalden, 1990, Byrkjedal and Thompson, 1998, Finney *et al.*, 2005). Disturbance of golden plover has been the subject of a long-term study at Snake Summit in the Peak District National Park. This is an area with an intensively used footpath, the Pennine Way, running through it. Yalden and Yalden (1989) suggested that the high level of recreational disturbance has actually led to population declines in areas of the national park. Finney *et al.* (2005) studied avoidance of the Pennine Way before and after resurfacing of the footpath in 1994. Prior to the footpath being resurfaced, 30 % of people strayed from it, whereas after resurfacing the figure was just 4 %. This reduction in people straying from the path coincided with a reduction in displacement of nesting golden plover from 200 m to 50 m (Finney *et al.*, 2005). Comparisons were also made with another area of the national park, which was subjected to lower visitor pressure. In this area, there was no evidence that golden plover avoided nesting close to the path, indicating that there is a threshold level of disturbance tolerated by golden plover (Finney *et al.*, 2004).

Yalden and Yalden (1989) studied the response of golden plover at Snake Summit to an observer approaching and found that adults with chicks alarm-called when approached to within 200m. Disturbance was also found to reduce time spent incubating (Yalden and

Yalden, 1989). During the chick-rearing period, adults spent about 11 % of their day responding to disturbance, which was estimated to increase energy expenditure by 15 %. Less time was also spent brooding chicks (Yalden and Yalden, 1989). Responses to disturbance, such as flushing and reduced time spent brooding, can lead to eggs and chicks being exposed to possible chilling or predation, and impose an energetic cost on adults (Nudds and Bryant, 2000, Bolduc and Guillemette, 2003). However, there was no evidence that chick growth rates and breeding success, in terms of number of chicks, were reduced close to the Pennine Way footpath (Finney *et al.*, 2004).

Disturbance from wind farms

A recent review by Whitfield (2007) on effects of wind farms on waders, with particular reference to golden plover, suggests that the main potential impact is from disturbance displacement, and that this is particularly likely to affect wintering birds. Displacement by wind farms is generally more of an issue for non-breeding birds (Hötker *et al.*, 2006), displacement is likely to be related to factors such as investment made in the site and availability of alternative suitable habitat. Whitfield (2007) suggests that evidence for displacement of waders, including golden plover, during the breeding season is limited. However, a recent field study comparing bird distributions at 11 wind farm sites and paired control sites found evidence for displacement up to distances of at least 200 m from wind turbines (Pearce-Higgins *et al.*, 2008). Densities at wind farm sites were also lower than predicted, after controlling for habitat associations (Pearce-Higgins *et al.*, 2008). Disturbance displacement was also observed for other breeding waders, with displacement occurring at distances of 0 – 800 m, depending on the species (Pearce-Higgins, unpubl.). The cumulative effect of displacement for breeding waders may be considerable.

Hötker *et al.* (2006) reviewed 127 wind farm impact studies for evidence of displacement and collision risk, with an emphasis on bird species that occurred in Germany. The review included 29 studies of non-breeding golden plover, and found evidence for disturbance displacement in 72 % of these. Minimum displacement distances for non-breeding golden plover ranged from less than 50 m to 850 m, with the median being 135 m (n = 22). This distance was similar for a suite of non-breeding wader species (curlew: median = 190 m, n = 24; oystercatcher: median = 15 m, n = 6; lapwing: median = 135 m, n = 32; common snipe: median = 300 m, n = 5). Distances for breeding waders were also of a similar order of magnitude (black-tailed godwit: median = 300 m, n = 5; redshank: median = 188 m, n = 8; oystercatcher: median = 25 m, n = 13; lapwing: median = 100 m, n = 6), although no figure was provided for golden plover due to there not being sufficient studies available. In non-breeding and breeding golden plover, which often roost or feed in agricultural habitats, short vegetation height and large fields are apparently preferred, so any changes in these features associated with wind farm development risk increasing displacement (Whitfield, 2007). Hötker *et al.* (2006) also found that, of the species and species groups investigated, only waders and gamebirds displayed reduced numbers during the breeding season in response to wind farms. This concurs with Stewart *et al.*'s (2007) meta-analysis of wind farm impact studies, which found reduced abundances at wind farm sites were most common in waterfowl.

Hötker *et al.* (2006) also investigated whether turbine height affected displacement distance, and found a positive relationship between turbine height and disturbance displacement distance for wintering golden plover. It should be noted that the relationship was not significant, and that a significant relationship was only found for one of the 33 species investigated (non-breeding lapwing). However, Whitfield (2007) suggested that, given the possibility of a relationship between turbine height and displacement- 'For modern turbines

with a hub height of, for example 80 m, it may be that the maximum observed displacement distance (850 m) would have to be assumed in EIAs, at least until more information is forthcoming from studies involving modern turbine dimensions; whilst 850 m is exceptional in existing studies, smaller displacement distances are likely to have been recorded for smaller turbines'.

In contrast to European studies, in America there is no evidence of displacement of waders by wind farms (Whitfield, 2007). Whitfield (2007) suggests that this could be due to the fact that studies in America are predominantly of spring migrant or breeding waders, where displacement is less common than in wintering waders. However, recent studies by Pearce-Higgins *et al.* (2008, unpubl.) provide evidence for disturbance displacement in breeding waders.

Breeding golden plover thus appear to be affected by human disturbance over distances of a few hundred metres, with similar displacement distances being observed in response to the presence of wind turbines by breeding and non-breeding birds. Whether or not this scale of displacement is important biologically will depend upon its effects on survival and productivity, which will be related to factors such as availability of alternative locations and energetic costs of moving.

Although evidence of habituation was found in three of four studies of non-breeding golden plover, Hötker *et al.* (2006) concluded that, generally, evidence for habituation of birds to wind farms was neither widespread nor of a strong effect. A meta-analysis of the effects of wind farms on birds by Stewart *et al.* (2007) found that the effects of wind farms on bird abundance actually became greater over time (i.e. the opposite to that expected if habituation was to occur).

Collision risk

During the breeding season, golden plover would be most at risk of collision when performing breeding display flights, which occur at turbine height (30 - 100 m, Cramp and Simmons, 1983; 15 - 100 m, Byrkjedal and Thompson, 1998), and during foraging flights. Adults sometimes forage away from the nesting territory and can make foraging flights at night (Parr, 1980, Whittingham *et al.*, 2000, Pearce-Higgins and Yalden, 2003). Foraging usually occurs away from the moorland nesting habitat during incubation, but more commonly, although not exclusively, on the breeding territory once broods are present (Whittingham *et al.*, 2000, Pearce-Higgins and Yalden, 2003).

However, there are few documented cases of golden plover colliding fatally with wind turbines. A review of the European literature found two recorded casualties of golden plover as a result of collision with wind turbines in Germany since 1989, with a further casualty in both Sweden and the Netherlands (Hötker *et al.*, 2006). It should be noted that this review is the result of collation of records from a variety of sources, as opposed to controlled corpse searches at wind farms, and different taxa may feature more or less prominently due to factors such as size. There are also records of collisions during the breeding season in Norway (Norwegian Institute for Nature Research (NINA), unpubl.).

The review by Whitfield (2007) considered that collision risk for waders was generally low. Whitfield (2007) used wind farm impact studies to estimate turbine avoidance rates for waders. It was not possible to use data from European studies, as displacement from the wind farm area meant that avoidance of wind turbines within the wind farm area could not be measured. Instead, data from two wind farms in America were used, where, as discussed,

no evidence of displacement of waders by wind farms has been found (Whitfield, 2007). As discussed by Whitfield (2007) there are a number of assumptions underlying collision risk modelling which mean that predicted fatality rates should be treated with caution, and used as comparative, rather than absolute, measures. Corpse searches were conducted at the two sites at two or four week intervals, and search efficiency and scavenger removal biases corrected for (see Whitfield, 2007, for details). Estimates of avoidance rates for waders were high, ranging from 99.1 % to 99.9 % (Whitfield, 2007). Only a single wader carcass was found at either site, despite four years of observations at one, and three years at the other (Whitfield, 2007).

As discussed earlier, golden plover feed nocturnally as well as diurnally, and different foraging sites are often used at night, both during the breeding (Pearce-Higgins and Yalden 2003) and non-breeding seasons (Gillings *et al.*, 2005). Although it is not feasible to conduct large-scale (i.e. national) nocturnal surveys for wintering golden plover, nocturnal feeding sites should be identified during Environmental Impact Assessment, as these are arguably more important than day time feeding sites (Gillings *et al.*, 2005). Whitfield (2007) considers that whilst it is 'entirely appropriate to argue that nocturnal activity of non-breeding shorebirds should be considered as part of wind farm proposals, there is little apparent justification for arguing that if waders are present at night then collision risk would be markedly elevated'. This conclusion is based on the fact that day-time behaviour, and nocturnal behaviour of other bird taxa would suggest that golden plover are likely to be displaced from wind farms at night given that they are during the day, making them less prone to collision. The conclusion is also supported by Gillings and Sutherland's (2007) statement that 'Darkness does not hinder foraging plovers since they have specialised night vision (Rojas de Azuaje *et al.*, 1993, Rojas *et al.*, 1999)'. However, it is possible that behaviour of foraging birds commuting to and from nests at night, or between day time and nocturnal feeding sites, put them at higher risk of collision.

Sensitivity criteria

It appears that the main issue for breeding and wintering golden plover is likely to be disturbance, with evidence for displacement of breeding golden plover occurring up to distances of at least 200 m, lower than expected breeding densities at wind farm sites, and displacement of non-breeding birds occurring at distances up to 850 m (median = 135 m). Although it is not possible to map nocturnal distribution, night-time surveys should be undertaken during Environmental Impact Assessments for this species.

The three English Special Protection Areas (SPAs) for golden plover (the North Pennines, South Pennines and North York Moors SPAs) contain important breeding populations, and will automatically be included as 'high sensitivity'. SPAs for wintering golden plover will also be included as 'high sensitivity', together with two further sites identified as Important Bird Areas due to nationally important numbers of wintering golden plover, these are a suggested extension of the SPA at Bodmin Moor and new site at Nene Valley. Lack of comprehensive data meant that breeding and wintering golden plover were not included on the map separately. SSSIs selected as holding important populations or breeding waders or wintering waterfowl, and principal sites based on WeBS counts containing nationally or internationally important numbers of any waterfowl species, or a five-year mean of over 20 000 birds were included, which will have increased coverage for this species.

Acknowledgements

These sensitivity criteria are based on a previous version written for a Scottish sensitivity map (Bright *et al.*, 2006, 2008), updated in light of new information and to place them in the context of English population sizes and trends. There were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thanks to Simon Gillings and James Pearce-Higgins who provided helpful comments on an earlier draft of these sensitivity criteria.

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6. Dunlin

Introduction

The dunlin *Calidris alpina* has a wide global distribution around the Arctic, and is found in nearly all Arctic regions (Stroud *et al.*, 2001). In Europe, they also extend south to temperate regions where they are found in wetland habitats (Stroud *et al.*, 2001). The breeding population extends from east Greenland, across the Russian Arctic (Lappo and Tomkovich, 1998) to the Alaskan coast of the Bering Sea. Within Europe, dunlin breed mainly in the north, this area accounting for less than half of its global breeding range. Its European breeding population is large (over 300 000 pairs) and was probably stable between 1970 and 1990 (although the European wintering population declined markedly). The species was also stable in north west Europe between 1990 and 2000, but declined around the Baltic and Russia, thus the species is given an evaluation of 'Depleted' in *Birds in Europe* (BirdLife International, 2004).

Breeding dunlin are characteristic of moorland and upland habitats. Concentrations in the UK are found in the Flow Country of Caithness and Sutherland, and on peat moors in the Orkneys, Shetland, Grampians, Pennines and Outer Hebrides (Stroud *et al.*, 1987, 1988, Gibbons *et al.*, 1993). The machair of the Outer Hebrides is a favoured area, and up to a third of the British population was formerly estimated to breed there (Fuller *et al.*, 1986). Dunlin breeding in Britain and Ireland are of the temperate population of *C. a. schinzii*, which also occurs in the Baltic region. The UK breeding population of dunlin is estimated to be 9150 – 9900 pairs (Baker *et al.*, 2006, based on Reed, 1985), which represents 83 % of the biogeographical population. This is a dated estimate that is overdue revision, but is the best available at present.

The British population is currently considered one of three distinct populations of the *schinzii* subspecies (Stroud *et al.*, 2004). There is no recent information on population trends at the national level, although Gibbons *et al.*'s (1996) review of population trends for UK breeding birds suggested that the breeding population of dunlin was probably in decline between 1800 and 1940, but that there was little change between 1940 and 1995. However, the RSPB's Repeat Upland Bird Survey, which resurveyed nine study areas in 2000 and 2002, that were first surveyed between 1980 and 1991, found widespread population declines of dunlin (Sim *et al.*, 2005). These were large enough to suggest a population decline of at least 50 % over the last 25 years, leading to the recommendation that the dunlin's conservation status should be upgraded from 'medium' to 'high' conservation concern (Sim *et al.*, 2005).

Several regional studies have also demonstrated declines. In the Flow Country, numbers of dunlin were estimated to have fallen by 17 % due to the afforestation that has occurred there since 1945 (Stroud *et al.*, 1987, see also Avery and Haines-Young, 1990), and numbers have continued to decline following the cessation of active afforestation in the area. Whitfield (1997) found that numbers fell on five of 12 sites in Caithness and Sutherland surveyed in the period between 1979 and 1987, and in 1993 - 4, and the overall numbers fell by 2.4 % per year. Afforestation is also estimated to have resulted in the loss of 400 pairs of dunlin from Wales, the North York Moors and Southern Uplands (Stroud *et al.*, 1987). Agricultural intensification has caused additional population losses, for example in Orkney (Booth *et al.*, 1984, Ratcliffe, 1990). In the southern Outer Hebrides, the large populations that breed on the machair are in decline, largely due to egg predation by introduced hedgehogs *Erinaceus europaeus* (Jackson and Green, 2000). The percentage of first nests hatching fell from 59.6 % in 1985 - 1987, when hedgehogs were absent, to 8.8 % in 1996 - 1997, although declines have also been recorded in hedgehog-free areas of North Uist (Fuller and Jackson, 1999).

In contrast to these declines, on peatland sites in Lewis and Harris, where there has not been substantial habitat change, numbers of dunlin showed a slight increase of 1.4 % per year between surveys in 1987 and 1994/5 (Whitfield, 1997, see also Stroud *et al.*, 1988).

The subspecies of dunlin that breeds in Britain and Ireland, *schinzii*, is an Annex I species (EC, 1979) and is on the amber list of conservation concern (Gregory *et al.*, 2002), but see earlier text indicating that it may be red-listed at the next revision of Birds of Conservation Concern. Around 74 % of the British breeding population of dunlin occur within eight SPAs, two of which are in England (Stroud *et al.*, 2001). There are also 38 SPAs for non-breeding dunlin, within which approximately 78 % of the wintering population occurs (Stroud *et al.*, 2001).

Breeding system

Dunlin lay eggs from late April (Cramp and Simmons, 1983). A clutch of three or four (rarely 2 – 6) is laid (Cramp and Simmons, 1983). The clutch is incubated for about 22 days by both sexes; mainly the female at night and the male during the day (Cramp and Simmons, 1983). Young are precocial and nidifugous, they are cared for by both parents at first, and fledge after 19 - 21 days (Cramp and Simmons, 1983). There is usually one brood (very rarely two), although replacement clutches may be laid (Cramp and Simmons, 1983).

Site fidelity

Breeding densities are often clumped (Jönsson, 1988), with territory sizes of 5.7 - 6.9 ha being recorded in Alaska (Holmes, 1966). Dunlin are site faithful and philopatric (Soikkeli, 1970). Territorial behaviour decreases after pairing and ceases after hatching (Soikkeli, 1970). Broods may travel up to 1 km in two days and adults may feed up to 3 km from broods (Soikkeli, 1970).

Disturbance

Yalden and Yalden (1989) studied the effect of disturbance on golden plover and dunlin in the Peak District National Park. They suggested that golden plover had declined in some areas due to the high levels of recreational disturbance, with dunlin not having declined in the same areas. They found that breeding dunlin at Snake Summit, an area with a high level of visitor pressure, reacted to a human observer by alarm calling at a much closer distance than did golden plover, not reacting until approached to within about 35 m (n = 30, 95 % C. I. = 23-38), as opposed to 187 m (n = 333). A study in Scotland found similar results, with dunlin only taking flight when approached to within 20 m (Thompson and Thompson, 1985). Yalden and Yalden suggested that dunlin appeared more tolerant to human disturbance due to the fact that they rely on neighbouring Golden Plover as 'watch dogs' (see also Byrkjedal and Kalas, 1983, Thompson and Thompson, 1985).

Finney *et al.* (2004) studied dunlin on Snake Summit before and after resurfacing of the Pennine Way. They found that distribution of dunlin was affected by habitat and distance from the Pennine Way, with occurrence of dunlin being more likely closer to the footpath. However, following resurfacing work, which reduced the number of people straying from the footpath from 30 % to 4 %, the use of areas within 200 m of the path increased by about 50 %. Median distance between nests and the footpath declined from 175 m to 97 m (although this was not statistically significant). Thus, it was concluded that dunlin are affected by disturbance, and that the preference for areas close to the footpath was likely to be due to the habitat in this area (Finney *et al.*, 2004).

Collision risk

Males perform display flights during most of the early breeding season, and these are at turbine blade height (10 - 50 m, Holmes, 1966, Cramp and Simmons, 1983), which presents a

collision risk factor during the breeding season. However, a review of wind farm impact studies (predominantly of European wind farms) found no records of collision fatalities for dunlin (Hötker *et al.*, 2006). However, this review is the result of collation of records from a variety of sources, as opposed to controlled corpse searches at wind farms, and, being small birds, dunlin could be overlooked due to higher scavenger removal rates or lower search efficiency, hence a precautionary approach being taken.

Sensitivity criteria

Although there are not studies specific to dunlin, it appears that waders as a group can be susceptible to disturbance from wind farms (Hötker *et al.*, 2006, Stewart *et al.*, 2007, Whitfield, 2007), and dunlin also may be at risk of collision during breeding displays. Special Protection Areas for dunlin will be included on the map. Lack of comprehensive data meant that dunlin were not included on the map separately. IBAs and SSSIs selected as holding important populations or breeding waders or wintering waterfowl, and principal sites based on WeBS counts containing nationally or internationally important numbers of any waterfowl species, or a five-year mean of over 20 000 birds were included, increasing coverage for this species.

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Appendix 3: Sensitivity criteria

1. Bittern

Introduction

The range of the great bittern ('bittern') *Botaurus stellaris* extends from Britain eastwards to northern China and from Finland and Russia south to Turkey and Iran (Kushlan and Hafner, 2000). The European range is fragmented and localised due to loss of suitable wetland areas (Hagemeijer and Blair, 1997). Birds in the south and west of the European range tend to be sedentary, whilst those in the north and east tend to be migratory, moving south-westwards in search of ice-free feeding areas (Stroud *et al.*, 2001). In the UK there is a small breeding population, confined to England, and small numbers of bittern also arrive in the UK in autumn or winter, departing again in spring (Stroud *et al.*, 2001).

Although once a widespread and common species across the UK, in 1886 the bittern became extinct in the UK, due largely to persecution and drainage of wetlands for agriculture (Self, 2005). However, it then returned about 20 years later (Turner, 1924). Following this recolonisation, the population increased in size to an estimated peak of about 70 booming males in the 1950s (Day and Wilson, 1978). The population then declined again, showing a decline of 39 % between 1970 and 2005 (Eaton *et al.*, 2007), reaching its lowest point of 11 booming males in 1997 (Gilbert *et al.*, 2005a), and similar declines occurred elsewhere in Europe (Tucker and Heath, 1994). The reasons for these recent declines are unclear, although habitat degradation and pollution have been suggested (Tyler *et al.*, 1998). Efforts since then by the RSPB and other partner organisations to restore degraded breeding sites, or create new sites, and to increase abundance and accessibility of key prey species have resulted in a population increase (Self, 2005), and in 2008 the population was estimated at 76 booming males (Wotton *et al.*, 2008). The Suffolk Coast and Norfolk Broads remain the stronghold for the population, with birds also being found in the Fens and in southeast, northeast and northwest England (Wotton *et al.*, 2008).

The bittern is an Annex I species (EC, 1979) and is on the red list of conservation concern (Gregory *et al.*, 2002). There are five Special Protection Areas (SPAs) for breeding bittern, and ten for wintering bittern, in the UK (Stroud *et al.*, 2001). At the time of the SPA review, these were estimated to hold 90 % of the breeding population and 50 % of the wintering population (Stroud *et al.*, 2001). However, the UK bittern population has increased considerably since the review, and current coverage is estimated at 59 % of booming males (using 2008 data, S. Wotton, pers. comm.). Estimates of the size of the wintering population (e.g. of 100 birds during the SPA review, Stroud *et al.*, 2001) require updating.

Monitoring

Full surveys of the British bittern population during the breeding season have occurred annually since 1990. Surveys record numbers of booming males, deduced by triangulation of booming, and also nest locations, estimated from female feeding flights. The figure used to estimate population size is usually the minimum number of booming males, which includes only males that boomed for a week or more. There is also a maximum figure, including the males that boomed for less than a week, or that could not be confirmed as different to adjacent boomers.

Breeding ecology

Breeding habitat

In Britain, the bittern is mainly restricted to reedbed *Phragmites australis* dominated habitat, which is subject to loss and degradation (Gilbert *et al.*, 1996). Comparisons of reedbeds with

contrasting population trends have found that sites where declines have occurred have also undergone a higher degree of scrub encroachment, and recommended habitat management for bitterns should aim to maintain or restore reedbeds at an early stage of seral succession (Hawke and José, 1996, Tyler *et al.*, 1998). These recommendations have since been refined by radio-tracking eight adult males in Britain during the booming, moulting and wintering periods. This work found that home ranges constituted an average of 30 % open water (excluding ditches), 48 % reed edge (within 30 m of open water and ditches) and 16 % reed (over 30 m from open water and ditches). Home range size seemed to be driven by the available areas of reed-fringed open water, and the study recommended increasing open water within reedbeds, or reedbed habitat around existing open water (Gilbert *et al.*, 2005b).

Nest locations are situated in continuous blocks of reed, a mean of 2.8 ha in size and 100 m at their narrowest width (Gilbert *et al.*, 2005a). These were on average within 70 m of open water, 30 m of a ditch and surrounded by water 22 cm deep at the time the first egg was laid (Gilbert *et al.*, 2005a). Nest locations contained a greater length of reedbed and open water's edge within 100 m than random locations, and nests were located at points where deeper water was maintained into the drier parts of the season (Gilbert *et al.*, 2005a).

Breeding system

Polygyny is common in the bittern (Cramp and Simmons, 1977), with males having one to five mates (Gauckler and Kraus, 1965). Male bitterns make their booming call throughout the breeding season to advertise their territory (Cramp and Simmons, 1977). Males are strongly territorial from late winter until at least June to July, defending a large territory.

Distribution of nests with respect to the booming location is variable. In Britain, Gilbert *et al.* (2005a) found that 27 of 50 nests were within the nearest male's home range, with the others being an average of 130 m away, although boomer location was a poor predictor of precise nest location. In Poland, Gauckler and Kraus (1965) suggested that nests of various females were situated within the male's territory, usually within 50 m of the male's booming location, Cramp and Simmons (1977) state that this distance can be up to 500 m. In the Netherlands, a number of cases were observed where the nests were close to booming location, and foraging by males and females occurred within the immediate surrounding area, whilst in other cases, the female nested at some distance (up to 350 m) from a booming location (White *et al.*, 2006). The latter seems more common when suitable nest sites are further from foraging locations (White *et al.*, 2006). In Poland, booming locations associated with no nests, one nest, or a cluster of two to four nests were observed (Zimmerman, 1931, Gauckler and Kraus, 1965). In the UK, the smallest recorded distance between two active nests is 19 m (Gilbert *et al.*, 2005a). In Poland, clusters of nests were often 15 – 20 m apart (Zimmerman, 1931, Gauckler and Kraus, 1965). It should be noted that breeding habitat and behaviour is very different in Britain to that on the continent.

Clutch initiation in Britain occurs between around the 25th March and 20th June (Gilbert *et al.*, 2007). Mean clutch size is 4.0 (S. D. = 0.76, n = 14, range 3 - 5, Gilbert *et al.*, 2007). Replacement clutches may be laid (Mallord *et al.*, 2000, Dmitrenok *et al.*, 2005), and are likely to be common following nest failure at least up to the middle of May (Gilbert *et al.*, 2007). An incident of double brooding has been recorded, but this is probably rare (Mallord *et al.*, 2000). Incubation takes 25 - 26 days (Cramp and Simmons, 1977). There is no pair bond, with females being responsible for all care of young (Cramp and Simmons, 1977). Fledging occurs after 50 - 55 days (Cramp and Simmons, 1977), although females often provision young for longer than this (Mallord *et al.*, 2000). The main cause of chick mortality in the UK has been found to be

starvation, with daily losses of chicks due to exposure and/or starvation accounting for 76 % of those studied (Gilbert *et al.*, 2007).

Home range sizes and site fidelity

In the UK, male home range size estimated from the booming locations of males ranges from only a few hectares to over 100 ha (White *et al.*, 2006). Radio-tracking of eight adult males at two sites in the UK found median home range sizes of 14.6, 19.3 and 33.1 ha for the booming, moulting and winter periods respectively, with home range sizes being driven by area of open water (Gilbert *et al.*, 2005b). It is possible that these breeding home range estimates are large compared to a British average, as the sites where the radio-tracking was conducted were large by British standards.

Territories of male bitterns are often similar in area and position between years (Gilbert *et al.*, 2005b), even when different males are involved (Gilbert *et al.*, 2002). Analysis of male bittern vocalisations found that most resident males in Britain occupy the same territory from one year to the next (Gilbert *et al.*, 2002). Six ringed or radio-tracked individuals all remained in the same territory from one year to the next (Gilbert *et al.*, 2002). These individuals were used to provide a method of identifying individuals from vocalisations, and this found that there have been no incidents of males changing territories over the past ten years on the Norfolk Broads, west England and southeast England, just one incident at Leighton Moss, and a small number of within site or between site movements in coastal areas of Norfolk and Suffolk (Gilbert *et al.*, 2002). Similar results were found in a population of bitterns breeding at a marsh in central Italy, although distribution of booming males varied from year to year depending on changes in vegetation structure (Puglisi *et al.*, 2003), four radio-tracked individuals occupied the same or nearby areas during successive breeding seasons (Puglisi *et al.*, 2003).

Of six males tracked during the booming or moulting period and the subsequent winter, four remained on the same site through the winter. Two birds travelled 12.5 km and 53 km from the site in October, and returned in November and February, respectively (Gilbert *et al.*, 2005b). In the two cases where booming period home ranges were estimated in two separate years, most of the previous year's home range was used in the subsequent booming period (64 % and 75 %, Gilbert *et al.*, 2005b). The British radio-tracking work suggests that British birds may also remain during the winter, sharing the breeding site with visiting continental birds (Gilbert *et al.*, 2005b).

At Minsmere, a single female nested seven times between 1997 and 2001, moving long distances (e.g. 1.3 km in 1998, 1.5 km in 1999) between re-nesting attempts, but with the first (and second) nest each year occurring in a similar position (Gilbert *et al.*, 2005a).

Foraging flights

During the breeding season, males feed within their home ranges (Gilbert *et al.* 2005b; White *et al.*, 2006). In Italy, a single radio-tagged female fed mostly within 50 m of the nest (Puglisi *et al.*, 2003) and video-recording has shown them fishing directly from the nest (Adamo *et al.*, 2004). Nesting females can forage on foot. Distances travelled at Colfiorito, Italy, were usually less than 300 m (Adamo, 2002, Adamo *et al.*, 2004), and foraging areas usually coincided with male booming areas (Adamo *et al.*, 2004). However, British females almost always make flights to feed their young, probably due to lower availability of prey species near the nest (G. Gilbert, pers. comm.). Foraging flights of up to 2 km are regularly made (unpublished data in Gilbert *et al.*, 2005a).

Collision risk

Cramp and Simmons (1977) state that bittern fly at low or medium heights, but sometimes circle and soar to great heights. 'Gull calling' flights can occur between February and April. The gull call is given by both males and females, and is usually associated with pre-migration behaviour when birds fly in a circle above the reedbeds in the autumn and spring (Gauckler and Kraus, 1965). However, the calls used in spring can stimulate males to start booming or increase their booming rate (Witherby *et al.*, 1958, Gilbert *et al.*, 1994).

Although Cramp and Simmons (1977) suggested that most movement by bitterns occurred at night, radio-tracking of eight male bitterns at two sites in Britain found that most activity (assumed to be mainly feeding flights) occurs during the day, with birds being active for 62 % of fixes during the day, compared to just 20 % at night (Gilbert *et al.*, 2005b). The most important area for feeding males is the strip of reedbed adjacent to open water pools, with 88 % of active radio-tracking fixes, taken throughout the year, being within 30 m of water's edge (Gilbert *et al.*, 2005b).

Collisions with powerlines have been reported at Dungeness in the UK, as well as Aragon in Spain and the Po Delta in Italy (White *et al.*, 2006). The species seems to be generally quite prone to collisions, with deaths of adults and newly fledged young due to collisions with overhead cables, powerlines, fences, trees, a truck, a train and a car being reported (G. Gilbert, pers. comm.).

Breeding females may be at risk of collision during foraging flights, which may be over trees, houses, roads etc if these are in between the nest site and foraging area (G. Gilbert, pers. comm.). Males may also defend several fragmented bits of territory that may be separated by tall trees or other obstructions that they regularly fly over (G. Gilbert, pers. comm.).

Disturbance

Cramp and Simmons (1977) suggests that the bittern is more shy of disturbance than most species of heron, but can become accustomed to human activities. There is very limited specific information about disturbance to bitterns, although they do breed at nature reserves with large numbers of visitors, which may indicate tolerance to certain types of disturbance.

Sensitivity Criteria

All reedbeds where booming males or nests have been located between 1990, when monitoring began, and 2008 were mapped as 'high sensitivity'. This was thought reasonable to encompass breeding and main feeding areas due to the small and localised, albeit recovering, population. Some regional data exist on key foraging sites over the last few years, which could be of use when responding to individual proposals, and foraging flight information is also collected at the national level for nesting females, but was not yet available for use in this project.

Reedbeds that are currently being restored or created for breeding bittern also should be considered in wind farm proposals. Whilst it was not considered appropriate to include these sites in the mapping exercise if they were not currently being used, clearly there would be a conflict if proposals potentially jeopardised sites where lots of resources and conservation effort were going into habitat creation for bittern. Only information on breeding bittern has been included on the sensitivity map. Environmental Impact Assessments need to apply fieldwork protocols for bittern during Vantage Point watches at sites where they are likely to occur, notably where reedbed habitat is fragmented (see information above on foraging behaviour and territoriality) and in areas suitable for wintering bittern (although birds can be forced to disperse more widely and occupy unusual sites during periods of harsh weather).

Early consultation with local/regional conservation bodies will be helpful in defining study requirements for this species.

Acknowledgements

These sensitivity criteria were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thanks to Nick Droy, Gillian Gilbert, Kirsty Turner and Simon Wotton who provided helpful comments on an earlier draft of these sensitivity criteria.

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2. Bean goose

Introduction

The bean goose *Anser fabalis* is widespread and abundant across the Palaearctic, breeding from Scandinavia to eastern Siberia (Madge and Burn, 1988). Two races of bean goose occur in the western Palaearctic; the Taiga (or Western) bean goose *A. f. fabalis*, and the Tundra bean goose *A. f. rossicus* (Stroud *et al.*, 2001). Bean geese from the eastern parts of the breeding range migrate to winter in Europe, where the species occurs discontinuously in most European countries (Stroud *et al.*, 2001). Birds wintering in Britain are nearly all Taiga bean geese from western Russia and northern Scandinavia (Stroud *et al.*, 2001), although there is also a recent population of Tundra bean geese around North Warren (RSPB, 2005).

Both races breed at high latitudes eastwards from Fenno-Scandia, Taiga birds generally breeding further south and west than Tundra bean geese (Hearn, 2004). Most of the population of Taiga bean geese winters in the Baltic region (southern Sweden and Denmark, with smaller numbers in other neighbouring countries), and these birds, which originate from Finnmark and Russia, have increased in numbers in recent times (Parslow-Otsu, 1990). Smaller, generally decreasing numbers, from Scandinavian breeding grounds, winter on the North Sea plain, with Parslow-Otsu (1990) estimating the following population sizes: Jutland: 200 - 800 at five sites, Netherlands: 800 - 2000 at eight sites, Britain: 400 at two sites, although note this was in 1990. Bean geese are not threatened internationally (Hearn, 2004), however the British population of around 522 is relatively tiny (calculated using figures in RSPB, 2005, Maciver, 2006). There have also been dramatic declines in the breeding range and numbers in Norway and Sweden throughout most of this century, due to persecution and habitat changes, giving the British wintering population further importance (Parslow-Otsu, 1990). The British wintering population all hail from Scandinavia; Parslow-Otsu (1990) estimated that the total population in central Scandinavia at approximately 3000 individuals, with 17 % of these wintering in Britain.

In the first half of the 19th century, the bean goose was one of the most numerous goose species wintering in Britain, being a common visitor to northern Britain and East Anglia (Sutherland and Allport, 1994). A widespread decline began in the 1860s, and by the 1920s most populations had undergone severe declines or disappeared completely (Parslow-Otsu, 1990). There have been further declines since the 1940s, and now only approximately 522 bean geese (469 Taiga bean geese) winter in Britain (calculated using figures in RSPB, 2005, Maciver, 2006). The population occurs in small, widely dispersed groups, with there being just three regularly used sites. The oldest of these are visited by Taiga bean geese, and are at the Yare Valley in Norfolk, where there was a maximum count of 169 birds in winter 2005/06 (WWT, 2006) and Slamannan Plateau in Central Scotland, where there was a maximum count of around 300 birds during winter 2005/06 (occupied since 1985/6, Hearn, 2004). Most of the Yare Valley population occur in the Broadland SPA, although many sites in the area are used these are used infrequently and by small numbers of geese (T. Strudwick, pers. comm.). A third recent population of Tundra bean geese (maximum count of 53) winters at North Warren (RSPB, 2005).

Until recent years, the Yare Valley was the most important of the British sites (Owen *et al.*, 1986), but recent increases in the Scottish population mean that it now constitutes over half of the total British population (Maciver, 2006).

The bean goose is listed under Annex II/I of the EU Birds Directive (EC, 1979) and is on the amber list of species of conservation concern (Gregory *et al.*, 2002). The Yare Valley bean goose population mainly fall within the Broadland Special Protection Area, which was

estimated to hold 51.8 % of the British wintering population at the time of the last SPA review (Stroud *et al.*, 2001) and the Slammanan Plateau population is currently a proposed SPA.

Collision Risk

Moorehead and Epstein (1985) suggested that large wetland birds, such as geese, might be especially susceptible to collision. However, Hötker *et al.* (2006), in a review of wind farm impact studies, found that collisions in geese were relatively rare (although note that this review is the result of collation of records from a variety of sources, as opposed to controlled corpse searches at wind farms). Generally, there have been few recorded collisions of geese with wind farms, and this may indicate a lower likelihood of collision, although many studies have not corrected for scavenger bias, or correctly assessed goose numbers and activity at wind farm sites, although some recent studies are doing so. Fernley *et al.* (2006) calculated goose avoidance rates for incorporation into collision risk modelling, and produced high avoidance rates (around 99.93 %) although there are some methodological problems with the impact studies that these were based on (Pendlebury, 2006).

Hötker *et al.* (2006) reported one bean goose fatality at a wind turbine in Germany, and there have been two further recorded fatalities in Germany, although it was not clear whether these were bean geese or white-fronted geese *Anser albifrons* (T. Dürr, pers. comm.).

Disturbance

A review of wind farm impact studies found that in the non-breeding season, geese showed significantly more negative effects from wind farms (defined as lower densities after construction of the wind farm, or than at a control site) than neutral or positive effects (Hötker *et al.*, 2006). Hötker *et al.* (2006) reviewed 13 studies involving goose species, and found a median minimum distance from turbines of 300 m (mean = 373 m, S. D. = 226 m). On the basis of this, Hötker *et al.* (2006) suggested that important roosting areas for wildfowl should be kept free of wind farms, and recommended a buffer distance of at least 500 m for goose roosts. Several reliable studies of geese indicate negative effects up to 600 m from wind turbines, i.e. reduction in bird use of, or absence from, the area close to the turbines (e.g. pink-footed geese, European white-fronted geese, reviewed in Langston and Pullan, 2003), although there are also studies indicating much smaller displacement distances and no displacement in some cases. Average displacement distances of 100 m for wind turbines in lines and 200 m for turbines in clusters have been observed in pink-footed geese in Denmark (Larsen and Madsen, 2000). Disturbance displacement of European white-fronted geese in a German study was more marked, with substantially lower densities of feeding geese within 600 m of wind turbines (Kruckenberg and Jaene, 1999).

Displacement by other sources of disturbance (e.g. roads or people) is also discussed below, as it is of relevance to possible effects of wind farm construction or maintenance visits, as well as being of value in general consideration of possible susceptibility to disturbance. Bean geese on the Slamannan Plateau show selection for areas more distant from buildings and roads (Smith *et al.*, 1994), and Mooij (1982) also found avoidance of roads by bean geese in Germany. Avoidance of roads has also been shown for other geese species (pink-footed and greylag geese in north-east Scotland: Keller, 1996; pink-footed geese in Denmark: Madsen, 1985; white-fronted geese in Germany; Mooij, 1982).

Flocks of wintering pink-footed and greylag geese foraging on agricultural land in north east Scotland showed avoidance of roads, being found a median distance of 400 m (range 100 - 1100 m) but not within 100 m, of the nearest road (Keller, 1996). Mooij (1982) found that

feeding intensity of bean geese and white-fronted geese was reduced 250 m from quiet roads and 400 m from heavily used roads in Germany. In central feeding areas, feeding intensity was classified as 'severely reduced' within 250 m, 'irregular' between 250 - 350 m, 'frequent' from 350 - 450 m, 'regular' from 450 - 550 m but only 'intense' over 550 m from a road or other source of disturbance (Mooij, 1982). Madsen (1985), studying pink-footed geese in Denmark, found that grazing intensity was reduced within distances from roads that corresponded with flight distances shown by flocks when approached by a car. These were about 500 m in autumn, and 300 - 400 m in spring (Madsen, 1985). Madsen (1985) also found a link between disturbance of feeding geese at stopover sites with reduced breeding productivity. Keller (1996) found lower flight distances for pink-footed and greylag geese in north-east Scotland, ranging from 100 - 250 m (mean = 190 m). This was based on just six observations, as most flocks were found at distances where they did not react to cars, but it was suggested that the difference could be due to habituation, as geese in the study area in Scotland were exposed to much higher traffic density than the Danish population. Even at distances where geese did not show any reaction to passing cars they often reacted to cars stopping on the road or other kinds of disturbance such as tractors or walkers (Keller, 1996), this could be due to perceived risk. The presence of low power lines has been shown to reduce foraging intensity of white-fronted geese and bean geese in the adjacent 40 - 80 m in western Germany, although no effect was observed for lines over 60 m high (Ballasus and Sossinka, 1997).

Parslow-Otsu (1990) considered bean geese, and particularly Taiga bean geese, were more wary of disturbance than other species of geese. Increased levels of disturbance (from traffic and hikers) have been suggested as a reason for the abandonment of a historically occupied site in Ayrshire in the 1930s (Parslow-Otsu, 1990). Changes in land use and disturbance due to recreational activity or industry operations have been suggested as threats to the Slamannan bean goose population (Fraser, 2003, Hearn, 2004).

Sensitivity criteria

Most of the English population of Taiga bean goose occur within the Broadland SPA, which will be classified as 'high sensitivity'. The main traditional sites for English Tundra bean geese are at North Warren and Minsmere (RSPB, unpublished, D. Thurlow, pers. comm.). These sites will be mapped and buffered by 600 m, given that disturbance distances for bean geese from wind farms have not been quantified, but the species are considered particularly sensitive to disturbance. This represents the higher end of disturbance distances reported for geese (European white-fronted geese) in response to wind turbines (Kruckenberg and Jaene, 1999). Hötter *et al.* (2006) suggest buffer distances to goose roosts of 'at least 500 m', this maximum buffer of 600 m is considered justifiable due to the small, localised population. This area will be classified as 'medium sensitivity' on the basis that bean goose is not an Annex I species.

Acknowledgments

These sensitivity criteria are based on a previous version written for a Scottish sensitivity map (Bright *et al.*, 2006, 2008), updated in light of new information and to place them in the context of English population sizes and trends. They were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thanks to Dejan Bordjan, Tim Strudwick and Dave Thurlow who provided helpful comments on an earlier draft of these sensitivity criteria.

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3. Marsh harrier

Introduction

The marsh harrier *Circus aeruginosus* has a wide breeding distribution from western Europe and northern Africa throughout Asia to Pacific coasts on Sakhalin and northern Japan (Stroud *et al.*, 2001). It occurs throughout Europe, although the distribution in western Europe is discontinuous (Stroud *et al.*, 2001). Historically, marsh harriers were widespread throughout Britain and Ireland (Holloway, 1996), but the breeding population became extinct in 1899, probably due to persecution and drainage of wetlands for agriculture (Underhill-Day, 1998). Following recolonisation in 1911, the population then peaked at 15 pairs in 1958 (Underhill-Day, 1984). However, there was a further decline to just a single pair in 1971, and Clarke (1995) suggests this was due to organochlorine pesticides such as DDT, which caused declines in a number of raptor populations. A population recovery then occurred in Britain, probably attributable to withdrawal of these pesticides and a reduction in persecution, as well as reedbed creation due to flooding of large coastal areas in eastern England during World War II for defence reasons (Stroud *et al.*, 2001). Increases also occurred elsewhere in Europe, and increased population sizes in the Netherlands, due to development of huge reedbeds in the polders, probably aided the British population recovery (Clarke, 1995). Despite these increases elsewhere in Europe, there is still a general decline in the south of the species' range, including Spain (Hagemeijer and Blair, 1997).

In Britain, most of the breeding population occurs in England, with the main concentrations being in Norfolk, Suffolk and north Kent, although there are also populations in Lincolnshire, Humberside, Lancashire and southern and eastern Scotland (Underhill-Day, 1998). Nearly all records from 1983 - 1990 were in southern or eastern England (97 %), with 89 % being from Norfolk and Suffolk (Underhill-Day, 1998). By 1995, most records (88 %) were still from the southeast, but the proportion in Norfolk and Suffolk had declined to 60 % with small numbers breeding regularly in northwest England, Humberside and Scotland (Underhill-Day, 1998).

The 2005 national survey recorded 364 confirmed and possible breeding pairs in Britain (M. Eaton, pers. comm.), with the population having benefited from reedbed enhancement and recreation schemes in Norfolk and Suffolk. The marsh harrier is an Annex I species (EC, 1979) and is on the amber list of conservation concern (Gregory *et al.*, 2002). There are ten Special Protection Areas for marsh harriers, which covered an estimated 74 % of the British breeding population at the time of the SPA review (Stroud *et al.*, 2001).

Monitoring

An annual monitoring programme was run from 1983 to 1990, after which it was replaced with national surveys in 1995 (Underhill-Day, 1998) and 2005 (M. Eaton, pers. comm.). A large proportion of the population is still covered by annual records collated by the Rare Breeding Birds Panel (RBBP, M. Holling, pers. comm.).

Breeding ecology

Marsh harriers require open freshwater wetlands with dense, tall vegetation (particularly reedbeds) for nesting (Clarke, 1995). In England and Scotland, they breed in reedbeds and increasingly in intensive arable farmland (Underhill-Day, 1998). During 1983 - 1990 and 1995, nearly all recorded nests (86 %) were found in reedbeds, with 6 % in oilseed rape and 7 % in winter cereals, and 22 % of nests for which surrounding habitat was recorded during the 1995 national survey were associated with arable crops, particularly winter cereals (Underhill-Day, 1998). First breeding occurs at one to four years (Sternalski *et al.*, 2008). Birds return to

breeding areas from mid March (Clarke, 1995). Marsh harriers are frequently polygynous, for example, 21 % of all pairings recorded between 1983 and 1995 were bigynous or trigynous (Underhill-Day, 1998). Mean clutch size is 4.4 ($n = 19$, Underhill-Day, 1998), incubation lasts about 33 days and young fledged at around 28 days (Clarke, 1995). About 24 % of all nest failures during 1983 - 1990 and 1995 were due to human persecution and disturbance (Underhill-Day, 1998).

The majority of food is supplied by the male (Underhill-Day, 1990, Clarke, 1995). Fernandez and Azkona (1994) observed ten breeding pairs in Spain, for 128 hours during incubation and 538 hours during the nestling stage. Approximately 80 % of observed prey deliveries were by the male ($n = 467$), and the female supplied 18 % of observed food items directly to the nest. Male marsh harriers transfer food to females in flight during the incubation and nestling stages (Cramp and Simmons, 1980, Johannesson, 1975) and Fernandez and Azkona (1994) found that 60 % of food was supplied by males in this way.

Winter roosts

Gatherings of more than 30 birds have been recorded in north Norfolk, over 20 in Lincolnshire and up to 15 on the Isle of Sheppey in Kent (Hadrill, 1989). North Norfolk roosts grow in size during August and early September, but can disappear suddenly in mid September when migration occurs (Clarke, 1995). On Sheppey, the age composition of birds varies during late August and early September, indicating passage birds are involved, but with some marked individuals being present for the duration (Clarke, 1995). Winter communal roosts form at just a few sites in Britain, in Norfolk, Suffolk and Kent (Clarke, 1995).

Studies of communal roost sites in North Kent found they were used in 13 - 15 winters, with other sites being used occasionally, generally by small numbers (< 7) of birds (Oliver, 2005). Most wintering birds were female, with males only accounting for about 10 % of winter totals (Oliver, 2005). It is generally the case that most adults wintering in Britain are female (Underhill-Day, 2002). The marsh harrier is a partial migrant (Underhill-Day, 2002), and although birds were not tracked at North Kent, Oliver (2005) suggested that the wintering population substantially reflects the breeding population, and found that one distinctive female observed in winter was also present the following breeding season. Birds usually arrive at roost sites singly in low flight (2 - 3 m, but up to 30 m), before repeatedly circling over the roost at a height of up to 30 - 40 m above the roost before settling (Oliver, 2005). Occasionally, group-circling behaviour over the roost occurs, lasting several minutes (Oliver, 2005).

Home range sizes and site fidelity

Natal philopatry

Research by radio-tracking and ring re-sighting of European populations have found varying degrees of natal philopatry (Ilyichew, 1982, Witkowski, 1989, Bavoux *et al.*, 1992, Buczek, 1995, Clarke, 1995, Sternalski *et al.*, 2008), although some of this could be explained by whether or not the populations in question were migratory (Sternalski *et al.*, 2008).

Site fidelity

Clarke (1995) states that marsh harrier breeding sites are generally traditional. Sondell (1970) found that three pairs in central Sweden bred in the same parts of the reedbeds in two subsequent years, Altenburg *et al.* (1987) also found site fidelity in a population in Holland, and Witkowski (1989) recorded one male holding the same nesting territory for four years, and another for two in Poland. A study in the Fens showed fidelity to nest sites between

years, with nest sites being more predictable than those for Montagu's harrier (M. Meadows, pers. comm.).

Home range sizes

Marsh harrier breeding densities seem to be lower than those found in hen and Montagu's harriers (Clarke, 1995), however mean nearest-neighbour distances as little as 100 m have been recorded (Berg and Stieffel, 1968). Witowski (1989) recorded a mean density of down to 26.5 ha per pair in Poland.

During courtship and incubation periods, activity is concentrated on the breeding marsh and the breeding territory at this stage is typically just a radius of 100 - 300 m around the nest (Sondell, 1970, Clarke, 1995). Once the young have hatched, males extend their activity beyond the nesting territory and make longer foraging flights often into surrounding agricultural land (Clarke, 1995). Sondell (1970) suggested that the foraging territory is defended, whilst others suggest they are not and can be overlapping (see Clarke, 1995). Underhill-Day (1990) reported that females leaving the nest spend most of the time close by, with occasional absences from the nesting area.

Foraging ranges from a study of birds in Holland and France ranged from 250 to 800 ha for males, and 80 to 400 ha for females (Schipper, 1977). Radio-tracking of breeding birds in central western France in 2001 found a mean range of 349 ha (S. D. = 185, Sternalski *et al.*, 2008). In Holland, territory size ranged from 310 ha to 2610 ha over a breeding season (Altenburg *et al.*, 1982). Much smaller foraging range sizes have been recorded for semi-colonial breeding birds in Poland (Witowski, 1989). At Titchwell, foraging ranges of marsh breeding birds over arable farmland were 1250 ha and 1000 ha in 1982 and 1983 (Sills, 1988). In East Anglia, Underhill-Day (1990) recorded mean foraging range sizes of 569 ha (+/- 132) during courtship, 750 ha (+/- 132) during incubation, increasing to 1125 ha (+/- 147) during the nesting period, based on observations of six males and one female. Seaton (1999) recorded a territory size for one polygamous male of 150 to 350 ha, and territory sizes for the two females ranged between 75 and 125 ha (figures using 95 % kernel method quoted). Territory size is likely to depend upon factors such as number of females, brood size (Underhill-Day, 1990), habitat quality, prey density and the stage of the breeding cycle (Clarke, 1995).

Maximum hunting distances of male marsh harrier in Holland and France were only 1.5 km to 3.1 km, and 1.4 km to 1.8 km for females (Schipper, 1977), but Glutz von Blotzheim *et al.* (1971) found that birds may hunt up to 5 - 6 km from the nest, and exceptionally up to 8 km. In Cambridge, Clarke (1995) observed a male foraging up to 12 km from the nest.

Collision risk

Marsh harrier typically fly low above ground level at about 2 to 6 m when foraging (Clarke, 1995), although flight height seems to be linked to height of vegetation (Schipper, 1977). On account of this generally low flight height, Clarke (1995) suggests that, although harriers occasionally collide with power cables, the national grid did not constitute a serious threat to marsh harriers. However, Clarke (1995) suggests there could be a risk during migration and travelling flight.

Risk of collision could also occur during sky dancing breeding displays. As in other harrier species, these are high flying climbing and tumbling displays, with a territorial or courtship function (Clarke, 1995). The display begins with the bird soaring to flights of perhaps hundreds of metres ('to at least 75 m and often considerably more', Cramp and Simmons, 1980) in circles, followed by a series of ascents and descents (Clarke, 1995). The display may

last for 10 to 15 minutes, and is generally performed by the male up to the end of incubation, but may also on occasion be performed by the female (Clarke, 1995). High circling usually occurs above the nesting territory and up to 1 km beyond (Axell in Cramp and Simmons, 1980)

Disturbance

The effect of disturbance by crayfish fishermen on six breeding pairs of marsh harrier was investigated in Spain in 1991 (Fernandez and Azkona, 1993). The fishermen frequently enter the reedbeds to place or empty traps, and increased numbers of fishermen have led to fishermen being distributed further from paths and penetrating the reedbeds more. Harriers took a mean of 18 minutes to return to the nest following flushing by human disturbance (range = 1 - 89, S. D. = 21, n = 33). Disturbance during incubation and nesting reduced the time spent in the nesting area, particularly during incubation, and also the number of food items delivered to the female during and chicks. There was an increase in flight activity (by a factor of three in males and ten in females) and more time was spent chasing and alarm calling during disturbed periods. Although disturbance had no observed effect on annual productivity, nestlings of disturbed birds exhibited higher levels of blood urea (an indication of poor nutritional condition), and it was suggested that long-term effects on reproductive success could occur due to increased parental energy expenditure and/or lower nestling condition. Observations in this study were made from a distance of 500 m from nests, presumably the distance at which observers were not considered to affect behaviour.

Sensitivity to disturbance can also limit availability of nesting and foraging habitat in the marsh harrier. Sills (1983) found that birds at Titchwell marsh RSPB reserve in Norfolk would not build nests closer than 450 m from a path, and on average sited nests 600 m and 800 m from two heavily used footpaths. Gamauf (1993) found that birds in Neusiedler See, eastern Austria, would not fly closer than about 90 m to visible tourist activity in the open, and that routes frequently used by walkers and cyclists resulted in an avoided corridor about 240 m across, thus limiting available foraging habitat (Gamauf, 1993). Ruddock and Whitfield's (2007) survey of expert opinion suggested a 300 - 500 m buffer to disturbance for marsh harrier, although sample sizes (in terms of numbers of experts contacted) were low for this species.

Sensitivity Criteria

Breeding locations from the 2005 national survey were plotted, the area within 1 km of these classified as 'high sensitivity', and the additional area within 2 km of breeding locations classified as 'medium sensitivity'. This is on the basis that Cramp and Simmons (1980) suggest that most aerial displays occur within 1 km of the nest, but observations suggest that these can occur over a wider area (e.g. 2 km, I. Higginson, pers. comm.; 3 km, J. Day, pers. comm.) and that, whilst foraging flights are generally low to the ground, birds can return from foraging areas by spiralling to gain height before descending on the way to the nest (I. Carter, pers. comm.). Assessments of individual proposals should check for use of suitable foraging habitat within 3 km of marsh harrier nest sites.

Recent roost locations were also plotted. As data on roost locations are collected by a variety of regional counts, rather than by a systematic survey, age of survey varied; the most recent year's data available was used in each case. Roosts holding over 1 % of the summed total number of birds were included on the map. The area within 1 km of these was classified as 'high sensitivity', to account for movement of roost location and the fact that aerial displays may occur above roost sites.

Acknowledgments

These sensitivity criteria were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thanks to Ian Carter, John Day and Ian Higginson who provided helpful comments on an earlier draft of these sensitivity criteria.

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4. Hen harrier

Introduction

The Eurasian hen harrier *Circus cyaneus* was previously considered polytypic, with three separate races occurring in the Palaearctic, North America and South America (Stroud *et al.*, 2001). However, recent work suggests three separate species; the Eurasian hen harrier *Circus cyaneus*, which occurs throughout the Palaearctic, the northern harrier *Circus hudsonius*, which occurs in North America, and the cinereous harrier *Circus cinereus*, which occurs in South America (Simmons, 2000). Global population trends have not been quantified, but there is evidence of a population decline (Ferguson-Lees and Christie, 2001). The hen harrier is widely but patchily distributed across much of northern and central Europe, with the European breeding population numbering between 8332 and 10 840 pairs (Hagemeijer and Blair, 1997). The European breeding population underwent a large decline between 1970 and 1990 (BirdLife International, 2004). These declines abated to some extent during 1990 to 2000, with many populations stabilising, and the species declined only slightly overall (BirdLife International, 2004). Nevertheless, the population remains far below the level preceding the declines (BirdLife International, 2004).

Hen harrier were once common and widespread throughout the UK, but became virtually extinct in mainland Britain by the start of the 20th century, due mainly to persecution by gamekeepers (Watson, 1977). At this stage, the remaining population was largely confined to Orkney and the Outer Hebrides (Watson 1977). It then increased around the middle of the century, due to a reduction in shooting interests, and protective legislation, which was introduced in 1954 (Watson 1977). Upland afforestation between the 1940s and 1970s favoured hen harrier recolonisation of mainland Britain, at least in the early stages of development, and the population increased to an estimated 500 breeding pairs with a further 250 - 300 pairs on the island of Ireland by the mid 1970s (Watson, 1977).

The UK and Isle of Man breeding population was estimated at 578 - 700 pairs in 1988 - 9 (Bibby and Etheridge, 1993), and 570 territorial in 1998 (Sim *et al.*, 2001). The 2004 national survey showed a 41 % increase for the estimated UK and Isle of Man population to 806 territorial pairs (Sim *et al.*, 2007). However, regional trends have varied, and the English population has decreased during this time, from an estimated 18 pairs in 1990 and 19 pairs in 1998 to just 11 territorial pairs in 2004 (Sim *et al.*, 2007). By 2004, the English breeding range was restricted to the Forest of Bowland in Lancashire (Sim *et al.*, 2007). Hen harrier were also found in two counties in south east England, but no confirmed breeding or territorial behaviour occurred (Holling and the RBBP, 2008). The proportion of pairs in England on grouse moors fell from 54 % to 20 % between 1998 and 2004, suggesting that persecution may be responsible for the decline (Sim *et al.*, 2007). Persecution has been shown to be on a sufficient scale to limit the numbers and range of the UK hen harrier population (Etheridge *et al.*, 1997, Potts, 1998). The recent recovery of the Welsh population is attributed in part to decreased persecution, which can have a large influence on breeding productivity even when only aimed at a minority of the breeding population (Whitfield *et al.*, 2008). A study of English grouse moors found that only 28 % of territorial females bred successfully, compared to 70 % on moors with special protection schemes (Stott, 1998), and it has been estimated that there is enough suitable habitat in England to support 232 pairs of hen harrier in the absence of persecution (Potts, 1998).

The hen harrier is an Annex 1 species (EC, 1979) and is on the red list of conservation concern (Gregory *et al.*, 2002). There are 14 Special Protection Areas for hen harrier in the UK, two of which are in England, and these were estimated to hold 47 % of the British breeding

population at the time of the last SPA review (Stroud *et al.*, 2001). Twenty SPAs are designated for non-breeding hen harrier, 16 of which are in England, and in total these hold 33 % of the British wintering population (estimated at 750, Stroud *et al.*, 2001).

Monitoring

Surveys of breeding hen harrier have been conducted in 1988/9 (Bibby and Etheridge, 1993), 1998 (Sim *et al.*, 2001) and 2004 (Sim *et al.*, 2007). Winter roost locations have been surveyed for the past 25 years (e.g. Clarke and Watson, 1990, 1997, Hawk and Owl Trust web-site, 2008a).

Breeding system

Hen harrier are monogamous in many cases, although some populations are polygynous (Cramp and Simmons, 1980). A clutch of four to six (although occasionally as large as eight or even 12, mean on Orkney = 4.46) is laid from late April to late May (Cramp and Simmons, 1980). One clutch is laid, although replacements can be laid following loss at the egg stage (Cramp and Simmons, 1980). Incubation lasts from 29 - 39 days (Cramp and Simmons, 1980). Young are semi-altricial and nidicolous (Cramp and Simmons, 1980). Young are brooded continuously for the first 10 - 15 days, during which the male brings food; later on the female may also bring food, direct feeding is always by the female (Cramp and Simmons, 1980). Breeding hen harrier show preference for foraging in young first rotation conifer forests, with heathland and grassland habitats being selected over closed canopy woodland (Madders, 2000). Young may leave the nest at around 15 days and crouch in nearby vegetation, but the fledgling period is usually 32 - 42 days, after which the young are not fully independent for a further few weeks (Cramp and Simmons, 1980).

Home Ranges

Site fidelity

In the absence of illegal persecution, breeding hen harrier would be relatively sedentary, with wing-tagged harriers usually nesting in the same place having bred once (Etheridge *et al.*, 1997). The median distance moved between breeding sites in Scotland in successive years found by Etheridge *et al.* (1997) was 0.71 km (n = 51). Picozzi (1984) found that on Orkney females that moved to a new territory moved further following a failed (mean = 2.29 km, S. D. = 2.41, n = 73) than a successful breeding attempt (mean = 1.32 km, S. D. = 0.90, n = 44). However, no such effect was found by Etheridge *et al.* (1997).

Range size

Harriers are not known to actively defend hunting territories and hunting ranges can overlap considerably (Watson, 1977, Redpath, 1991, Arroyo *et al.*, 2005), which led Watson (1977) to suggest that males will only defend a nest territory of 600 m in diameter. However, interactions between neighbouring birds have also been observed (Picozzi, 1984, García and Arroyo, 2002) probably leading to avoidance of occupied areas.

Habitat close to the nest is used disproportionately when hunting during the breeding season (Picozzi, 1978, Madders, 2003, Thirgood *et al.*, 2003), and this has also been shown during the pre-settlement period (Amar and Redpath, 2005). A number of estimates of the size of hunting territories have been made and range size appears to vary considerably between regions, probably as a function of nesting and foraging habitat availability (Madders, 2004). Males' hunting ranges are larger than those of females, and home range sizes increase as the nesting period progresses (Madders, 2004).

Picozzi (1978) made visual observations of hen harrier in Scotland and suggested that most hunting by females was carried out within sight of the nest until young were feathered. It was

estimated that ranges were larger than 15 km² in area, equivalent in area to a circle with a radius of 2.2 km. Thirgood *et al.* (2003) found a significant positive relationship between females hunting and nest site proximity at Langholm, although this was less significant for males.

Similarly, Martin's (1987) study of radio-tracked breeding northern harriers *Circus hudsonius* (a very close relative of the hen harrier) in North America found that female harriers never ranged further than 2 km from their nest sites, whereas males spent 26 % of their time ranging over 2 km from the nest. They estimated ranges of 15.7 km² and 1.13 km² for males and females respectively, equivalent to the areas of circles with radii of 2.3 and 0.3 km. Another North American study estimated range sizes at 72 - 366 km² (equivalent to the area of a circle with radii of 2.6 - 6.1 km, Thompson-Hanson, 1984).

Visual observations of hen harrier in continental Europe estimated average range sizes of 5 km² and 1.4 km² for male and female hen harrier, respectively (equivalent to the areas of circles with radii of 1.3 km and 0.7 km, Schipper, 1973). More recently, radio-telemetry studies at two Scottish sites, on Orkney and at Langholm, have estimated average ranges of 9.3 km² for males and 4.6 km² for females (equivalent to the areas of circles of radii 1.7 km and 1.2 km, Arroyo *et al.*, 2005). Furthermore, it has been shown at Langholm that the amount of certain habitats within a 2 km circular radius of a nest influences the type of prey brought to that nest (Amar *et al.*, 2004). Watson (1977) considered that in Scotland, male hen harrier will hunt more than 3 km from the nest, and are often recorded 4 km or more from the nest site. Watson also concluded that female hen harrier can regularly hunt 2 km to 3 km from the nest, and some have been recorded foraging as far as 8 km from the nest. Hunting territories are not circular and so radii estimated from ranging areas give a fairly crude illustration of the extent of a hunting range.

Roosting behaviour

The UK winter distribution of hen harrier is significantly different from the breeding distribution, with birds wintering in the lowlands, particularly around the coast (Stroud *et al.*, 2001). There are significant concentrations on the south and east coast, particularly the East Anglia estuaries and fens, the Greater Thames estuary and Solent area, with birds also occurring in lowland heaths and chalk downland, for example in Hampshire and Dorset, on downland in Oxfordshire, Berkshire and Wiltshire, and in the East Anglian Brecks (Lack, 1986, A. Dobson, pers. comm.). Although a proportion of birds are resident throughout the year, most young birds disperse widely (Stroud *et al.*, 2001). Hen harrier gather at communal winter roosts, which are surveyed annually by the Hawk and Owl Trust (e.g. Clarke and Watson, 1990, 1997, Hawk and Owl Trust web-site, 2008a). Roosts sites exceptionally hold up to 90 birds (in the Isle of Man), but more usually numbers are in the order of 3 - 4 birds (Dobson, pers. comm.). These are usually located in wetlands, such as carr woodland, marshes and reedbeds, but sometimes also occur on heather moorland, lowland heath, conifer plantations (Stroud *et al.*, 2001) and also in long grass (e.g. in north west England, A. Dobson, pers. comm.). There is some southward movement in winter to continental Europe, with birds ringed in the breeding season in Scotland and Wales having been recovered from France and Iberia as well as England (Lack, 1986), and there are also movements of birds from continental Europe to Britain in periods of severe weather (Stroud *et al.*, 2001). It was previously thought that most of the wintering population consisted of migrants from the continent, however, recent work suggests that the majority of birds at British roosts are from the Scottish and English breeding populations (Dobson, 2008).

There has been an apparent decline in wintering numbers since the 1980s, and although this could in part be attributable to reduced survey effort (Hawk and Owl Trust web-site, 2008b), Clarke and Watson (1997) found that evidence for a decline remained once this was accounted for. In 1985 - 86, 202 roost sites were located in Britain, and 12 in Ireland, of these communal roosting had been recorded at over 90 % of sites, and 43 % were used each winter (Clarke and Watson, 1990). Overall, 39 % of sites were in England, although coverage for England was more comprehensive (Clarke and Watson, 1990). It was estimated that England held about 300 wintering birds, Scotland about 400, Ireland fewer than 150 and Wales about 50 in the mid 1980s (Clarke in Lack, 1986). About 33 % of the British wintering population occurs within 20 Special Protection Areas, 16 of which are in England. Birds can range considerable distances from roost sites, with evidence for dispersal of at least 16 km from roost sites in southwest Scotland (Watson, 1977). Aerial displays over winter roost sites have not been documented, although birds can approach roosts at dusk at heights of about 50 m (A. Dobson, pers. comm.).

Disturbance

Madders (2004) estimated that, based on hen harrier's tolerance to other disturbance events, nesting and foraging harriers may be displaced from distances of around 500 m from a wind farm construction site, with some disruption to nesting and foraging behaviours extending up to 1 km along lines of sight. Disturbance distances from turbines were estimated at 250 m, with some disruption to nesting and foraging behaviours extending up to 500 m (Madders, 2004). A more recent review of eight studies of displacement effects of hen harrier by wind farms in the USA and continental Europe found good evidence of displacement in only one of these (Madders and Whitfield, 2006, Whitfield and Madders, 2006). This study, by Johnson *et al.* (2000) found evidence of both small scale (< 100 m from turbines) and larger scale avoidance of turbines by harriers in the year following wind farm operation. Whitfield and Madders (2006) concluded that if displacement of foraging occurs it is likely to be limited to within 100 m of turbines. However, a recent field study at 12 wind farms and paired control sites in Scotland and England found reduced hen harrier flight activity within 250 m of turbines (Pearce Higgins *et al.*, unpubl.). Preliminary results relating to nest site selection in Argyll in Scotland, and Northern Ireland, indicate that local displacement of nesting attempts may occur within 200 - 300 m of turbines (Whitfield and Madders, 2006).

In terms of other sources of disturbance, analysis of nest site distribution in relation to roads and human settlements has suggested avoidance of human activity (Tapia *et al.*, 2004). Another study found that northern harrier nests did not occur closer than 188 m from the nearest building (Combs-Beattie, 1993). Hiking trails have also been found to decrease abundance of wintering harriers in riparian zones (Fletcher *et al.*, 1999). However, northern harriers have been found to be extremely tolerant of aircraft and missile bombing, with a harrier continuing to forage despite noise levels of 80 – 87 dB and explosions within 60 m (Jackson *et al.*, 1977).

Disturbance free zones of 500 - 600 m (Petty, 1998), and 500 - 1000 m (Currie and Elliot, 1997) have been recommended around occupied hen harrier nests in advice for forestry workers in Britain, although it is not clear what these distances were based on. Romin and Muck (1999) recommended a disturbance-free buffer of 500 m for northern harriers during the breeding season, although, again the basis for this is unclear. Ruddock and Whitfield (2007), in their survey of expert opinion, found that a maximum disturbance free buffer of 500 – 750 m was suggested for hen harrier.

Collision Risk

Hen harrier typically hunt low over the ground (Watson, 1977) but may be more at risk of collision with turbines during display flights or flights made as newly fledged birds (Madders, 2004). High circling is common over the nesting area, particularly in spring, and can be by one or both birds (Cramp and Simmons, 1980). This may be followed by sky dancing displays, usually conducted by the male (Cramp and Simmons, 1980). Madders (2004) suggested that display flights by male harriers occur mainly within 500 m of the nest, but can be made up to 1 km away, and thus collision risk is probably highest within about 500 m from the nest, declining to there being little risk at a distance of approximately 1 km away. There is no direct evidence of birds colliding with obstacles during aerial displays, probably because the likelihood of observing it is very low. However, in Argyll, west Scotland, at least two male hen harrier were killed due to collision with overhead lines close to traditional nesting areas in late April/early May, the peak display period (Madders, 2004). Juveniles are regularly killed in their first few weeks due to collision with fences (Madders, 2004). It is not known how harriers respond as they approach operational turbines but if they respond by actively gaining height then this would substantially increase collision risk. This risk would potentially affect harriers throughout the year and would apply to breeding birds several kilometres from their nests (Madders, 2004).

More recently, Whitfield and Madders (2006) reviewed nine studies of collision rates at wind farms where hen harrier occur; mortalities of hen harrier were recorded in three studies. Two of these involved migrating birds, and the third involved breeding and passage birds. Only one of the studies recorded mortality of more than one bird, this was at the Altamont Wind Resource Area, California, where three casualties occurred over five years. The review concluded that mortality rates are probably only rarely expected to be high, and this may be when turbines are located in the vicinity of several nest sites. A review by Hötter *et al.* (2006) of collisions at wind farms (mainly in Europe) does not report any hen harrier collisions. However, it should be noted that this review is the result of collation of records from a variety of sources, as opposed to controlled corpse searches at wind farms.

Sensitivity criteria

The main risk of collision for hen harrier is likely to be during aerial display flights, which are usually conducted within 1 km of the nest. However, given the extremely small size of the English population, and the fact that its size and distribution is dramatically constrained by persecution, nest locations for the past 10 years will be plotted and buffered by 2 km, as foraging flights usually occur within 1 km to 2 km of the nest, and this area classified as 'high sensitivity'. Locations of roost sites from the hen harrier winter roost survey 2004/05 (the most recent readily available data) were also included, with roosts included if they held over 1 % of the British wintering population (using the population estimate of 750 from Stroud *et al.*, 2001). In the absence of detailed information on flight behaviour at roost sites, these were buffered by 1 km, and classified as 'high sensitivity'.

Acknowledgments

These sensitivity criteria are based on a previous version written for a Scottish sensitivity map (Bright *et al.*, 2006, 2008), updated in light of new information and to place them in the context of English population sizes and trends. There were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thanks to Ian Carter, Andrew Dobson, Martin Kerby, Peter Robertson, Innes Sim, Pete Wilson and Tim Youngs who provided helpful comments on an earlier draft of these sensitivity criteria.

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5. Montagu's harrier

Introduction

The Montagu's harrier *Circus pygargus* is a migratory species, breeding from north Africa and Europe to central Russia and Asia (Cramp and Simmons, 1980, del Hoyo *et al.*, 1994). All populations are migratory, with populations of western Europe wintering in west Africa, and Asian breeding populations wintering on the Indian subcontinent and Sri Lanka (García and Arroyo, 1998). There is some evidence that birds often stay in the winter quarters during their first summer (Mead, 1973, García and Arroyo, 1998).

Southern Britain represents the northwestern limit of the Montagu's harrier's range, and it has probably always been quite a scarce breeding species in Britain (Cramp and Simmons, 1980). The population declined during the 19th century, to just to four pairs between 1890 and 1918, due to persecution and habitat destruction (Cramp and Simmons, 1980). At this point the population was concentrated in Norfolk, however it then spread to southern England and south Wales until the 1930s, when the population reached 15 to 25 pairs (Cramp and Simmons, 1980, Sharrock *et al.*, 1983). By the mid 1950s, the population peaked at 40 to 50 pairs, concentrated in southwest England (Cramp and Simmons, 1980, Sharrock *et al.*, 1983). Following this, the population declined again, with no breeding occurring in 1974 - 75 (Sharrock *et al.*, 1983). Similar declines have occurred in parts of northwest Europe (Gamell, 1979). There has been some recovery since, with an estimated 13 nesting attempts in 2007 (M. Thomas, pers. comm.). Population declines have also occurred elsewhere. The reasons for the decline during the end of the last century are not known, but organochlorine pesticides, which affected a number of raptor populations during the 1950s and 1960s (Newton and Haas, 1984, Clarke, 1996), human persecution (Brown, 1976) and changes in climate (Wilkinson, 1984) possibly via drought conditions in central Africa leading to decreases in locust populations (Underhill-Day, 1990) have been suggested.

The Montagu's harrier is an Annex I species (EC, 1979) and is on the amber list of conservation concern (Gregory *et al.*, 2002). No Special Protection Areas are designated for Montagu's harrier (Stroud *et al.*, 2001).

Breeding Ecology

Breeding habitat

The Montagu's harrier is a ground-nesting species, which until recently bred in marshes or steppe grasslands (Simmons, 2000). However, loss of natural habitat means that the species now mainly nests in cereal crops (Clarke, 1996). Natural vegetation is still used in some European areas (Clarke, 1996, García and Arroyo, 2003, Limiñana *et al.*, 2006a), and preference for natural habitats has been suggested (Leroux, 1987), with nests in natural habitats also having a higher breeding success (Pandolfi and Giacchini, 1991, Castaño, 1997, Corbacho *et al.*, 1997, Koks *et al.*, 2001, Millon *et al.*, 2002, Limiñana *et al.*, 2006a). It is likely the main shift from nesting in natural habitats to cereals occurred during the 1950s and 1960s, when changes in agricultural practices dramatically altered the landscape (Pain and Pienkowski, 1997). In Europe, the proportion of Montagu's harrier nesting in cereals increases westwards (Arroyo, 1995). Polish harriers have only recently begun to shift from marshes into crops (Krogulec and Leroux, 1994), and a study in the 1980s found that over 80 % of the Russian harrier population still bred in natural and semi-natural habitats (Flint *et al.*, 1984). Populations nesting in cereals are vulnerable to loss of eggs and nestlings through harvesting activities, one of the largest threats to the population in the Western Palearctic (Arroyo *et al.*, 2002), and require human intervention to be sustainable (Limiñana *et al.*, 2006b).

In Britain, 82 % of nests between 1980 and 1989 were in cereals, with most others being in grass (silage, seed and rough grazing), where nesting success is lower (Underhill-Day, 1990). Most nests since 1976 have been individually protected due to the low number, so nest losses due to harvesting operations have been low (2 % of all known failures since 1900, Underhill-Day, 1990).

Breeding system

The species is predominantly monogamous, although polygyny is observed occasionally (Cramp and Simmons, 1980). It is likely that the same pairs could form in subsequent years due to faithfulness to the same nesting territory, although there is no evidence for this (Cramp and Simmons, 1980). First-breeding occurs at two to three years (Cramp and Simmons, 1980). Montagu's harriers arrive in Britain from late April to early May (Clarke, 1996) and egg laying begins from mid to late May (Cramp and Simmons, 1980). Clutch sizes range from 3 - 10 (Cramp and Simmons, 1980), with a mean of 4.16 being recorded from 32 nests (BTO nest record card data, Clarke, 1996). Replacement clutches are laid following early clutch loss (Cramp and Simmons, 1980). The incubation period is usually 28 to 29 days (Clarke, 1996; range of 27 to 40 cited in Cramp and Simmons, 1980), and fledging occurs at 31 to 33 days (Clarke, 1996; 35 to 40 days cited in Cramp and Simmons, 1980). All brooding and direct feeding is performed by the female, but the male does most of the provisioning of the female and brood, with aerial food passes being quite common (Cramp and Simmons, 1980, Clarke, 1996).

Home ranges and site fidelity

The Montagu's harrier is often semi-colonial on breeding grounds (Cramp and Simmons, 1980). In Britain, nesting groups are small, for example three nests in a conifer plantation 0.6 ha in area, a minimum of 20 m apart; four nests in a different plantation 50 - 60 m apart (Khan in Cramp and Simmons, 1980).

It is not clear how site faithful Montagu's harrier are. Thirty eight nest sites in natural habitat were monitored between 2001 and 2004 in France; 40 % of Montagu's harriers nested at the same site more than once, and in successive years actual nests of individual females were on average 449 +/- 129 m apart (Cormier *et al.*, 2008). Cramp and Simmons (1980) state that 'the same nesting territories are occupied in successive years. At least at times, the same birds are involved', and 'In Cornwall or Devon, birds return to nesting territories occupied the previous year singly, male or female first, or in pairs'. However, it should be noted that a lot of this may refer to birds nesting in natural habitat, and that the majority of nest sites in Britain are now in cereals. Nest location within cereals is determined largely by vegetation height, with sites where vegetation is equal in height or higher than the surrounding vegetation (Claro, 2000, Arroyo *et al.*, 2004), this varies between years altering the distribution of breeding birds (Claro, 2000, Arroyo *et al.*, 2003). Clarke (1996) states that Montagu's harriers show a strong tendency to return to traditional breeding sites year after year, but that returning to the exact same nest site is rare. Observations in Wiltshire would agree with this, with some traditional areas being used for nesting each year, but individual nesting fields moving due to crop rotation (P. Castle, pers. comm.). Ryves (1948) recorded a nest moving only 15 yards, but this was in an area with just a small patch of cover. It seems that Montagu's harrier are site faithful in some areas in Britain, such as west Norfolk and the New Forest, but not others such as Wiltshire and Cambridgeshire (Clarke, 1996). In the Fens, Montagu's harrier nest sites are considered to be less predictable than those of marsh harrier (M. Meadows, pers. comm.).

Hunting ranges

Breeding birds occupy a nesting territory, but only the immediate vicinity of the nest is defended by both sexes (Cramp and Simmons, 1980). Cramp and Simmons (1980) state that the male's hunting range is situated away from the nesting territory, and does not usually overlap with those of other males, but it is not known if this is actively defended as a territory. García and Arroyo (2005) suggest that Montagu's harriers do not defend feeding territories, and Salamolard (1997) found no difference in home range size between non-breeding and breeding birds.

Male Montagu's harriers hunt over long distances (Studinka, 1942, Schipper, 1973, Thiollay, 1968). In Flevoland, Holland, hunting ranges have been estimated at 28.3 km², with males travelling up to 12.2 km from the nest site (Schipper, 1973, 1977). This was compared to sympatric hen harrier and marsh harrier populations, where males rarely travelled more than 2 - 4 km (Schipper, 1973, 1977). In Vendée, France, smaller hunting ranges of 600 to 700 ha per pair have been recorded (range 400 - 1200 ha) and foraging distances of 1 - 5 km, exceptionally up to 7 km. In Madrid, Spain, males have been observed hunting up to 7 km from the nest (Arroyo, 1995). In Cornwall, hunting ranges within a radius of up to about 8 km from the nesting territory have been recorded, with an inner 'restricted zone' of 300 - 400 m around the nest (Cramp and Simmons, 1980). Elsewhere in Britain, sightings of foraging males at maximum distances of up to 10 - 12 km from the nest have been recorded in Cambridgeshire and Norfolk (B. Imagen, pers. comm. in Clarke, 1996) and males regularly forage at distances up to 15 km from the nest in Wiltshire (P. Castle, pers. comm.). Although foraging flights themselves are generally low, birds often soar to considerable heights before gliding back to the nest (I. Carter, pers. comm.).

The female is more likely to hunt in or near the nesting territory (Cramp and Simmons, 1980), within a radius of a few kilometres, moving further away only occasionally, although more so at the end of the nesting period (female hunting area in the Netherlands of 1 km² and 1.5 - 5 km² recorded by Schipper, 1973).

Roosting

Males may roost up to 6 - 8 km from the nesting territory even when rearing young, and males and non-breeders may frequent a communal roost (Cramp and Simmons, 1980). A few Montagu's harrier often join marsh harrier autumn roosts, which have been recorded in Lincolnshire, Cambridgeshire, Norfolk and Kent (Clarke, 1996). Communal roosts of up to 20 males occur in Wiltshire, and can be approached from considerable height (> 100 m, P. Castle, pers. comm.).

Collision Risk

Montagu's harrier usually fly low over the ground, and hunting is mainly by flying fixed routes at low heights. However, they may fly to considerable heights over breeding grounds or on migration (Cramp and Simmons, 1980). During the breeding season, both sexes, but particularly males, may perform aerial displays such as solo high circling, mutual high circling and flight play, flight drifting and sky dancing (Cramp and Simmons, 1980). During solo high circling the bird rises by soaring to heights of 300 - 600 m and occasionally higher (Cramp and Simmons, 1980). Mutual high circling is similar with birds soaring together in wide spirals, and flight play can occur for prolonged periods (Cramp and Simmons, 1980). The sky dance resembles that of the hen harrier, including somersaults, loop the loops and plunging. The bird starts sky dancing by soaring quickly to 100 - 1000 m, followed by interspersed periods of level flight and plunging (Cramp and Simmons, 1980). Cramp and Simmons (1980) state that the display is mainly over the nesting territory, but occasionally

may start up to 3 km from the territory. Observations in Wiltshire suggest that sky dancing can occur over wide areas, for example up to 4 km from the nest (P. Castle, pers. comm.).

Birds can return to their nesting territories singly or in pairs (Cramp and Simmons, 1980). Lone males spend much of the day in flight drifting patrol or high circling over the territory, and may commence sky dancing even if no female is present. Mutual high circling and flight play over the nesting territory is most common immediately before and after pair formation until egg laying, but is also performed later in the season after food passes, even when young are present (Cramp and Simmons, 1980). Solo high circling may also occur late into cycle at times. Sky dancing by the male largely ceases after the first egg has hatched (Cramp and Simmons, 1980).

Disturbance

The species avoids densely settled and intensively disturbed areas, and is highly susceptible to interference and persecution (Cramp and Simmons, 1980). Four percent of known failures between 1900 and 1983 were due to accidental disturbance by humans, with most failures (46 %) being due to egg collectors, and 9 % due to persecution (Underhill-Day, 1990).

Sensitivity Criteria

Nest sites submitted to the RBBP during the 10-year period spanning 1997 - 2006 were plotted. Collision risk is probably most likely during display flights, although high flying can occur throughout the foraging range. However, it was not thought practical to include buffers of foraging ranges, and it is considered likely there will be a higher density of high flights near to the nest. Initially, a two-tier approach was planned, with the area within 1 km of breeding locations being classified as 'high sensitivity' and the additional area within 3 km classified as 'medium sensitivity', on the basis of estimated likelihood of aerial displays. However, this was rejected in favour of a 3 km 'high sensitivity buffer' on the map in order to mask nesting locations of breeding Montagu's harrier. It may be appropriate for Environmental Impact Assessment to assess habitat use by Montagu's harrier within 10 - 15 km of nest sites, subject to local consultation with NE/RSPB.

Acknowledgements

These sensitivity criteria were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thank you to Paul Castle, Jeff Knott, Mike Shurmer, Mark Thomas and Colin Wilkinson for comments on an earlier draft of this review.

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6. Black grouse

Introduction

Black grouse *Tetrao tetrix* are distributed throughout continental Europe and northern Siberia (Cramp and Simmons, 1980). Black grouse populations declined in both numbers and range in most parts of western and central Europe during the last century as a result of habitat loss, degradation and fragmentation, caused by changes in land use, particularly agricultural intensification (Cramp and Simmons, 1980, Baines, 1994). In the UK, black grouse have declined over the last 150 years, with a decline in numbers of black grouse shot in Scotland and northern England of around 90 % since 1900 (shooting bag counts reflecting population densities; Baines and Hudson, 1995), and a simultaneous severe range contraction. Black grouse were previously common throughout the southern counties of England, but are now largely confined to Scotland and north east England, with a small population in Wales (Gibbons *et al.*, 1993, Baines and Hudson, 1995). This range contraction has continued in recent decades, with black grouse being found in 28 % fewer 10 km squares between the 1968-71 and 1988 - 91 breeding bird atlases (Sharrock, 1976, Gibbons *et al.*, 1993). Range contraction is most severe from the south and west of the British range (Baines and Hudson, 1995).

The second national survey of black grouse took place in 2005, and gave a population estimate of 5078 lekking males in Britain; a decline of 22 % since the first national survey in 1995/6 (Hancock *et al.*, 1999, Sim *et al.*, 2008), and 80 % since the previous population estimate of 25 000 in 1990 (Baines and Hudson, 1995). Concern over the decline in range and numbers of the British black grouse population has led to several conservation initiatives (Baines *et al.*, 2000, Calladine, 2002, Warren and Baines, 2004). The English population of black grouse has become fragmented into just two subpopulations, in north Northumberland and in the North Pennines, although recent lek counts in England indicate increases in numbers and range between 1998 and 2006, with the 2006 population being estimated at 1029 males (Warren and Baines, 2008). Thus, instigation of recovery programmes in England have demonstrated that management can aid population recovery.

The black grouse is an Annex II/2 species (EC, 1979) and is on the red list of conservation concern in the UK (Gregory *et al.*, 2002).

Monitoring

Surveys of black grouse are usually based on counts of males at lek sites (Baines, 1996, Anon., 2003). National surveys were conducted in 1995 (Hancock *et al.*, 1999) and 2005 (Sim *et al.*, 2008). The 2005 national survey was based on a stratified random sample of 5 km squares, constituting approximately 8 % of the species' range in northern England and Scotland (Sim *et al.*, 2008), however the Game and Wildlife Conservation Trust (formerly Game Conservancy Trust) conducted a comprehensive survey of the population in the north of England in 2006 (Warren and Baines, 2008).

Breeding system

Black grouse males are polygamous and play no part in rearing young (Cayford, 1993). They display communally and mate on traditional arenas called leks, where each male defends a small area (Cayford, 1993). Males attend leks throughout the year, but the main period of activity is between March and June with another peak in late September and October in some areas (Koivisto, 1965, Hjorth, 1968). Females visit leks from April onwards (N. Picozzi in Cayford, 1993). Several females nest in the vicinity of the lek, and clutches of 6 - 11 are laid between April and early June, with incubation lasting 25 - 27 days (Cramp and Simmons, 1980). Young are precocial and nidifugous, and largely feed themselves, but are brooded by the female in rain or at night for the first 10 days (Cramp and Simmons, 1980). They are

capable of precocious flight at 10 - 14 days and become independent at around three months (Cramp and Simmons, 1980). There is usually just one brood, and this is led away from the nest within 24 hours (Cayford, 1993). The brood travels successively further from the nest in the following 2 - 3 weeks before settling into relatively small home ranges (30 - 60 ha: Wegge *et al.*, 1982; less than 50 ha: Starling, 1990; 10 - 30 ha: Warren and Baines, 2004).

Home ranges and site fidelity

Site fidelity

Dispersal in black grouse is confined to first-year hens, with adults of both sexes and first-year cocks showing high site fidelity (Caizergues and Ellison, 2002, Warren and Baines, 2002). A study of natal dispersal in the North Pennines, found that all juvenile females dispersed, on average 9.3 km (n = 8), so remaining within the study area, whilst all cocks (n = 11) remained within 1 km of their natal site (Warren and Baines, 2002). Birds of both sexes sometimes occupy separate winter and summer ranges, and generally remain faithful to the ranges they establish in their first winter and summer in subsequent years (Willebrand, 1988, Caizergues and Ellison, 2002). In the French Alps, ten birds (six males and four females) captured as juveniles, and two males and one female captured as yearlings, were monitored during at least two consecutive summers by radio-tracking (Caizergues and Ellison, 2002). The summer ranges of 92 % (12/13) of two-year old birds overlapped with the summer ranges occupied as yearlings, with the exception being a female who moved her nest site by about 1 km. Ninety one percent of males marked as juveniles were still in the study area the next summer. Furthermore, two one-year-old cocks occupied about the same summer range that they had established as yearlings (Caizergues and Ellison, 2002). Solitary display sites may be used for just one or two years compared to the regular use of traditional leks (Cayford, 1993).

Densities

The largest lek recorded during the 1995/6 national survey had 17 males. Densities varied from 0 to 2.4 displaying males per km² of apparently suitable habitat (Hancock *et al.*, 1999). In a survey of 69 leks in Perthshire, the numbers of males at a lek ranged from 2 - 30 (median and mode = 7; Robinson *et al.*, 1993). Thirteen percent of leks had 15 or more cocks and 25 single displaying males were found. Densities ranged from 0.8 - 2.3 males per km², with a mean of 1.6 males per km². This mean is similar to estimates from other British studies, and is within the range of densities found by Baines (1992) but is slightly higher than that found by Picozzi (unpublished) for parts of Deeside (Robinson *et al.*, 1993). Higher densities have also been reported by studies in Sweden, the Alps and Estonia, but these studies have either used smaller areas within which birds may be concentrated, or only favourable habitats have been searched.

Home ranges

Both sexes may occupy separate winter and summer ranges (Willebrand, 1988, Caizergues and Ellison, 2002), however, males and females both remain in the same general area throughout the year, rarely moving more than a few kilometres from the lek they attend in spring. For example, Warren and Baines (2004) found that in good, continuous habitat, leks are approximately 2 km apart, with most birds attending the leks thus being found within 1 km of the lek. Willebrand (1988) found that females usually nest within a 2 km radius of the lek. A study in north east Scotland found that 50 % of females nested within 1 km of the lek they were caught on, the maximum distance recorded being 3.9 km (Picozzi, 1986). The UK Biodiversity Action Plan for black grouse states that most females will nest within 1.5 km of the lek (Anon., 2003).

Warren and Baines (2004) suggest that suitable habitat needs to be available within an area of 200-500 ha around a lek (equivalent to the areas of circles with radii of 0.8 - 1.3 km), whilst Cayford (1993) suggested that an area of 500 - 700 ha may be enough to sustain one lek (equivalent to the area of circles with radii of 1.3 km and 1.5 km).

Estimated home ranges for black grouse are usually equivalent to the area of circles with radii of around 1 - 1.5 km for males and 0.5 km for females (Johnstone, 1969, Robel, 1969, Picozzi, 1986, Cayford, 1990, Cayford, 1993). Johnstone (1969) found that home ranges of lekking males in north east Scotland were about 500 ha (equivalent to the area of a circle of radius 1.3 km). Robel (1969) reported home ranges of males in Scotland of 303 - 689 ha with a mean of 458 +/- 126 S. D. (equivalent to the area of a circle with radius 1.2 km). Picozzi (1986) obtained range sizes of 355 ha +/- 178 S. D. for males (equivalent to the area a circle of radius 1.1 km), 86 ha +/- 37 S. D. for moorland females (equivalent to the area of a circle of radius 0.5 km) and 45 ha +/- 28 S. D. for woodland females (equivalent to the area a circle of radius 0.4 km), in Scotland. In coniferous plantations in Wales, home ranges of radio-tagged males averaged 325 ha +/- 71 S. D. (equivalent to the area of a circle of radius 1 km, Cayford, 1990), whilst mean home range of two females was 50.2 ha (equivalent to the area of a circle of radius 0.4 km, Cayford and Hope Jones, 1989). Home ranges of males attending the same lek overlapped considerably and were discrete from males attending adjacent leks (Cayford, 1993).

The UK Biodiversity Action Plan suggests that most females nest within 1.5 km of the lek, and that management for black grouse should be targeted at this area (Anon., 2003), Rural Development Contract packages for black grouse under the Scottish Rural Development Programme (SRDP) are also targeted at the area within 1.5 km of important lek sites, and this is the distance used to target management for the Wales recovery project. Provisional safe working distances of 300 - 1000 m from nests and leks are recommended for forestry workers in Britain (Currie and Elliot, 1997).

Collision Risk

Black grouse are particularly susceptible to collisions, for example with deer fences (Catt *et al.*, 1994, Baines and Summers, 1997, Moss *et al.*, 2000), elevated cables associated with ski tows (Miquet, 1990), power lines (Bevanger, 1995, MacLennan, 2000) and stock fences (Warren and Baines, 2002). Three of 22 radio-tagged black grouse for which the cause of mortality was known were attributed to stock fences or overhead power lines in the North Pennines (Warren and Baines, 2002). For capercaillie at least, any additional mortality associated with collision is likely to affect the UK population size (Moss *et al.*, 2000), and an influence on some local populations of black grouse is suspected (Baines *et al.*, 2000). Bevanger (1995) used counts of fatalities from a boreal site in central Norway and applied correction factors to calculate an average collision rate per kilometre of power line. It was estimated that about 56 % of the 95 000 km Norwegian high-tension power line system crossed black grouse habitat, and that this would result in approximately 26 000 black grouse mortalities per year, which is about half of the number attributed to hunting. Marking of deer fences with orange barrier netting can be highly successful in reducing black grouse collisions (91 % reduction, Baines and Andrew, 2003, although note, many other marking methods are frequently used which are as yet untested) and marking of power lines to reduce black grouse mortality has been suggested (MacLennan, 2000).

Hjorth (1970) states that display flights usually involve flights at less than 15 m, but Koivisto (1965) suggested they may sometimes reach heights of up to 30 m. More recently, Wright (2007) monitored flight heights at four lek sites in Scotland for 144 hours, and found that

mean flight height was 3 m (standard deviation = 2.3, n = 49), with no flights over 15 m being recorded. This suggests that black grouse are not likely to be at risk of collision with turbine blades (20 m being the lowest 'at risk' flight height for species in the UK suggested by SNH, 2005). There was a diurnal peak in flight activity, with most flights occurring at dawn, or from evening to dusk (Wright, 2007). It should be noted that all flights were monitored in May and June, and all were of males (as females will have been sitting on nests during this period). However, this may depend on habitat, for example I. Mackenzie (pers. comm.) has frequently seen birds flying above tree height when commuting between areas, and P. Warren (pers. comm.) suggests that birds may be at risk when approaching leks at dawn and dusk when heights of up to 50 m can be observed.

Two collision fatalities have been reported at a wind farm site in Argyll, Scotland (Y. Boles, pers. comm.) and two in Austria (T Dürr in Hötter *et al.*, 2006). These have all been with the wind farm towers, rather than turbines (Y. Boles, pers. comm., H. Zeiler, pers. comm.). It should be noted that the review by Hötter *et al.* (2006) is the result of collation of records from a variety of sources, as opposed to controlled corpse searches at wind farms. Proper quantification of collision rates at wind farms would need regular searches and an assessment of search efficiency and scavenger removal (Baines and Summers, 1997, Langston and Pullan, 2003, Smallwood, 2007). Black grouse also attend leks before dawn, which may increase the risk of collision.

Disturbance

Disturbance of lekking birds has been identified as a problem at some isolated sites (Anon., 2003). Protection of important or isolated lek sites from human disturbance is one of the objectives of the Species Action Plan (Anon., 2003), and black grouse is one of just two species for which access restrictions have been implemented under the provisions for recreational access of the Countryside and Rights of Way Act 2000 (Bathe, 2007). However, there are few studies of disturbance at lek sites. In the Alps, disturbance appeared to lead to shifts in winter-feeding and display areas away from intensively used skiing areas to less disturbed sites with concurrent declines in numbers (Miquet, 1986, Zeitler, 2000). Flushing distances occurred in response to skiers and snow ploughs at < 10 – 30 m if birds were in cover, but > 30 – 100 m when birds were in the open, and new installations of generators were avoided by 150 m (Zeitler, 2000). Human disturbance is also cited as a major factor causing black grouse to fly into overhead cables (Miquet, 1990). Numbers of black grouse were reduced by two-thirds within 250 m of roads, with some depression in numbers up to 500 m (Raty, 1979). Baltic (2005) found that experimentally disturbed grouse took significantly longer to feed. A recent study in which black grouse were subjected to different levels of disturbance ranging from high (twice weekly) to low (no disturbance) found no impact on black grouse breeding success, although birds that were disturbed more regularly flushed at greater distances (Baines and Richardson, 2007). However, it is possible the levels used may not have been high enough to cause an effect (Baines and Richardson, 2007). Flushing distances were 71 m in winter, 80 m in spring, 22 m in summer and 27 m in autumn (Baines and Richardson, 2007). Birds with greater prior experience of disturbance responded at greater distances, averaging 55 m compared to 34 m for birds with less experience of disturbance, implying a negative effect of experience of disturbance, as opposed to habituation (Baines and Richardson, 2007). Active response to dogs varies but is on average 2 m for incubating birds, and 39 m for birds with chicks (Storaas *et al.*, 1999, cited in Baines and Richardson, 2007). Observations of leks from hides can occur at 10 – 40 m without behavioural modification (Rintamanki *et al.*, 1995, Karvonen *et al.*, 2000).

Currie and Elliot (1997) recommended a disturbance free zone for forestry workers of 300 – 1000 m for black grouse. Ruddock and Whitfield's (2007) survey of expert opinion suggested that 'static disturbance' (i.e. the 'alert distance' at which a bird changes its behaviour but without taking flight) in response to a human would occur at distances greater than 100 – 150 m for breeding females, and at distances over 500 – 750 m for leks. Active disturbance distances (i.e. 'flight initiation distances') of 10 – 50 m for breeding females and 300 – 500 m for leks were suggested.

Sensitivity criteria

Although not an Annex I species, the black grouse's rapid rate of population decline in the UK (Sim *et al.*, 2008) and low dispersive ability (Caizergues and Ellison, 2002, Warren and Baines, 2002), which makes them poor at recolonising their range, means that potential negative effects of wind farms may be of concern.

Estimates of ranging distances from the lek by males and females vary slightly, but are usually between 1 and 2 km, and management for black grouse is targeted at the area within 1.5 km of the lek. Thus, it was decided to buffer black grouse leks by a distance of 1.5 km, the distance encompassing the range of most birds attending a lek and most nests around an associated lek, and classify the area within this buffer as 'medium sensitivity', on account of the black grouse's non-Annex I (i.e. national, rather than international) conservation status.

Acknowledgments

These sensitivity criteria are based on a previous version written for a Scottish sensitivity map (Bright *et al.*, 2006, 2008), updated in light of new information and to place them in the context of English population sizes and trends. They were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thanks to John Barrett, Ian Court, Murray Grant, James Pearce-Higgins, Fiona Hunter, Martin Kerby, Jonathan Morley and Phil Warren who provided helpful comments on an earlier draft of these sensitivity criteria.

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7. Stone-curlew

Introduction

The stone-curlew *Burhinus oedicnemus* has a breeding range extending from North Africa and Iberia, through southern Europe to central Asia, India and south-east Asia (Cramp and Simmons, 1983). Northern populations are migratory, whereas southern tropical and sub-tropical populations are sedentary (Stroud *et al.*, 2001). Approximately 95 % of the European population occurs in the Iberian Peninsula, France and Russia (Hagemeijer and Blair, 1997). The British population of stone-curlew is confined to central southern England and East Anglia, with strongholds on Salisbury Plain, and in the Brecklands of Norfolk and Suffolk, and small populations also occurring in Berkshire, north Norfolk, south Cambridgeshire and the Suffolk coast (Stroud *et al.*, 2001).

Stone-curlews were formerly widespread in Britain, but underwent a severe decline in numbers and range from the 1930s, due largely to loss of heathland and semi-natural grassland, their preferred breeding habitats (Green and Taylor, 1995), numbering just 167 - 169 pairs by 1991 (Green, 1995). The population reached a low point of 150 pairs in the early 1980s, restricted to just two areas: the Breckland of Norfolk and Suffolk, and downland in Wiltshire and Hampshire (Wessex) (Green and Taylor, 1995).

Stone-curlews prefer to nest on semi-natural dry grasslands and heaths, but most pairs now nest at lower densities on spring-sown arable farmland (Green and Griffiths, 1994, Green *et al.*, 2000). A study in the Brecklands concluded that a decline in the number of pairs nesting on heathland was an important driver of the population decline observed between 1968 and 1991 (Green and Griffiths, 1994). Stone-curlew require short vegetation and bare ground for breeding (Green *et al.*, 2000). Long-term declines in the UK are likely to be due to habitat loss, with recent declines being driven by habitat becoming unsuitable due to reduced grazing by rabbits and livestock resulting in increased sward heights (Green and Griffiths, 1994, Green and Taylor, 1995). On arable land, the switch from spring to autumn sowing has reduced the area of suitable nesting habitat, with autumn sown crops often being too dense for breeding attempts, and the species is also vulnerable to losses from agricultural operations (Aebischer *et al.*, 1990). Declines also occurred throughout the rest of Europe during the 20th century, particularly after the Second World War (Hagemeijer and Blair, 1997).

Stone-curlew recovery work began in the late-1980s, with measures such as protecting stone-curlew nests on arable land from agricultural operations, and providing more suitable breeding habitat with short, sparse vegetation on sandy, stony soils, partly by establishment of stone-curlew plots on arable land or semi-natural grassland (Taylor *et al.*, 2007). This conservation action (now a joint RSPB/Natural England project) has led to partial population recovery, with the breeding population reaching *ca* 350 pairs in 2007 (R. Wynde, pers. comm.). However, the population still remains small and vulnerable (Gibbons *et al.*, 1996, Gregory *et al.*, 2002).

The stone-curlew is an Annex I species (EC, 1979) and is on the red list of conservation concern (Gregory *et al.*, 2002). There are three Special Protection Areas for stone-curlew (Breckland, Porton Down and Salisbury Plain, Stroud *et al.*, 2001). At the time of the SPA review, these were estimated to cover 98 % of the British breeding population (Stroud *et al.*, 2001), however, this was using a population estimate of 184 pairs, thus current coverage is likely to be significantly lower.

Monitoring

Comprehensive monitoring of the stone-curlew population is conducted annually as part of 'Action for Birds in England', a conservation partnership between Natural England and the RSPB. Two project teams, one in Wessex and one in the Brecklands, survey and monitor the population throughout the breeding season, and work with farmers and landowners to improve nesting success and establish safe and suitable breeding habitat. Chicks are colour-ringed to monitor annual productivity and adult survival through colour ring re-sighting.

Breeding ecology

The stone-curlew is a nocturnal bird that nests and forages on sparsely vegetated ground (Green and Griffiths, 1994). Birds arrive at breeding grounds in England from early March, sometimes paired at the time of arrival (Cramp and Simmons, 1983). Small numbers of birds may overwinter. Stone-curlew are ground-nesting, the nest consisting of a scrape (Cramp and Simmons, 1983). Most clutches are laid from about the 10th of April to the 10th of May (Green *et al.*, 2000). A clutch of one or two eggs is laid (mean = 1.9, n = 74, Glue and Morgan, 1974). Incubation is by both sexes and lasts 24 - 26 days (Cramp and Simmons, 1983). Young are precocial and nidifugous, and are fed by both parents (Cramp and Simmons, 1983). The fledging period is 36 to 42 days, and during this time chicks remain largely within their parent's original territories (Cramp and Simmons, 1983). There is usually one clutch, but sometimes two are laid, replacements may be laid after the loss of eggs or small young, and may be on or near the same nest site (Cramp and Simmons, 1983). Stone-curlews can have a long breeding season with clutches being laid into August (P. Sheldrake, pers. comm.). It is not uncommon for a pair to lay three or four clutches in a season after failure (P. Sheldrake, pers. comm.)

Home range sizes and site fidelity

Site fidelity

Cramp and Simmons (1983) suggest that birds are highly site faithful, with birds returning to traditional sites, even after the habitat has been altered, for example by tree planting. Likewise, Westwood (1983) suggests that the stone-curlew shows great site faithfulness, and when not disturbed continues to breed in the same areas.

Home range sizes and foraging behaviour

Cramp and Simmons (1983) state that the nature of territoriality in stone-curlew is unclear, with pairs using feeding areas beyond the general nesting territories, which may be shared with other birds. Stone-curlew are cited as flying up to 1.2 km to arable land from heath or breck to feed (Morgan in Cramp and Simmons, 1983). Radio-tracking of stone-curlew breeding on semi-natural grassland and spring sown arable land in southern England found that they were most active at night and at dawn and dusk (Green *et al.*, 2000). Foraging occurred around the nest site or locations of chicks, but also at a number of other areas up to about 3 km from the nest (Green *et al.*, 2000). The radius around the nest containing 95 % of active locations of radio-tagged birds varied depending on stage in the breeding cycle, but most activity occurred within 1 km of the nest or chicks (75 %, 83 % and 95 % of active locations during the pre-laying, incubation and chick-rearing periods, respectively, Green *et al.*, 2000). Chicks rarely moved more than a few hundred metres from the nest (Green *et al.*, 2000). Home ranges of individuals were fragmented and on average consisted of about 30 ha of short semi-natural grassland, short improved pasture and spring sown crops for foraging (Green *et al.*, 2000). Densities can vary widely and are likely to be influenced by habitat quality (Cramp and Simmons, 1983).

Roosting behaviour

Following breeding, stone-curlew form 'roosting' flocks at a number of areas within the current breeding range from July onwards until migration (P. Sheldrake, pers. comm.). Habitat similar to that preferred for nesting is chosen. Birds remain largely inactive during daylight hours, however, movement of individuals between roosts has been shown to occur through roost monitoring and resighting of colour-ringed birds (P. Sheldrake, pers. comm.).

Collision risk

Information on flight heights in stone-curlew is lacking. Stone-curlews are adapted to crepuscular and nocturnal activity, having large eyes and good night vision, but it is not really known whether unfamiliar structures such as wind turbines will pose a significant collision risk. Green (unpubl.) created models to estimate the potential risk of collision with wind turbines for breeding stone-curlews, and concluded that collision risk could be a problem where turbines were placed within the foraging range of breeding birds. In the absence of turbine collisions, average annual mortality rate is about 0.2 (Green *et al.*, 1997), and Green (unpubl.) suggested that as population trends of species with such low adult mortality can be particularly sensitive to increases in mortality, even a low level of additional mortality would be undesirable for a species with such a small population size in England.

Disturbance

Responses to various forms of disturbance were observed in a population of stone-curlews in Wiltshire and Hampshire (Taylor *et al.*, 2007). These suggested that stone-curlews appeared to be more sensitive to disturbance than other species of wader, showing an active response (i.e. running or flying) at large distances, for example over 500 m from a person with a dog. Response distances to walkers without dogs were lower, and those to vehicles lower still. These observations were used to develop a conservation tool, the 'Stone-Curlew Access Response Evaluator' (SCARE), which predicts the effects of different disturbance types, distributions and frequencies on stone-curlew populations, and relates the number of active responses per hour during settlement to the probability of a plot being occupied for nesting (Taylor *et al.*, 2007).

Green *et al.* (2000) found that density of stone-curlew on arable fields was low next to major roads, and reached a maximum at 3.6 km from the nearest road, where it was 6.3 times higher. There was no tendency for stone-curlews to be more tolerant of nesting near to roads where other suitable habitat was scarce, suggesting that this disturbance may limit stone-curlew population size, and models estimated that the observed population was 47 % lower than might be expected if there was no effect of roads on stone-curlew (Green *et al.*, 2000). More recent analysis of stone-curlew nests in relation to proximity to housing indicates avoidance of human settlements (R. Green, pers. comm.)

Sensitivity Criteria

Nest sites used for the last five years (2003 - 2007) were plotted. The time span of data was chosen following consultation with species' experts, to adequately represent breeding locations for this species, which is recovering from a very low population base. These locations were buffered by 1 km, and this area classified as 'high sensitivity'. This is on the basis that Green (unpubl.) suggests that risk of collision may be a problem, and the majority of foraging activity occurs within 1 km of the nest (Green *et al.*, 2000).

When individual wind farm proposals are being considered, it is recommended that any wind farm proposals within 3 km of a stone-curlew nest or nest locations be assessed for impacts on breeding stone-curlews, given the very small population and range of foraging by

breeding birds. Such assessments need to include any new roads and buildings associated with the wind farm.

Acknowledgments

These sensitivity criteria were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thanks to Tim Cowan, Phil Sheldrake and Robin Wynne who provided helpful comments on an earlier draft of these sensitivity criteria.

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8. Nightjar

Introduction

The nightjar *Caprimulgus europaeus* is a migratory species, breeding from North Africa and western Europe, widely across temperate regions of Eurasia as far as central Asia and western China, and over wintering in Africa south of the Sahara (Stroud *et al.*, 2001). The British nightjar population declined in numbers and range throughout most of the 20th century (Morris *et al.*, 1994). The species' range has also contracted from the north and west in Britain, with the population now being concentrated in areas of lowland heath along the south coast and increasing in recently restocked areas within coniferous plantations in the south and east (Conway *et al.*, 2007). Large declines have also occurred in most other European countries (BirdLife International, 2004).

For a full review of population trends for the UK nightjar population, and possible causes, see Langston *et al.* (2007a). The population was estimated at between 3000 and 6000 churring males in 1972 (Sharrock, 1976), but by the 1981 national survey the population estimate was just 2100 churring males (Gribble, 1983). The population decline up to this point is likely to be largely due to reduction in area, and fragmentation, of lowland heath habitat, formerly from afforestation for conifer plantations and conversion to arable and then from scrub invasion due to lack of grazing and better control over heath fires, and building development (Morris *et al.*, 1994).

However, the population has been increasing in recent decades, with an increase of about 75 % by the second national survey in 1992, which gave a population estimate of about 3400 males. However, there was no corresponding increase in range by the 1992 national survey (Morris *et al.*, 1994). This partial population recovery is likely to have been caused largely by changes in forest structure (Morris *et al.*, 1994). The expansion of commercial forestry after 1945 resulted in loss of breeding habitat for nightjar once the forests had matured, and become too dense for nightjar to nest in, which occurs after 10 to 20 years (Morris *et al.*, 1994). Many of these forests were harvested during the 1980s, providing nesting habitat for nightjars in the form of clearfells and restocks, and additional woodland clearings were created by the gales of 1987 and 1990 (Morris *et al.*, 1994). This resulted in an increased proportion of records in conifer plantations between 1981 and 1992 (Morris *et al.*, 1994).

There was a further increase of about 36 % between the 1992 and 2004 national surveys, and the current British population estimate is 4606 males (Conway *et al.*, 2007). There was also a small increase in range, with the number of occupied 10 km squares increasing by 2.6 % (Conway *et al.*, 2007). However, population declines and range contractions still occurred in the north west of Britain, including north Wales, north west England and Scotland (Conway *et al.*, 2007). Fifty seven percent of the nightjars surveyed in 2004 were associated with forest plantations and 59 % with heathland (the two not being mutually exclusive, Conway *et al.*, 2007). The continued overall increase was probably due to habitat protection, heathland restoration and management, and to forestry providing habitat in terms of rotational clearfell or young plantations in conifer forests (Conway *et al.*, 2007).

The nightjar is an Annex I species (EC, 1979) and is on the red list of conservation concern (Gregory *et al.*, 2002). There are ten Special Protection Areas for nightjar, and the SPA review estimated that these contain 53 % of the British breeding population (although this was based on the 1992 national survey, Stroud *et al.*, 2001).

Monitoring

National surveys have been conducted as part of the SCARABBS scheme in 1981 (Gribble, 1983), 1992 (Morris *et al.*, 2004) and 2004 (Conway *et al.*, 2007) and are based on numbers of churring males (Gilbert *et al.*, 1998).

Breeding Ecology

In East Anglia, the first males usually arrive in the second week of May and start churring (Berry, 1979), with males at Minsmere arriving between 10th May and 1st June and females between the 11th May and 16th June (Berry and Bibby, 1981). Males roost about 50 m (occasionally 100 m) from the incubating female and show remarkable fidelity to roost sites (Berry, 1979). In the evening, on leaving the roost area, a male nightjar will chur at various regular song posts in his territory and then either relieve the female at the nest or feed (Berry, 1979).

First eggs are laid between the 17th May and 28th June (Berry and Bibby, 1981). Second clutches were generally thought to be relatively uncommon (e.g. 24 % of territories at Minsmere in Suffolk; Berry and Bibby, 1981), but have been more regularly observed in recent years (e.g. G. Conway, pers. comm.). They are usually laid relatively close to the first clutch, with a mean distance of 124 m (range 70 - 160 m) recorded in Dorset (Cresswell, 1996). Young hatch after 17 to 21 days incubation and are probably dependent on their parents for about 30 days (Berry and Bibby, 1981).

Home ranges and site fidelity

Site fidelity

Birds are faithful to a nest site area, though not the precise nest site, from year to year (Berry, 1979).

Foraging ranges

Lengths of foraging flights recorded in different studies vary. Alexander and Cresswell (1990) radio-tracked nightjars in Dorset and found that mean distance traveled from the nest site to foraging area was 3.1 km +/- 1.2 km. The maximum distance recorded was 5.8 km, but most flights were between 2 km and 4 km. Nightjar in this study showed preference for foraging in deciduous or mixed woodlands and gardens/orchards, whereas conifer plantations and improved grassland/arable were little used despite their greater availability, and there were no records of visits to dry heath (Alexander and Cresswell, 1990).

Bowden and Green (1994) radio-tracked nightjars in Norfolk and Suffolk, and recorded shorter foraging distances, with most foraging occurring within 1 km of the territory centre, and no flights being over 2 km. Various other studies have found that nightjar forage mainly within their nesting territories (Lack, 1930, Koenig, 1952, Schlegel, 1967, Wichmann, 2004). Glutz von Blotzheim (1962) and Schlegel (1967) also described nightjars foraging several kilometres from suitable nesting areas, although this was prior to the laying period. The observed variation in length of foraging flights is likely to be related to habitat surrounding the breeding territory and to local food abundance. The population studied by Alexander and Cresswell (1990) had to travel 1.5 km to reach the forest edge, which could account for the relatively high foraging distances found by this study.

It is not known how much nightjars enter other bird's territories when hunting. In sites where prey is abundant, nightjars have been reported to hunt together in groups (Lack, 1932, Berry, 1979). Nightjars often use the same feeding sites for many nights in succession (Cresswell, 1996).

Collision Risk

It is not clear how susceptible to collision with wind turbines nightjar are. Risk of collision would occur mainly during display or migratory flights, and possibly during some foraging flights.

Foraging flights are generally low over, for example, bracken or heather, but along the forest edge, birds fly close to the top of the canopy (Cramp *et al.*, 1985). It is not clear whether, or how often, nightjars fly at sufficient height to be at risk of collision with turbines (mean lowest sweep of rotor = 27 m, n = 84, calculated from SNH data of wind farms in Scotland and this lowest sweep height is likely to be getting higher as new, larger turbines are developed). There is little published evidence of flights over 27 m, although one case of an unpaired nightjar making a patrolling flight at 10 - 15 m above trees was cited by Cramp *et al.* (1985). Nightjars approach their prey not horizontally, but from below in steep flight (Schlegel, 1967, Lehtonen, 1969, Bühler, 1987). P. Palmer (pers. comm.) has observed high spiraling flights in nightjars and C. Rollie (pers. comm.) observed a flight to a height over 30 m; presumably these birds were chasing prey. S. Wotton (pers. comm.) has frequently seen birds flying above mature conifer height (over 20 m+) during display flights, flying from one churring spot to another, and on foraging flights.

However, a recent study for the Environmental Impact Assessment of a wind farm in Dorset using thermal imaging cameras, Vantage Point watches with image intensifiers and other observations recorded no flights over 20 m during 134 hours of data collection (138 nightjar records, Infinergy, 2008). Of 27 observations using a thermal imaging camera, about 89 % of flights were at heights of 10 m or less (Infinergy, 2008). Eleven breeding pairs of nightjar were present in the area and observations were made between late July and September 2007 (Infinergy, 2008). Further surveys to investigate flight heights earlier in the season, when display flights are more likely to occur, were conducted in May and June 2008 (R. Henderson, pers. comm.). These also found no flights of over 20 m (R. Henderson, pers. comm.). The study occurred in an area of open habitat. Nightjar flights are generally low over the vegetation, so collision risk is most likely to occur when wind turbines are situated in areas adjacent to forestry, as flight heights over trees will be higher.

Nightjars are quite manoeuvrable in flight (Cresswell, 1996), and, being crepuscular, have enhanced night vision, both of which are likely to reduce their risk of collision.

Disturbance

Nightjars are sensitive to disturbance at breeding sites (Murison, 2002, Liley and Clarke, 2003, Woodfield and Langston, 2004, Langston *et al.*, 2007b; see Langston *et al.*, 2007c for review). Liley and Clarke (2003) studied 36 heathland sites in Dorset and found that nightjar densities had a strong negative relationship with surrogates for disturbance (amount of developed land within 250 m, 500 m and 750 m of the heath, and number of houses surrounding the heath). Murison (2002) found that urban heaths supported fewer territories, and that path proximity and the amount of footpath surrounding nests were both significant predictors of nightjar breeding success. Path effects correlated strongly with nest failure up to 225 m from the path edge, but significant effects of paths were found up to 500 m, which was the outer range investigated (Murison, 2002). Murison (2002) suggested that predation may be the mechanism for these effects of disturbance on nightjar breeding success. Woodfield and Langston (2004) studied nightjars on four Dorset heaths with particularly high levels of access in order to investigate the possible mechanisms for these relationships. There were non-significant trends for nightjar nests to be in areas of fewer footpaths, further from main access

points, and surrounded by higher vegetation cover. There were relationships between hatching success and proximity to footpath. These were not statistically significant, but this may have been a result of lack of power. Unfortunately these studies do not allow the identification of a 'cut-off' distance at which disturbance is important, as indices of disturbance within different distances of breeding sites may be correlated.

Currie and Elliot (1997) proposed safe working distances for forestry workers of 50 – 250 m, although it is not clear on what this is based. As nightjar rely on cryptic plumage to escape detection, estimates of 'static disturbance distances' (i.e. the 'alert distance' at which a bird changes its behaviour but without taking flight) should be viewed with caution, as avoidance of movement is part of a nightjar's behaviour to escape detection (Ruddock and Whitfield, 2007). This will probably also lead to low 'active disturbance distances' (i.e. 'flight initiation distances'), with birds flushing when potential predators are at close range (Ruddock and Whitfield, 2007). Ruddock and Whitfield's (2007) survey of expert opinion suggested active disturbance was considered likely at a maximum upper limit of < 10 m during incubation and 50 – 100 m during chick rearing. Passive disturbance is likely to occur at greater distances, however (Ruddock and Whitfield, 2007).

Sensitivity criteria

The two potential risks posed to nightjar by wind farms are collision risk and disturbance displacement. There is little published information on flight heights in nightjar, although recent information collected in Dorset found that all flights were less than 20 m, suggesting that collision risk may not be a problem. The study site was an area of open habitat, higher flights would be most likely to occur over forestry. Thus, wind farms located adjacent to, or in clear-felled pockets within, forestry are most likely to pose a risk of collision to breeding nightjar, and this should be considered during Environmental Impact Assessment. Nightjar are susceptible to disturbance from human activity, with effects on breeding densities and breeding success being found (Murison, 2002, Liley and Clarke, 2003, Woodfield and Langston, 2004). There is no information on effects of wind farms on breeding nightjars, but disturbance displacement from the turbines themselves, or due to increased human activity or access at the site, may affect breeding success or the amount of habitat available.

As the English nightjar population is relatively numerous and increasing, a less precautionary approach was taken here than for the Scottish sensitivity map (Bright *et al.*, 2006, 2008), where all 1-km squares containing breeding nightjar were mapped and buffered. For England, Special Protection Areas designated for breeding nightjar have been included on the map, as well as sites containing 1 % or more of the British breeding population in light of the 2004 national survey data (current SPA boundaries being based on earlier data). 'Sites' were classified as clusters of males, with overlapping 2 km buffers. This distance of 2 km was chosen as an estimate of foraging range, based on the fact that nightjars in Thetford Forest foraged up to 2 km from their nesting area, with most flights being less than 1 km (Bowden and Green, 1994). A range of other studies found that nightjars forage mainly within their nesting territories (Lack, 1930; Koenig, 1952; Schelgel, 1967; Wichmann, 2004). Although birds in Dorset foraged a mean distance of 3.1 km (+/- 1.2 km) from their nest site, birds at this site had to travel 1.5 km to reach the forest edge, which could account for the relatively high foraging distances (Alexander and Cresswell, 1990).

These were included as 'medium sensitivity' in order to highlight sites where wind farms have the potential to impact on nationally important populations of breeding nightjar. The 'medium sensitivity' rating is on the basis of the nightjar's relatively numerous and increasing population size. Environmental Impact Assessments should also establish the availability of

suitable foraging habitat within 2 km of SPAs designated for nightjar, and the associated collision risk.

Acknowledgments

These sensitivity criteria are based on a previous version written for a Scottish sensitivity map (Bright *et al.*, 2006, 2008), updated in light of new information and to place them in the context of English population sizes and trends. They were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thanks to Renny Henderson and Tim Melling who provided helpful comments on an earlier draft of these sensitivity criteria.

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9. Chough

Introduction

The red-billed chough (hereafter 'chough') *Pyrrhocorax pyrrhocorax* has a discontinuous world range, with populations found across Europe, Asia and in North Africa (Cramp and Perrins, 1994). Over 75 % of the European breeding population occurs in Spain, Greece and Italy (Tucker and Heath, 1994). Choughs are largely resident on their breeding areas, and there is little evidence of interchange between breeding populations; previous contractions and declines have eventually resulted in extinction, e.g. from England in the 1970s and Austria in the 1950s (Stillman *et al.*, 1998). The European breeding population underwent a large decline between 1970 and 1990, and although populations were stable across much of the chough's European range between 1990 and 2000, key populations in Spain and Turkey declined, and the population underwent a moderate decrease overall (BirdLife International, 2004). These declines are almost entirely attributed to the loss of traditional low intensity livestock farming (Bignal and Curtis, 1988, Tucker and Heath, 1994).

In the UK and Ireland, the chough is largely restricted to areas of low-input agricultural land. The chough was historically much more widespread in Britain, but its British range has contracted over the last 300 years and the breeding population is now restricted to western coasts and islands (Gibbons *et al.*, 1993). The population has also been decreasing in numbers over this time (Warnes, 1983). Persecution led to declines during much of the early part of the 20th century, and population declines continued due to agricultural intensification (Bignal *et al.*, 1997). However, the UK breeding population has increased in recent decades, with 399 probable and confirmed breeding pairs being located in 2002 (Johnstone *et al.*, 2007). Populations on the Isle of Man and in Wales increased, whilst numbers have fluctuated in Scotland (Johnstone *et al.*, 2007). Recent increases could be an artefact of changes in survey methodology (although this unlikely to be the case for increases since the 1992 national survey), but are more likely due to conservation management or changes in climate (Johnstone *et al.*, 2007). Wales held over 50 % of the breeding population in 2002, with over 99 % of choughs found occurring in Wales, the Isle of Man and on Islay or Colonsay in Scotland (Johnstone *et al.*, 2007).

Chough went extinct in England in the 1970s (Stillman *et al.*, 1998), having last bred successfully in Cornwall in 1947 (RSPB, 2008a). However, in 2002, a pair returned to Cornwall and bred successfully (Johnstone *et al.*, 2007), and this was followed by successful breeding of a second pair in 2006 (RSPB, 2008b).

The chough is an Annex I species (EC, 1979) and is on the amber list of conservation concern (Gregory *et al.*, 2002). The majority of birds are concentrated within a very small area, making the population vulnerable to threats such as changes in land use. Around 33 % of the British breeding population of chough are within nine SPAs (seven in Wales and two in Scotland), eight of these are also designated for non-breeding chough, and support an average of 241 birds during the non-breeding season, or 35 % of the non-breeding population (Stroud *et al.*, 2001).

Breeding Ecology

Most sub-adult choughs spend their first two years as part of a flock or flocks with other sub-adults (Bignal *et al.*, 1997). These flocks forage widely but not randomly, always focusing on some key areas of sand dune, machair, sand grassland or limestone grassland, and form communal roosts at night, typically on rock faces (Still *et al.*, 1987). Most choughs first breed successfully in their third year of life (Bignal *et al.*, 1997). Established breeders maintain contact with their nest sites and their home range foraging area throughout the year, roosting

at or close to their nest (Bignal *et al.*, 1997). Early in the year, choughs sometimes congregate to forage in small flocks, sharing their home ranges without territorial disputes (Bignal *et al.*, 1997).

Nest building begins in March and egg-laying in late March or early April (Bignal *et al.*, 1997). The chough uses a variety of nest sites, including ledges in caves, crevices, and both used and abandoned buildings (Finney and Jardine, 2003). Most nests are 10 - 30 m above the ground or sea (range 1 – 250 m, Cramp and Perrins, 1994). Each pair may have several potential nest sites, and in any year, nests may be rebuilt at more than one of these before they decide which one to use (Bignal *et al.*, 1997). With the onset of egg-laying, pairs become strongly territorial and will not tolerate adjacent pairs in their home ranges (Bignal *et al.*, 1997). A mean clutch size of 4.6 was found on Islay (S. D. = 1.1, n = 63), the most common clutch size in Scotland is five (Cramp and Perrins, 1994). While the female incubates, the male defends the home range, forages within it and feeds the female at or close to the nest (Bignal *et al.*, 1997). Both parents make regular foraging trips within their home range (Bignal *et al.*, 1997). Young leave the nest after six weeks (Bignal *et al.*, 1997). Second broods are extremely rare (Cramp and Perrins, 1994).

Home ranges and site fidelity

Natal philopatry

Re-sightings of colour-ringed birds have shown that male choughs tend to nest close to their natal site whilst female choughs disperse to breed, with natal dispersal distances generally being less than 4 km for males and more than 10 km for females (Bignal *et al.*, 1989). Post-fledging dispersal distances of over 50 km have been recorded (Wernham *et al.*, 2002).

Site fidelity

Choughs are very site faithful and once established at a nest site will continue to attempt to breed there despite repeated failures, for instance due to disturbance, predation or management changes (Scottish Chough Study Group, 1995).

Densities

Robertson *et al.* (1995) found a mean density of 1.3 chough km⁻² in County Donegal in Ireland; this is comparable to the Irish national average (Berrow *et al.*, 1993). In Britain, nests are usually several hundred metres or more apart where densities are relatively high, such as on the Isle of Man, whilst in Wales, nests were on average 1.4 km (n = 32) apart in good habitat, but 2.6 km (n = 27) apart in poorer habitat (Bullock *et al.*, 1983). The highest densities in Britain are probably on Islay with up to 0.33 nesting pairs per km² (Monaghan *et al.*, 1989), with densities being much lower in a mountainous area of North Wales (Holyoak, 1972).

Three quarters of 1-km squares visited in the 2002 national survey with possible/probable/confirmed breeding records contained only one site, indicating that choughs mainly nest at low densities in the UK and Isle of Man (Johnstone *et al.*, 2007). Densities were significantly higher in the Isle of Man than in Wales and Scotland, and in a few places on the Isle of Man and in Wales nest sites were clustered on favoured sea cliffs and quarries within 1-km squares (Johnstone *et al.*, 2007). This 'colonial' breeding has been reported previously for chough on the Isle of Man (Bullock *et al.*, 1983) and occurs more markedly elsewhere in Europe (e.g. Blanco *et al.*, 1998). However, the UK and Isle of Man chough population still consists mainly of solitary breeders nesting at low densities with exclusive home ranges (Johnstone *et al.*, 2007).

Home range sizes

Territory around the nest is defended, with pairs being observed driving conspecifics away when they intruded within several hundred metres of the nest sites (Holyoak, 1972). The radii of home ranges in County Donegal ranged from 1.2 km to 2 km (n = 5, Robertson *et al.*, 1995). The five breeding pairs observed showed a preference for foraging in machair and maritime turf, with rough grass also being important, and pairs avoiding heather and improved pasture. These foraging preferences were similar to those from previous work on habitat selection by choughs in Ireland (Bullock *et al.*, 1983).

Cramp and Perrins (1994) state that 'much but by no means all feeding is carried out within the home range away from the nesting territory'. A study of three pairs on Islay found that they had maximum foraging flight distances of 1.2 km, 0.8 km and 0.6 km respectively (Bignal *et al.*, 1996). A more recent study on Islay, conducted during the 1998 national survey, found choughs foraging a mean distance of 595 m (+/- 71 S. E., n = 38) from the nest (Cook *et al.*, 1999). Inland in north Wales, several pairs were regularly seen flying from nests to collect food 2 - 3 km away (Holyoak, 1972), although in Welsh coastal areas distances between nests and observed feeding places averaged 0.7 km (n = 58, Bullock *et al.*, 1983). Whitehead *et al.* (2006) studied 14 pairs of breeding chough in north Wales and found that the density of habitat use decreased asymptotically with distance from the nest, such that 95 % of foraging flights occurred within 1 km of the nest. A study in south west Ireland of birds from 12 sites in 2004 found that birds spent 95 % of their time within 1.3 km of the nest (Gray *et al.*, 2004).

Breeding birds often flock with conspecifics, breeders and non-breeders, flying high into the air and up to several kilometres from the nest site (Holyoak, 1972). Members of flocks often soar for short periods and frequently perform skilful acrobatics, commonly making earthward swoops from high up (Cramp and Perrins, 1994).

Disturbance

A study of radio-tagged alpine choughs *Pyrrhocorax graculus* in Italy found that human developments were avoided by foraging chough, and that use of a meadow was reduced in 1990 following extensive ski development (Rolando and Pattersen, 1993). However, other studies of alpine chough have found that they can be attracted to skiing developments due to increased food (Rolando *et al.*, 2003, Storch and Leidenberger, 2003, Laiolo, 2007). Cramp and Perrins (1994) state that chough are often less shy than other corvids in Britain, sometimes allowing approach to within 10 m. Detailed studies of the reaction of choughs to tourists in south Wales found a mean minimum flushing distance of 35 m (S. D. = 6, n = 101, Owen, 1989), whereas minimum distances as low as 9.5 - 11 m were found elsewhere in Britain (Cramp and Perrins, 1994). Chough have also been found to be tolerant of disturbance at active quarries in North Wales (I. Johnstone, pers. comm.).

Collision risk

No information was found regarding likelihood of collision in chough.

Sensitivity criteria

Although chough have not been shown to be at risk of collision, and their sensitivity to disturbance does not seem particularly high, the extremely small and localised nature of the English breeding population justifies a precautionary approach.

Studies of breeding chough in north Wales have found that most of their foraging flights (95%) are within 1 km of the nest (Whitehead *et al.*, 2006). Other studies have found mean foraging distances of 0.6 km (Cook *et al.*, 1999), and 0.7 km (Bullock *et al.*, 1983), or maximum distances for individual pairs of 1.2 km, 0.8 km and 0.6 km (Bignal *et al.*, 1996). A study in

south west Ireland found that birds spent 95 % of their time foraging within 1.3 km of the nest (Gray *et al.*, 2004). SNH guidelines for defining SPA boundaries use a distance of 1 km from the nest site, excluding only unsuitable habitat within this radius and including areas beyond it known to be used by foraging chough at the site in question.

Based on this, all nest sites used since chough first bred in England in 2002 were buffered by 1 km, and this area classified as 'high sensitivity'. Roost sites were also included in part because they may become nest sites in the future. These were buffered by 1 km and this area classified as 'high sensitivity'. This approach was considered appropriate for England, given the very low population size, a less precautionary approach may be required in Wales, where population sizes are much higher. Identification of areas holding nationally important populations using the SNH SPA approach may be appropriate for identifying areas of concern for wind farms.

Acknowledgments

These sensitivity criteria are based on a previous version written for a Scottish sensitivity map (Bright *et al.*, 2006, 2008), updated in light of new information and to place them in the context of English population sizes and trends. They were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thanks to Claire Mucklow who provided helpful comments on an earlier draft of these sensitivity criteria.

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10. Crane

Introduction

The common crane *Grus grus* (hereafter 'crane') is a widespread summer visitor to northern Europe, occurring more patchily further south as far as Spain and Romania (BirdLife International, 2004). There are four main wintering areas: the Iberian peninsula and North Africa; North East Africa (Nile river basin); the northern part of the Arabian peninsula and the Near East; and the Hindustani Peninsula and South East China (Leito *et al.*, 2006). Crane is classified as a Lower Risk (Least Concern) species under the revised IUCN Red List Categories (IUCN, 2008). The species has shown a contraction in breeding range since the Middle Ages (Mathews and Macdonald, 2001) and strong population declines in the past, but its total population is now probably stable or increasing (Meine and Archibald, 1996). In Europe, the species is classified as 'Depleted' in 2008 according to the BTO website. Habitat loss and degradation are still considered the principal threats to the species (Tucker and Heath, 1994, Meine and Archibald, 1996).

The species bred in East Anglia until about 1600, and perhaps in Ireland until the 14th century, before going extinct through habitat loss and over-harvesting (British Ornithologist's Union, 1971). Place-name, archaeological and documentary evidence suggests that the crane was formerly widespread in Britain (mostly England), and was present as a breeding bird, rather than just a winter visitor (Boisseau and Yalden, 1998). In recent decades, the crane has become a regular visitor to Britain once more, although initially this was largely as a vagrant (Mathews and Macdonald, 2001), and it was removed from the British Rarities List at the end of 1987 (British Birds, 1989). A breeding population has established in Norfolk since 1981, in the Upper Thurne Conservation Area, a network of fenland sites (Mathews and Macdonald, 2001). Although mainly a migratory species, most the Norfolk population is resident (Mathews and Macdonald, 2001). Breeding birds elsewhere in England are migratory, with there being some evidence that they winter in Norfolk (RSPB, unpubl.). Wintering flocks in the rest of England are generally presumed to be short-staying cranes of continental origin, rather than from a local breeding population (RSPB, unpubl.). Overall in 2005, there were five to seven pairs of crane at four sites in Britain, with four confirmed breeding pairs and one probable breeding pair in Norfolk, one pair breeding elsewhere in England, and one pair and one single bird at two sites in Scotland, although only the Norfolk birds successfully reared young (Holling and the RBBP, 2008). In 2008, the UK population was estimated at 11 - 12 pairs, with 6 - 7 of these occurring in Norfolk, and 52 wintering birds (34 in Norfolk, RSPB, unpubl.). The increase in records, and recent nesting attempts outside Norfolk, suggest that the species may be capable of recolonising Britain successfully (Holling and the RBBP, 2008). Crane is also currently being assessed for potential reintroduction elsewhere in southern Britain (Carter *et al.*, 2008).

The crane is an Annex I species (EC, 1979) and is on the amber list of conservation concern (Gregory *et al.*, 2002). No Special Protection Areas are designated for Crane in the UK (Stroud *et al.*, 2001).

Breeding Ecology

Breeding habitat

Cranes prefer to roost and nest in quiet flat areas adjacent to water, and areas associated with open marshes, sedge meadows and alder swamps (Mathews and Macdonald, 2001), the main requirement being for a wet area where they won't be disturbed (Leito *et al.*, 2006). They often use small patches of suitable habit within intensively cultivated landscapes (IUCN, 1996) and forage in surrounding areas with short vegetation such as arable fields and grassland (Mathews and Macdonald, 2001).

Breeding system

Cranes are monogamous, and the pair bond is long-lived (Cramp and Simmons, 1980). The nest consists of a large pile of available vegetation with a shallow cup on top (Cramp and Simmons, 1980). First eggs are laid from early April in Fenno-Scandia, and this is similar over most of the range (Cramp and Simmons, 1980). A clutch of two (rarely one or three) is laid (Cramp and Simmons, 1980). Monitoring at the Norfolk breeding site found that the maximum clutch size recorded was two, with 75 % of broods producing only one chick (Mathews and Macdonald, 2001). There is one brood, although replacements may be laid following egg loss (Cramp and Simmons, 1980). Mean incubation period is 30 days (range 28 - 31), and incubation is by both sexes, but predominantly the female (England, 1963). Both parents care for the young, often caring for one chick each if two young are present (Cramp and Simmons, 1980). Young are precocial and nidifugous (Cramp and Simmons, 1980). Fledging is at 9 - 10 weeks, after which the family joins a post-breeding flock (Cramp and Simmons, 1980), but may remain together until the following spring (Moll, 1963). Estimates of age at first breeding vary (2 years, Dementiev and Gladkov 1951; 4 - 5 years, Glutz von Blotzheim *et al.*, 1973; 4 - 6 years, Makatsch, 1970). Mathews and Macdonald (2000) found that for the Norfolk breeding population, age at first breeding ranged from three or four to seven years. As with many other large, long-lived species, the crane has a characteristically long generation time and low breeding success, which will limit the speed of population growth. Leito *et al.* (2006) suggest that cranes may live for 20 - 30 years in the wild, and have lived for 42 years in captivity; the longest known life span of a bird from the Norfolk population is 13 - 14 years (Mathews and Macdonald, 2001).

Home ranges and site fidelity

Crane nesting territories are usually large (Cramp and Simmons, 1980) and are predominantly defended by the male (Nowald, 2001). Nest sites and nests are reused in successive years (Cramp and Simmons, 1980, Leito *et al.*, 2006). Feeding is often outside the territory, with courtship and copulation up to 1500 m from the nest (Moll, 1963). Nearest-neighbour distances up to 5 - 6 km, and seldom less than 2 - 3 km were reported in Dementiev and Gladkov (1951), however, distances of 1.2 km and 0.23 km have been recorded in England (RSPB, unpubl.). Densities as high as 3 - 5 pairs per 40 ha have been recorded (Sieber, 1932). Most nests monitored in Estonia were also over 1 km apart (54 %), although 34 % were 0.5 - 1 km, and 18 % less than 0.5 km apart. In the early breeding season, the pair roost together, often outside the nesting territory. Sub-adults and unpaired birds form groups of 6 - 10 in the summering grounds, and, once fledging is completed, are joined by paired birds with their young prior to migration (Mathews and Macdonald, 2001).

The average size of home ranges in Estonia is 0.5 - 2 km² before and 5 - 10 km² after the chicks have fledged (Leito *et al.*, 2006). The distance covered by a crane family in a day is 2 - 8 km before, and 10 - 40 km after, the fledging of the brood. Feeding areas are situated up to 2 km away from the roosting site before fledging, and up to 15 km after (Leito *et al.*, 2006). Average home ranges of three crane families in Sweden were approximately 2 km² until 22nd August, 0.36 - 2.0 km² of which was used on a daily basis (Röper and Hake, 2003). The home range of one crane family pre-fledging in Germany was 0.7 km² in 1995 and 0.4 km² in 1996, and the distances covered in one day were 12.6 km and 14.4 km (Nowald, 1999). In the Czech Republic, the home range of one crane family was 5.5 km² before fledging and its total area was 10.4 km² (Peske *et al.*, 2003). Territory size depends on habitat; areas of 0.5 - 4 km² have been recorded on open bog with sparse scrub (Cramp and Simmons, 1980). Four studies cited in Mathews and Macdonald report territory sizes of 2 km² - 10 km² (including dead ground; e.g. see Johnsgard, 1983, Winter *et al.*, 1995, cited in Mathews and Macdonald, 2001). Thus in

cranes, home range sizes seem to range from about 0.4 to 10 km², these are equivalent to the areas of circles with radii 0.36 - 1.8 km.

Cranes are solitary and territorial in the breeding season, but usually gregarious during migration and in winter (Cramp and Simmons, 1980). A study in north east Spain found that a low proportion (2 %) of birds kept winter territories, these were small (mean = 0.7 km²) compared to the areas visited by gregarious birds (mean = 11.7 km², Alonso *et al.*, 2004). Observations in England differ from this, however, and suggest that winter home ranges of two pairs were approximately 3.5 km² and 4 km² respectively, with exclusive feeding ranges being held, whilst summer home ranges were within the winter home range area, but were smaller, being approximately 3.5 km² for two pairs, it not being clear how this was split between the two (RSPB, unpubl.).

Roosting

Crane winter quarters vary and include marshes, grasslands and cultivated land (Cramp and Simmons, 1980), and also can be actually in water in the UK and Germany (RSPB, unpubl.). Birds roost at communal sites outside the breeding season (Cramp and Simmons, 1980). The location of winter foraging ranges, and fidelity to them, is related to size and dominance in cranes; a study of 13 radio-tagged cranes in north east Spain found that dominant cranes spent 69 % of mornings in their preferred foraging area, whereas subordinate birds were more mobile (Alonso *et al.*, 1997).

Collision Risk

A large number of cranes, including common cranes, are killed in collisions with powerlines throughout Europe (particularly in Germany and Spain), and elsewhere (Prange, 1989, Tucker and Heath, 1994, European Crane Working Group, 2008). A study at eight 0.8 km segments of powerline in Colorado during spring and autumn migration periods from 1988 - 1991 found 706 fatalities, with waterfowl, whooping cranes *Grus Americana* and sandhill cranes *Grus canadensis* accounting for over 80 % of these (Brown and Drewien, 1995). Powerline markers can help to reduce this mortality (Brown and Drewien, 1995). Fatalities from birds colliding with traffic and fences have also been recorded (Leito *et al.*, 2006). When migrating, birds fly low in bad weather, but in suitable conditions, fly at heights up to 2000 m, and have been recorded up to 3200 m (Glutz von Blotzheim *et al.*, 1973).

Evidence of effects of wind farms on cranes is sparse. Hötter *et al.* (2006) identified cranes as being particularly susceptible to barrier effects from wind farms, with evidence of this in five different studies, implying they may avoid turbines. However, observations at an offshore wind farm in Sweden found that flocks of cranes passed over the wind farms at altitudes of 120 - 200 m, bringing them into potential collision course as they cross the Sound, especially during misty conditions (Pettersson, 2002). Moorehead and Epstein (1985) identified large wetland birds such as geese and cranes as particularly susceptible to collisions with wind farms. A common crane is amongst collision casualties recorded in Spain (Portulano, pers. comm.), and there is also a record of a sandhill crane colliding at Altamont Pass (Altamont Pass Avian Monitoring Team (2008).

Disturbance

Holling and the RBBP (2008) consider the crane a shy bird requiring large undisturbed areas in which to breed. However, nests can be built in relatively disturbed areas in some parts of the range (e.g. nests adjacent to roads in Germany; P. Newbery, pers. comm.). Leito *et al.* (2005) found a negative effect of human activity on nesting success during their study of breeding cranes in Estonia. Nest monitoring in Estonia found that most cranes were found in areas with weak human disturbance (67 %), followed by areas with moderate (29 %) and then

strong (4 %) human disturbance (n = 155, Leito *et al.*, 2006). Analyses showed that number of nestlings was significantly higher in areas with weak (1.41 +/- 0.80 chicks, n = 46) than moderate (0.81 +/- 0.81 chicks, n = 21) disturbance (the number of nestlings correlated with the number of fledglings, too few nests were found in areas of strong human disturbance to enter into the analyses, Leito *et al.*, 2006). A study in northeast Germany found that crane offspring have to feed throughout most of the day, and thus disturbance could affect reproductive success if it interfered with this, and suggested that planning should avoid traffic structures, powerlines and buildings, such as wind turbines, in areas of high crane density (Nowald, 2001). Cramp and Simmons (1980) report that cranes can be slow or reluctant to flush from the nest, however, with flushing distances in response to intruders of 10 - 20 m being recorded (Schüster, 1931). This does not necessarily indicate low sensitivity to disturbance as a low flushing distance could occur for reasons of strong site attachment.

Disturbance of roost sites is also a problem, and there are several examples of relocation due to this in Estonia (Leito *et al.*, 2006). A study of wintering cranes in southern Portugal found avoidance of roads and villages by cranes, despite a relatively low density of roads and villages in the study area (Franco *et al.*, 2000).

Meine and Archibald (1996) considered that cranes generally avoid human activity by a distance of at least several kilometres (Meine and Archibald, 1996). No information has been published on disturbance displacement of cranes from wind farms.

Sensitivity Criteria

Moorehead and Epstein (1985) considered cranes particularly susceptible to collision with wind farms, although there is little published about effects of wind farms on cranes. Recorded home ranges range from about 0.4 to 10 km², these are equivalent to the areas of circles with radii 0.36 to 1.8 km (although it is recognised that home ranges will not be circular and this is a crude estimation). Therefore, nest locations or protected areas containing cranes were plotted, buffered by 2 km, and this area classified as 'high sensitivity'. A precautionary approach is considered suitable given the very low population size.

Acknowledgements

These sensitivity criteria were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thanks to Dejan Bordan, Peter Newbery, Andy Stanbury, and various other RSPB staff, unnamed in order to protect crane locations, for comments on an earlier draft of this review.

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11. Osprey

Introduction

The osprey *Pandion haliaetus* has a widespread global distribution, breeding in boreal and temperate areas throughout the Palearctic region, from Europe to the Pacific coast (Cramp and Simmons, 1980, Poole, 1989). Most populations are highly migratory, wintering in Africa, India, south-east Asia and South America, although a few populations are sedentary (e.g. in south-east Asia and New Guinea, Stroud *et al.*, 2001). The species is polytypic; the nominate race *P. h. haliaetus* occurs throughout the Palearctic from Europe to Japan (Stroud *et al.*, 2001). Three further subspecies occur elsewhere (Stroud *et al.* (2001) suggests there are four, but the Australasian subspecies *Pandion haliaetus cristatus* has recently been reclassified as a separate species, *Pandion cristatus* by Christidis and Boles (2008)). Over 90 % of the European breeding population occurs in Sweden, Finland and Russia (Stroud *et al.*, 2001). Local extinctions occurred in Europe during the late 18th and early 19th century due to persecution (Cramp and Simmons, 1980, Hagemeyer and Blair, 1997), but subsequent protection from persecution in the early 20th century has resulted in recovery in some former parts of the breeding range (Stroud *et al.*, 2001). However, populations suffered again from the 1950s due to effects of organochlorine pesticides on breeding success (Poole, 1989, Hagemeyer and Blair, 1997). Restriction of these products allowed many populations to begin to recover from the 1980s (Hagemeyer and Blair, 1997).

In previous centuries, osprey would have been well distributed in parts of Britain, especially Scotland (Dennis, 1984). However, the species became extinct in 1916 due to persecution and egg collecting (Stroud *et al.*, 2001), although recent evidence suggests there may have been a small remnant population in Scotland (R. Dennis, in Stroud *et al.*, 2001). Some birds were present in the intervening years, but successful breeding was first recorded again in 1954 (Brown and Waterson, 1962, Dennis, 1987). Intensive protection of the Scottish population has occurred since this recolonisation, and the population increased at a rate of 10 % per year in the latter half of the last century (Saurola, 1997).

In England, the osprey may once have been a regular breeder in parts of the country, however the last recorded nesting attempt prior to the reintroduction programme was in 1842 in Somerset (Dennis, 1985). A reintroduction programme began in 1996 at Rutland Water, which had been regularly visited by birds on passage previously (Rutland Osprey Project, 2008). Sixty four young from Scotland were released between 1996 and 2001, and a further 11 in 2005 (Rutland Osprey Project, 2008). First successful breeding occurred in 2001 (Hawk and Owl Trust, 2008). In the same year, a pair naturally colonised the Lake District and bred by Bassenthwaite Lake, and in 2004 the first pair reared young in Wales; the male was a bird released at Rutland and the female a Scottish bird (Hawk and Owl Trust, 2008).

Holling and the RBBP (2008) consider that the Scottish population now appears to be stabilising, with 158 pairs known to lay eggs in 2004. Overall, an estimated 161 - 187 pairs bred in Britain in 2005, with two pairs nesting in England and one in Wales (Holling and the RBBP, 2008). In England, there is one breeding site in Northamptonshire, two in Leicestershire and Rutland, one in Nottinghamshire and one in Cumbria (Holling and the RBBP, 2008). Recent, more widespread, occurrence of summering pairs and individuals suggest that the English and Welsh populations will continue to increase in numbers and range (Holling and the RBBP, 2008).

The osprey is an Annex I species (EC, 1979) and is on the amber list of conservation concern (Gregory *et al.*, 2002). There are nine SPAs for osprey, all of which are in Scotland, and these hold an estimated 39 % of the British breeding population (Stroud *et al.*, 2001).

Breeding Ecology

Breeding habitat

Ospreys forage over fresh or coastal waters, most nests in Scotland are in old Scots Pine trees (usually live trees), but osprey will also nest in other tree species and occasionally man-made structures (Dennis, 1984, Thom, 1986, Dennis, 1987).

Breeding system

Ospreys are usually monogamous (Cramp and Simmons, 1980). In Scotland, birds begin to arrive from wintering grounds on the West African coast in late March, with most arriving in early April (Dennis, 1984). Egg-laying in Scotland occurs from mid April, more usually in late April (Dennis, 1984). Clutch sizes can range from one to four, but are nearly always two or three (Dennis, 1984). A study in Scotland found that about 16 % of females lay two eggs, these are usually females breeding for the first time, and clutches of three are usually laid in subsequent years (Dennis, 1984). Replacements are probably laid in the south of the range, but have not been recorded in Scotland (Cramp and Simmons, 1980). Incubation is by both sexes, but with the major contribution coming from the female (Cramp and Simmons, 1980) and lasts 34 - 40 days (mean = 37, n = 13, Green, 1976). Fledging occurs at 49 - 57 days (mean = 53, n = 35, Stinson, 1977). All or most brooding and direct feeding is performed by the female, with the male providing all or most of the food to the female and brood (Cramp and Simmons, 1980), bringing roughly three to seven fish per day to the nest depending on the age of the brood (Dennis, 1984). Young are semi-altricial and nidicolous (Cramp and Simmons, 1980). Chicks continue to be provisioned after fledging are usually independent after a further 1 - 2 months (Cramp and Simmons, 1980, mean of 30.4 days for chicks at Loch Garten, n = 35, Bustamante, 1995).

Home ranges and site fidelity

Densities

Mean distances between nest sites of 8 km have been recorded in Scotland, with minimum distances of 200 m between successful nests (RSPB in Cramp and Simmons, 1980).

Site fidelity

Nests are usually reoccupied between years, unless breeding is unsuccessful, in which case a nest may be built late in the year following the failure, and used in the subsequent year (a 'frustration eyrie'; Cramp and Simmons, 1980, Dennis, 1984). Sites are often traditional and with a long history of occupation e.g. 10 - 92 years at 21 nest sites or restricted nestling localities in 19th century Scotland (Brown and Waterson, 1962), up to 125 years in North America (Bent, 1937, Moll, 1962), and with a nest series in Scotland involving five sites in an area of 50 km² being used over 23 years from 1954 (Weir in Cramp and Simmons, 1980). Dennis (1984) states that in general, if a pair breed successfully and survive over winter, they will breed again at the same nest the following year. A female ringed at Loch Garten in 1968 bred in 1973 and had the same mate and nest site every year until 1983 (Dennis, 1984). Ospreys from a population on Kangaroo Island, Australia were also found to stay together over many seasons and use the same primary nest site (Dennis, 2007), although it should be noted that the Australasian osprey *Pandion haliaetus cristatus* has been reclassified as a separate species (Christidis and Boles, 2008).

Foraging range

Birds defend a nesting territory, but foraging ranges are not defended and can be used simultaneously (Cramp and Simmons, 1980). The size of the home range is highly variable, and the feeding area can be located well away from the nest site. In Scotland, of 14 nests in

1976, four were more than 3 km from the nearest fishing water (RSPB in Cramp and Simmons, 1980). This distance is not uncommon (Prevost, 1977, Häkkinen, 1978) and males may fish up to 10 km or 20 km away (Dunstan, 1973). Radio-tracking of eight males at a breeding colony in North Carolina found that three sites, each about 14 km from the breeding grounds, were the primary foraging grounds, with individual males remaining very faithful to one site for the two months studied, and two males tracked the following year still using the same foraging sites they had used the year before (Hagan, 1986).

Collision Risk

Flight behaviour

Sky dancing is the most common display until incubation, and seems to have both courtship and territorial functions (Cramp and Simmons, 1980). The display occurs from the male's arrival at the breeding site, and rapidly declines in frequency following arrival of the female (Cramp and Simmons, 1980). The sky dance may begin and end at the nest site, and the male usually calls and carries nest material or fish (Cramp and Simmons, 1980). The male rises to about 300 m, before hovering and then diving, and then rising again steeply to repeat the performance once more or several times (Cramp and Simmons, 1980). The display can last up to nine minutes (Moll, 1962). A hovering display flight may also be performed by either bird high in the sky (Cramp and Simmons, 1980), and pairs may high circle together soaring above the nest site (Brown and Amadon, 1968).

Foraging flights involve plunging from the air; dives can be made from 5 - 70 m, but heights of 20 - 30 m are most common, and can be made from flight or from a perch (Cramp and Simmons, 1980).

Five osprey were present in the study area during pre-construction monitoring for Columbia wind farm #1 in Klickitat County, Washington, and observations found that flight heights ranged from 40 to 125 m, with a mean height of 79 m (Erickson *et al.*, 2003). Fifty percent of osprey flights were at rotor-swept height (defined as 25 - 75 m), and 50 % of flights at heights over 75 m (Erickson *et al.*, 2003). What types of flights these were, the time of year and age of the birds was not specified, however (Erickson *et al.*, 2003).

Disturbance

Cramp and Simmons (1980) suggest that osprey are generally wary of disturbance, but locally have adapted to intensive human activity. Nest site location and densities become flexible in areas with high food availability, with birds using pylons, windmills and bridges for nest sites in some areas (Cramp and Simmons, 1980).

Other accounts of effects of disturbance on ospreys vary. Poole (1989) suggests that nest visits and aerial surveys to monitor ospreys have no effect on breeding success. Ospreys in the eastern United States can breed in suburban habitat, less than 300 m from roads, railways, boat channels or houses, and these raise as many young as those in nature reserves with low numbers of visitors (Poole, 1981). Swenson (1979) however, found that pairs breeding near campsite in Yellowstone National Park (Wyomin, USA), had poor reproductive success compared to pairs in more isolated areas of the park. Likewise, ospreys in northern California had lower breeding success near logging roads (Levenson and Koplín, 1984). In Scandinavia, an increase in boating was thought to threaten the success of nests on certain lakes (Haga, 1981, Hallberg *et al.*, 1983). Poole (1989) suggested that habituation occurs in osprey and that sporadic disturbances are more likely to affect breeding success, and this could account for the variable responses to disturbance in the species.

Flushing distances of eight waterbird species whilst foraging or loafing were investigated in response to the approach of a personal watercraft and an outboard powered boat during September to November 1998 and April to June 1999. Osprey showed the highest response distances, with flushing distances of 49.5 m and 47.9 m to the respective stimuli (Rodgers and Schwikert, 2002). Flushing distance was related to a species' body size (Rodgers and Schwikert, 2002). It was suggested that buffer zones of 150 m would minimise disturbance of ospreys from human activity at foraging and loafing sites in Florida (for calculation see Rodgers and Schwikert, 2002). Rodgers and Smith (1997) found that five of nine species studied exhibited significantly greater flushing distances when foraging or loafing than when nesting. It should be remembered, however, that disturbance could still impact on breeding success despite not eliciting a flushing response.

Buffer zones restricting human activities within 660 feet (201 m) of active nests, and banning cutting of trees within 200 feet (61 m) have been recommended, with the option of reducing this radius to 130 feet (40 m) depending on topography and visibility of sources of disturbance (see Zarn, 1974, Westall, 1986, Rodrick and Milner, 1991). In remote areas, it has been recommended that campsites should not be located within 1100 m, and hiking trails not within 300 feet (91 m), of occupied nests (Levenson, 1979). Richardson and Miller (1997) recommended a protective buffer in the range of 400 – 1500 m depending on site characteristics, and gave a median distance of 1000 m, based on a review of other studies. Currie and Elliot (1997) suggested a buffer of 350 – 1000 m around osprey nests for forestry workers. A survey of expert opinion by Ruddock and Whitfield (2007) gave suggested disturbance distances ranging from 100 - 150 m to 500 – 750 m for static disturbance (i.e. the 'alert distance' at which a bird changes its behaviour but without taking flight), and an upper limit on active disturbance (i.e. 'flight initiation distance') of 500 - 750 m. Ruddock and Whitfield (2007) suggest that these distances should be used flexibly, due to presence of some birds in areas with high human activity, and habituation to disturbance

Sensitivity Criteria

Given the extremely scarce nature of this species in England, a precautionary approach has been taken. Use of a circular buffer to estimate foraging ranges does not seem appropriate as foraging flights will generally be to a limited number of feeding waters and can occur over large distances. Experts from the osprey projects were consulted about buffers containing nest sites and important foraging locations. In addition, records of nest sites from the 10 years from 1996 - 2007 were plotted, and included if more than one breeding attempt had occurred within 1 km during this period (otherwise the location was considered too transient). These locations were buffered by 2 km, to account for the fact that aerial displays can occur within 1 km to 2 km of the nest (R. Thaxton, pers. comm.) and collision risk may be increased during such flights. These areas were all classified as 'high sensitivity'.

Acknowledgements

These sensitivity criteria were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thanks to Tim Appleton, Nigel Butcher, Ian Carter, Jason Godfrey, Richard Mackrill, Richard Thaxton, Colin Wilkinson, Tim Youngs who commented on an earlier draft of the review.

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12. Golden eagle

Introduction

The golden eagle has a northern global breeding distribution that includes the Palaearctic (as far south as northern Africa and south-east Asia), as well as North America (Stroud *et al.*, 2001). The global population is estimated at 50 000 to 100 000 pairs (Watson, 1997), with the European population standing at 5239 to 5616 pairs (Hagemeyer and Blair, 1997). Global population trends have not been quantified, but populations appear to be stable (Ferguson-Lees and Christie, 2001) with the exception of some regions such as the east Baltic and south east Asia where there have been declines (Stroud *et al.*, 2001). The European breeding population was stable between 1970 and 1990, and remained stable or increased in the majority of countries between 1990 and 2000 (BirdLife International, 2004). However, because the small size of the European population makes it vulnerable, *Birds in Europe* classifies the golden eagle as 'Rare' (BirdLife International, 2004).

In the UK, the species occurs primarily in the uplands, although several pairs exploit coastal habitats (Stroud *et al.*, 2001). The vast majority of the population is found in the Scottish Highlands (Dennis *et al.*, 1984, Green, 1996, Watson, 1997). Prior to 1800, there were possibly 500 pairs in Scotland and at least 50 in England and Wales (Brown, 1976, Holloway, 1996). The UK population declined as a result of persecution in the 18th and 19th centuries to a low point of only 80 regularly used eyries by 1870 (Watson, 1997). Declines continued in the 19th century due to persecution (Watson, 1997), loss of open land to coniferous afforestation (Watson *et al.*, 1989), and in the late 1950s and early 1960s possible effects of organochlorine pesticides (Lockie *et al.*, 1969, Newton and Galbraith, 1991). The British population is thought to have increased since the 1950s following the introduction of protective legislation, notably the Protection of Wild Birds Act in 1954 (Watson, 1997). The last national survey in 2003 found 443 pairs of golden eagles breeding in Scotland (Eaton *et al.*, 2007), a small increase since the 1992 national survey when 422 pairs were found (Green, 1996), with highest densities occurring in the western Highlands and Islands (Stroud *et al.*, 2001). However, the size and range of the Scottish golden eagle population is still limited in by human persecution associated with grouse moors (Whitfield *et al.*, 2004a, 2004b, 2007).

One pair of golden eagle started breeding at Haweswater, Cumbria, in 1969, but despite being reasonably successful until recent years, this nest has remained the only one in England (RSPB website, 2008). The female died in 2004, leaving just a single male present in 2005 (Holling and the RBBP, 2008).

The UK golden eagle population constitutes around 7.4 % of the European breeding population, and virtually all of these birds occur in Scotland. The golden eagle is listed on Annex 1 of the EU Birds Directive (EC, 1979) and is on the amber list of conservation concern in the UK (Gregory *et al.*, 2002). Around 60 pairs, or 15 % of the British breeding population, occur with Special Protection Areas (Stroud *et al.*, 2001).

Breeding system

Golden eagles have a long-term monogamous pair bond (Cramp and Simmons, 1980). A clutch of one to three eggs (mean = 1.91 in Scotland, Gordon, 1955) is laid from early March, and these are incubated for 43 - 45 days per egg (Cramp and Simmons, 1980). Young are semi-altricial and nidicolous, and are cared for by both parents, but predominantly the female, who does all or most of the incubation, brooding and direct feeding (Cramp and Simmons, 1980). The male provisions the female and young, and may very occasionally feed the young directly (Cramp and Simmons, 1980). Incubation usually takes 65 - 70 days, and young are dependent for a further 90 to 100 days (Cramp and Simmons, 1980). There is one

brood, a replacement clutch may be laid after failure at the egg stage, but this is considered rare (Cramp and Simmons, 1980).

Home ranges and site fidelity

Site fidelity

Golden eagles breed habitually in the same nest sites, though pairs use alternate nest sites in different years up to 6 km apart (Watson and Rothery, 1986). Cramp and Simmons (1980) state that nests are re-used in successive years, although several nests may be used in turn, with 12 pairs in north west Scotland having 64 eyries in all (mean = 5.3 per pair, range 3 - 9, Watson, 1957). Pairs remain in the same home range throughout the year, and usually roost in the nesting territory in winter where possible (Cramp and Simmons, 1980).

Home ranges

A simple model to predict golden eagle's home ranges was developed by McGrady *et al.* (1997, 2002), from information gathered from radio-tracking of birds in Argyll. This is known as the RIN (Research and Information Note), RIN 292, or McGrady model and involves a number of stages:

Box 1: The RIN model

1 *Range Centre*

The range centre is found by taking the mean nesting location over the past 10 years. If there are clusters of nests that have means over 2 km apart then there will be two centres in the range.

2 *Core Area*

This is the central core of the range, which encompassed 50 % of the ranging locations of eagles. In its simplest form, it is a circle of 2 - 3 km radius around the range centre.

3 *Territory boundary*

a Where neighbours were present, this is defined by drawing a line between the neighbouring range centres, and then drawing a line perpendicular to this line at the mid-point between the two centres. This is done for all of the neighbouring ranges, and the lines joined to define the territory (a Thiessen polygon).

b In the absence of near neighbours, eagles ranged up to 9 km from the centre (although 98 % of records were within 6 km of the range centre). A curved line is drawn at 6 km from the centre to connect adjacent boundary lines drawn in step 3a.

4 *Altitudinal cut-off*

Eagles preferred an altitude range of 150 - 500 m (except on Mull where it was sometimes lower). High areas were avoided, but this may not be true in the eastern Highlands where ptarmigan and mountain hare are more plentiful. Thus, different cut-offs were used for eagles nesting at different altitudes:

- High altitude ranges, e.g. the Cairngorms, the altitudinal cut-off outside the core area was defined as 150 - 200 m above the valley floor.
- Medium altitude ranges, e.g. mainland Argyll, the altitudinal cut-off was defined as 150 m out of the core area.
- Low altitudes ranges, e.g. Isle of Mull, all altitudes were used, excluding areas of high human activity.

This model was refined by McLeod *et al.* (2002a, b) to create the PAT (Predicting *Aquila* Territories) model. The range centre was estimated as before, but a more complex model predicting percentage use of the range at a 50 m x 50 m pixel level was created. Whilst the RIN model used a set cut-off distance of 6 km to define the territory boundaries in the absence of near neighbours, the PAT model allows for the fact that this may vary according to density of breeding eagles, for example it is probably reduced in high density areas such as some Hebridean Islands. Maximum ranging distance was found to correlate with the area of the Thiessen polygon for a range (see 3b above). Areas close to the centre of a range were assumed to be used more. The maximum ranging distance was used to define the percentage use in concentric rings around the range centre, such that the total use was 100 %. Preference for ridge features (to aid soaring) was incorporated, as was avoidance of unsuitable habitat: fresh water bodies, sea, woodland over 12 years old and areas of high human activity (with various buffer zones, see below). When compared to the RIN model, the RIN predicted larger ranges. Home ranges may also vary depending on breeding status and season (Haworth *et al.*, 2006). However, the PAT model requires more detailed information on habitat and topography, making it less feasible to apply at a country level scale for sensitivity mapping.

Effects of wind farms on golden eagles

Work by Fielding *et al.* (2006) and Bright *et al.* (2008) found that overlap of golden eagle core ranges and wind farms is low at present. The sensitivity map is intended as a useful tool to maintain this low degree of overlap between sensitive species and wind farms. Wind farms may affect golden eagles predominantly by collision or disturbance displacement. Where a golden eagle pair's territory did coincide with a wind farm on the Kintyre Peninsula, a land management scheme was implemented with the aim of enhancing habitat to increase available prey (red grouse) away from the wind farm, whilst reducing the large red grouse population in the wind farm area by impoverishing the local habitat (Madders and Walker, 2002).

Collision Risk

Sky dancing is the most common aerial display in golden eagle, and can be by both sexes, but is usually by the male, and can be for courtship or territorial purposes (Cramp and Simmons, 1980). During this display, birds often make use of topographical features such as ridges or mountain tops (Cramp and Simmons, 1980). High circling by individuals or pairs is also common, and flight play may occur but is infrequent (Cramp and Simmons, 1980).

High numbers of collision fatalities have been reported at the Altamont Pass Wind Resource Area (APWRA) in California, a 165 km² wind farm with 5400 turbines. Carcass searches within 50 m of 4074 turbines for periods ranging from six months to 4.5 years estimated that there were 67 golden eagle fatalities annually at the whole wind farm (80 % confidence interval 25 – 109, this estimate is corrected for searcher detection and scavenger removal rates, Smallwood and Thelander, 2008). Proportionally more golden eagles were killed than would be predicted by chance, and this is probably due to hunting behaviour (Orloff and Flannery, 1992). A review of wind farm impact studies, focusing mainly on Europe, found just one reported casualty of a golden eagle due to collision with a wind turbine in Spain (Hötker *et al.*, 2006). It should be noted that this review is the result of collation of records from a variety of sources, as opposed to controlled corpse searches at wind farms. Raptors generally appear to be particularly susceptible to collision (e.g. NWCC, 2000, Langston and Pullan, 2003, Smallwood and Thelander, 2004, Thelander *et al.*, 2003, Anderson *et al.*, 2004, 2005) and raptors and gulls accounted for most of the fatalities reported in Hötker *et al.*'s (2006) review, with white-tailed eagle being of particular concern.

Disturbance

There is little information on disturbance distances, or understanding of which activities are affected most by disturbance in golden eagle (McLeod *et al.*, 2002b). The effects of disturbance also vary throughout the range. In recent years, an expanding population has moved into areas which are more disturbed to establish new breeding ranges (Haller, 1996). Such a situation does not apply to Scotland or England (Watson and Dennis, 1992).

Two studies in the USA found no evidence of disturbance displacement of golden eagles due to wind farms (Johnson *et al.*, 2000, Schmidt *et al.*, 2003), but a third study at the Altamont Pass wind farm in California found some evidence of displacement in comparison with a control area (Hunt *et al.*, 1995). Golden eagle behaviour was monitored before and after construction of the Beinn an Tuirc wind farm in Argyll in 2001 within the range of a pair of golden eagles. The eagles' range size remained the same, but the eagles appeared to change their ranging behaviour to avoid the wind farm site (although there are problems with interpreting this result as the construction of the wind farm coincided with some local land management changes, Walker *et al.*, 2005).

In the UK, golden eagles are most sensitive to disturbance near the nest in February to July (McGrady *et al.*, 1997), and disturbance can influence golden eagle behaviour and productivity (Watson, 1997). Disturbance free distances of 900 - 1000 m radius around the nest have been recommended in Britain (Petty, 1998). This may vary, being increased to around 1.5 km if disturbance is directly in the line of sight from the nest, and decreased if the disturbance is concealed in some way (to a minimum of 750 m, McGrady *et al.*, 1997). Disturbance-free distances of 200 - 1600 m have been recommended in North America (Suter and Jones, 1981). McLeod *et al.* (2002b) assumed areas of human activity were avoided by eagles, and used the following buffer zones: single building: 250 m; cluster: 400 m; village: 600 m; town: 800 m; single carriageway roads: 300 m and dual carriageway roads: 500 m, although these were based on limited observations. Disturbance displacement could effectively lead to habitat loss for golden eagle pairs, which could result in reduced productivity or abandonment. Whitfield *et al.* (2001) found that afforestation of just 10 % of a golden eagle's core range can lead to a large reduction in productivity (Whitfield *et al.*, 2001), and home ranges may be abandoned if up to 40 % of the core is occupied by closed canopy forestry (Watson, 1992).

Elsewhere, in a study area in Norway, Bergo (1984) found that all nests were found at a distance of more than 500 m from permanent human settlement, and usually more than 1 km away from roads. Three nests were found less than 500 m away from a cabin or mountain farm house (minimum 250 m), but one occupied nest situated 450 m from a cabin was deserted during the incubation period.

Sensitivity criteria

Although at present there is only a single male at Haweswater, a pair nested there until recently, and in light of the fact that this is the species' only breeding site in England and the potential for recruitment of a new mate, the territory was included on the sensitivity map. The mean nest location for the 10 years was taken as the territory centre, and the area within a buffer of 2.5 km around this classified as 'high sensitivity', with the area between this buffer and an outer buffer of 6 km classified as 'medium sensitivity'. This is based on the RIN model, which suggests that golden eagles are likely to spend 50 % of their time within a 2 - 3 km radius 'core range', and 97 % of their time within 6 km of the territory centre (McGrady *et al.*, 1997, 2002), and is intended to reflect use of the range. Use of the more refined PAT model is beyond the scope of this project.

The same approach will be taken for golden eagles nesting in Scotland, where mapped estimates of foraging ranges extend into England.

Acknowledgments

These sensitivity criteria are based on a previous version written for a Scottish sensitivity map (Bright *et al.*, 2006, 2008), updated in light of new information and to place them in the context of English population sizes and trends. They were written as part of a project to create a sensitivity map to aid the location of onshore wind farms in England with respect to a suite of sensitive bird species. Thanks to Ian Carter, Mark Eaton, Martin Kerby and Tim Youngs who provided helpful comments on an earlier draft of these sensitivity criteria.

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