Masden, Elizabeth Anne (2010) *Assessing the cumulative impacts of wind farms on birds*. PhD thesis.

Assessing the cumulative impacts of wind farms on birds

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This thesis is submitted in fulfilment of the requirements for
the Degree of Doctor of Philosophy

University of Glasgow
Division of Ecology and Evolutionary Biology

April, 2010

Abstract

As governments pledge to combat climate change, wind turbines are becoming a common feature of terrestrial and marine environments. Although wind power is a renewable energy source and a means of reducing carbon emissions, there is a need to ensure that the wind farms themselves do not damage the environment. There is particular concern over the impacts of wind farms on bird populations. In this thesis I have explored how to assess the cumulative impacts of wind farms on birds.

Cumulative impact assessment is a legislative requirement of environmental impact assessment but too frequently it has been tacked on to the end of assessments as an afterthought. Reasons for this are numerous but a recurring theme is the lack of clear definitions and guidance on how to perform cumulative assessments. Therefore I developed a conceptual framework to promote transparency. The core concept is that explicit definitions of impacts, actions and scales of assessment are required to reduce uncertainty in the process of assessment and improve communication between stakeholders. Only when it is clear what has been included within a cumulative assessment, is it possible to start to make comparisons between developments. I also recommend a more strategic approach to cumulative impact assessment.

If birds avoid wind farms then the structures act as barriers to movement and birds must fly further to reach their destination. If the additional distance has an associated energetic cost then this will impact an individual. With data collected using surveillance radar, I investigated the impact of the Nysted offshore wind farm on a population of common eider *Somateria mollissima* migrating from Finland to the Wadden Sea. The impacts of the wind farm appeared trivial and it required 100 equivalent wind farms before a significant impact was detected. Using the same radar data I also constructed a model to quantify the movement process of birds in response to wind turbines and therefore provide wind farm developers with a useful tool to predict the impacts of different wind farm designs.

The impacts of wind farms may be greater for birds that interact with the turbines on a daily basis than for migrating individuals. Using an energetic modelling approach I explored the impact for a suite of breeding seabirds commuting past a wind farm between their breeding and feeding areas. Impacts of flying increasing distances associated with increasing numbers of wind turbines were species-specific and costs were highest for species with high wing loadings and high daily frequency of foraging flights. However, costs of extra flight to avoid a wind farm appear much less than those imposed by low food abundance or adverse weather conditions.

Finally, a spatially-explicit individual-based model was developed to assess cumulative impacts of wind turbines through collision mortality and direct and indirect habitat loss, on a population of hen harriers *Circus cyaneus* on Orkney. Increasing numbers of wind turbines caused declines in the hen harrier population but the population response varied according to where turbines were located. Therefore, although wind turbines impact hen harriers, it may be possible to reduce the effects by considering hen harrier ecology during the planning procedure and/or implementing mitigating measures such as rough grassland restoration in strategic locations away from turbines.

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Acknowledgements

First and foremost I would like to thank my supervisors in both Scotland (Dan Haydon, Bob Furness and Rhys Bullman) and in Denmark (Tony Fox) not only for their unwavering support and encouragement but also for providing exceptional opportunities both academically and otherwise. Without Dan I would have no memories of snowy days on the hills, the Ben Lui Suicide Club, or sailing on Loch Lomond; without Bob I would not have set foot on St Kilda or Foula, two fascinating islands; and without Tony and the Fox clan of Nimtofte I would not have experienced Denmark from the inside out, Julenisse included!

Members of Room 303, the Boyd Orr Centre for Population and Ecosystem Health and more generally DEEB, have supported me throughout this journey. I am eternally grateful to Richard Reeve for answering never-ending questions (one day they will stop!), and to Hawthorne Beyer and Douglas Kerlin for discussing problems regarding spatial analysis and modelling (as well as giving me an excuse to buy doughnuts!). Thanks also must go to Gina Prior for who else would send me a gannet in the post, Clare Marsden, Pen Holland, and Harry (the collie dog); all of whom have kept me somewhat sane.

Despite undertaking a modelling project, I have had the opportunity to get out of the office and into the field on several occasions and for that I must thank Will Miles, Eliza ‘do-a-lot’ Leat, Kate Brookes (University of Aberdeen) and Mark Desholm (NERI). Outside of the department I would like to show my gratitude to James Pearce-Higgins whose enthusiasm knows no bounds and to Else-Marie Nielsen for ensuring my Danish excursions were always enjoyable (even if I did end up in hospital!). Also my weeks would not have been complete without the Midgierakers and beer and crisps in the Res Club.

This thesis would not have been possible without the help of others. Many thanks to all who attended the COWRIE workshops on cumulative impact assessment and the SNH Sharing Good Practice event where many of the issues discussed in chapter 2 were deliberated. Many thanks also to the Finnish Ringing Centre, Thomas Kjær Christensen and Henning Noer for information and advice on eider biology and Ib Krag Petersen for spatial analysis guidance. My thanks to Colin Pennycuick for enthusiasm and advice on calculating the energy required for bird flight (chapter 4) and Matt Denwood for advice on Bayesian methods

(chapter 5). Also, to Erik Meek and Steve Redpath for providing data and information on hen harrier ecology, and lest I forget, SNH for putting a roof over my head, and Lorna Kennedy for putting ice cream in my tummy!

And finally I must thank a very special group of people: my parents who have always given me support and encouragement to achieve what often seems impossible, I cannot thank you enough; ALG, for coming along for the ride; Jay and Sarah for opening up their home to me (and letting me steal biscuits!); and of course, to Neil, for always being by my side. Mo ghaol ort. Now let the next adventure begin...

Author's Declaration

I declare that the work recorded in this thesis is entirely my own, except where otherwise stated, and that it is also of my own composition. Much of the material included in this thesis has been produced in co-authorship with others, and my personal contribution to each chapter is as follows:

Chapter 2. Masden, E.A., Fox, A.D., Furness, R.W., Bullman, R., and Haydon, D.T. 2010 Cumulative impact assessments and bird/wind farm interactions: developing a conceptual framework. *Environmental Impact Assessment Review*. **30**:1-7. Initial concept developed by DTH, RWF and EAM. Manuscript drafted by EAM. Final draft enhanced by DTH, ADF and RWF.

Chapter 3. Masden, E. A., Haydon, D. T., Fox, A. D., Furness, R. W., Bullman, R., and Desholm, M. 2009. Barriers to movement: impacts of wind farms on migrating birds. *ICES Journal of Marine Science*. **66**: 746-753. Data collection by MD. Initial concept developed by ADF, RWF and EAM. Analysis conducted and manuscript drafted by EAM. Final draft enhanced by DTH, ADF, RB and RWF.

Chapter 4. Masden, E.A., Haydon, D.T., Fox, A.D., and Furness, R.W. 2010 Barriers to movement: modelling energetic costs of avoiding marine wind farms amongst breeding seabirds. *Marine Pollution Bulletin*. In Press. Initial concept developed by ADF, RWF, DTH and EAM. Analysis conducted and manuscript drafted by EAM. Final draft enhanced by DTH, ADF and RWF.

Chapter 5. *In preparation for submission as*: Masden, E.A., Haydon, D.T., Desholm, M., Fox, A.D., Furness, R.W., and Reeve, R.E. Modelling bird movements around wind farms: a Bayesian approach. Data collection by MD. Initial concept developed by DTH and EAM. Model development by RER, DTH and EAM. Analysis conducted and manuscript drafted by EAM. Final draft enhanced by DTH, ADF, RWF, MD and RER.

Chapter 6. *In preparation for submission as*: Masden, E.A., Haydon, D.T., Amar, A., Furness, R.W., Fox, A.D. and Pearce-Higgins, J. Assessing the cumulative impacts of wind farms on birds: an individual-based model of hen harriers in Orkney. Initial concept developed by DTH, EAM, JP-H and RWF. Data collection by AA. Model development by EAM, JP-H, AA and DTH. Analysis conducted and manuscript drafted by EAM. Final draft enhanced by DTH, JP-H, ADF and RWF.

I further declare that no part of this work has been submitted as part of any other degree.

Elizabeth A Masden
University of Glasgow, April 2010

1 General Introduction

1.1 Climate change and the role of renewable energy

“Climate change is the most severe problem that we are facing today – more serious than terrorism” (King, 2004) and will impact on the global economy (Stern, 2008), worsen problems such as drought, disease and famine (Galbraith and Baxter, 2008), and affect human well-being. However, the effects of climate change are not restricted to *Homo sapiens* but pervasive throughout the natural world (Parmesan and Yohe, 2003) with many species experiencing changes in physiology, phenology, and distributions, leading to changes in community and ecosystem structure and functioning (Walther et al., 2002).

“Climate change is real, and the causal link to increased greenhouse emissions is now well established” (King, 2004). Although there has been much variation in climate throughout history, the changes more recently observed would not have happened without the rapid expansion in wealth and numbers of humanity (Lovelock, 2008), the majority of whom have an unsustainable addiction to fossil fuels (MacKay, 2008)! There are numerous methods by which to reduce carbon emission however as fuel consumption comprises a large proportion of global carbon emissions, one obvious way of decreasing emissions is to switch to low carbon energy sources. The UK government has set a target to provide 20% of its electricity from renewable sources such as sunlight, wind, and tides, by 2020, and in Scotland there is a more ambitious target of 50% by 2020 which translates into a requirement of over 8 GW of energy generating capacity in the next 10 years.

When considering renewable energy, Scotland has a number of potential opportunities with approximately 23% of the total European wind energy resource, as well as the marine energy resource, forestry biomass and hydropower (RSPB Scotland et al., 2006). Unsurprisingly, it is wind that has received the most attention, as wind powered technologies can be dated back as far as 1000AD (Ackermann and Söder, 2000). Since the 1980s wind has been used for large-scale electricity generation but more recently there has been a worldwide growth in wind power, not least in the UK and particularly Scotland (Warren and Birnie, 2009). The total installed renewable capacity in Scotland is now 2834 MW with 49% of that being generated from onshore wind (SNH, 2009), meaning that energy generation by onshore wind has now overtaken hydro power in Scotland. There are also many more onshore wind schemes that have been granted planning consent but are not yet constructed and these will account for another 3350 MW. The rapid increase in the number of large wind farms has proved highly controversial in Scotland and often the controversy has focused on birds.

1.2 Birds and wind farms

All energy generating technologies have environmental, economic and social costs and benefits (RSPB Scotland et al., 2006). Climate change has been shown to impact birds, and although there is a real need for wind farms as a means of reducing carbon emissions, there is also a need to ensure that the wind farms themselves do not impact negatively on the environment (Elphick, 2008). The impacts of wind farms on bird populations can be classified into three groups: direct mortality of individuals due to collision with the turbines and infrastructure; physical habitat change due to the turbines and associated infrastructure; and displacement due to behavioural responses of the birds to the turbines. Here I shall review previous research in this area and highlight the methods used. Although not exhaustive, this literature review includes the majority of studies to date with examples covering all of the methods used thus far to assess the impacts of wind farms on birds.

1.2.1 Direct collision mortality

Direct collision mortality has been documented globally at many wind farm sites, although at the majority the collision rates are low (Hötker et al., 2006). However, high collision mortality has been recorded at some poorly-sited wind farms, with the best documented cases being the Altamont Pass in California, Smøla in Norway, and Tarifa in Spain. Direct collision impacts birds across a range of species from passerines to seabirds however, due to their ecology it is the long-lived birds such as raptors that have caused the most concern. For terrestrial sites, the collision rate is studied using protocols for measuring collision fatalities such as a systematic corpse search of the site (Barrios and Rodriguez, 2004, Langston and Pullan, 2003). When calculating collision rate from corpse searches however several factors must be taken into consideration; one must account for the removal of corpses by scavengers and also include an encounter probability and observer error i.e. the fact that a human is unlikely to detect all corpses in the area (Morrison, 2002). When studying the collision rate of birds with marine wind farms the task becomes more difficult as the avian casualties would most likely be removed from the immediate area by currents. Newton and Little (2009) tried to account for this when assessing wind-farm bird casualties on a Northumbrian beach by monitoring the transport of floating wooden blocks. When direct measurement of collision casualties is not possible collision risk modelling is used. Risk assessment models, such as the Band Collision Model, take into account factors that may affect mortality such as the volume of air swept by the blades, the flight behaviour of the birds, abundance of birds, and the size and alignment of the turbines (Band et al., 2007, Desholm and Kahlert, 2007, Drewitt and Langston, 2006, Smales, 2006, Cruz-Delgado et al., 2010). It should be noted however that a study of bird fatalities found no effect of turbine height or rotor blade diameter (Barclay et al., 2007) and de Lucas et al. (2008) also found that, for raptors at least, collision was not closely related to abundance. Alongside visual observations, remote sensing techniques such as Thermal Animal Detection Systems (TADS) and radar surveillance can be used to parameterise these models (Desholm et al., 2006, Kelly et al., 2009). Unfortunately, a persistent problem surrounding collision risk modelling is that many birds exhibit avoidance behaviour towards wind farms and the model results are highly dependent on this avoidance parameter (Chamberlain et al., 2006). To date, it has not been possible to adequately quantify the avoidance rate, and therefore best estimates are used. As a result, many collision risk models produce estimates that are likely to be very inaccurate.

Alongside estimating current collision mortality, a number of studies have also attempted to estimate the potential future impacts of wind farm collision mortality by taking a more general modelling approach. A geographical assessment was undertaken in Spain to estimate the impact of wind farms on migratory bird species (Tellería, 2009) and Bright et al. (2008) used spatial analysis tools to map the ranges of bird species of conservation priority in Scotland and thus assessed bird sensitivity to future wind farm developments. Sensitivity indices have also been suggested by both Garthe and Hüppop (2004) and Desholm (2009) in studies that used species-specific abundance and demographic parameters to characterise the sensitivity of waterbird species to wind farm associated mortality. Despite these studies of sensitivity, further studies extending the concept of sensitivity to estimate long-term population effects of wind farms are rare. One example (Carrete et al., 2009) uses population viability analysis to explore the effect of wind-farm mortality on population projections for a long-lived raptor.

1.2.2 Habitat loss, or possibly gain

Habitat loss, or possibly gain, can occur directly or indirectly as a consequence of wind farm development. Direct habitat loss is a function of the size of the footprint of the wind turbine tower and the number of turbines and is likely to be the least consequential impact of a wind farm with the loss amounting to between 2-5% of the total development area (Fielding et al., 2006, Fox et al., 2006). Displacement of birds due to disturbance can be viewed as effective habitat loss and can impact reproduction and survival. To date, several methods have been used to study effective habitat loss, for example aerial surveys were used at the Horns Rev wind farm in Denmark to assess the abundance and distribution of sea ducks whilst Pearce-Higgins et al. (2009) used visual observations to assess the distribution of breeding birds around upland wind farms. Using survey methods is viable for species that aggregate, for instance sea ducks, but less applicable for wide-ranging species hence Perrow et al. (2006) used radio telemetry to assess the foraging range of breeding little terns *Sterna albifrons* from a special protection area, in relation to Scroby Sands offshore wind farm. These methods however are only effective if a before-after-control-impact (BACI) design is used allowing comparisons to be made between the habitat use before the wind farm was present and afterwards.

Ecological models have been developed to estimate the impact of wind farms through habitat modification for upland bird species such as golden eagle *Aquila chrysaetos* (Madders and Whitfield, 2006) and golden plover *Pluvialis apricaria* (Pearce-Higgins et al., 2008) in an attempt to guide planning of wind farm developments. Individual based models have also been used to predict the effects of developments on birds (West and Caldow, 2006). Caution should be taken however when assessing the indirect effects of wind farms as it is possible that the behaviour is not temporally stable and that the birds may habituate to the turbines. A recent aerial survey in Denmark reported sea ducks (common scoter *Melanitta nigra*) within the area of a wind farm, a species that had shown avoidance immediately after construction (A.D. Fox pers. comm.) and Madsen and Boertmann (2008) documented habituation of spring-staging geese to wind turbines over a period of 8-10 years. Equally, it should not be discounted that the construction of a wind farm may provide new and novel habitats for birds and their prey i.e. the wind turbines acting as artificial reefs (Petersen and Malm, 2006) so the abundance of both may increase within the area of the wind farm. This last aspect, habitat gain, is one that may only become evident some years after site construction therefore the evidence base for these effects remains poor (Inger et al., 2009).

1.2.3 Behavioural responses

Behavioural responses of birds towards wind farm developments (i.e. avoidance) means that wind farms effectively become barriers to movement. Desholm and Kahlert (2005) used surveillance radar to monitor movements of sea ducks before and after the construction of an offshore wind farm in the Baltic Sea and documented an avoidance response that was greater during the day than at night. Larsen and Guillemette (2007) also observed that common eiders *Somateria mollissima* avoided flying close to or within the Tunø Knob offshore wind farm. The consequence of avoidance is that birds have to fly increasing distances around turbines and incur an energetic cost. It is not known how this additional cost impacts the individual or the population, however, with increasing numbers of wind farms it is likely that the cost incurred will increase.

1.3 Cumulative impacts

Increasing numbers of wind farms seem to be inevitable given the international legal responsibility to reduce CO_2 emissions. With increasing numbers of wind farms comes concern over cumulative impacts but despite an awareness of the issue, there is a lack of understanding of cumulative impacts of wind farms on bird demography and populations; a lack of understanding of cumulative effects becomes increasingly unacceptable as the numbers of wind farms increase on the land/seascape. Trivial impacts at single sites may no longer be trivial at a landscape scale. For example, home ranges of eagles may cease to provide viable foraging habitat if several wind farms block access to feeding areas of birds constrained to be central place foragers (at least while breeding). Although individual wind farm developments in Scotland are subject to detailed Environmental Impact Assessment, the requirement to consider the cumulative impact of a series of individual developments is vague and often evaded. There is a clear need, not only in Scotland, but also globally, for an understanding of cumulative impacts of wind farms on bird populations and so far the focus has been on individual developments and impacts.

1.4 Thesis development

Within Scotland, the authority responsible for allowing a wind farm proposal to proceed is Scottish Natural Heritage (SNH). Therefore, with a clear need for better understanding of both the individual and cumulative impacts of wind farms, SNH provided a PhD studentship for a research project ‘Assessing the cumulative impacts of wind farms on birds’.

After attending many workshops and meetings organised to address the topic of the cumulative impacts of wind farms, I came to realise that a major obstacle to progress was the lack of a common language between all interested parties. ‘Cumulative impacts’ is an ambiguous term and on posing the question, ‘What do you mean by cumulative impacts?’ I would receive differing replies from individuals in a room. As a consequence, in **chapter 2** I develop a conceptual framework in which to consider cumulative impacts.

Although much research is now underway to assess the effects of wind farms on birds, few studies have attempted to link the effects for example, habitat loss, to individual or population impacts for example, of changes in population growth. In **chapter 3** I ask one such question for a population of common eider that interacts with a wind farm on migration from Finland to the Wadden Sea. These birds show avoidance to wind turbines and thus travel further. Analysing flight trajectories recorded using surveillance radar I investigate the impact in terms of loss of body mass, of travelling the additional distance, and also the cumulative impact of many such wind farms.

Migrating birds may only interact with a wind farm twice a year and therefore the impact may be trivial, however an obvious progression was to assess the impacts for commuting birds i.e. birds that undertake several foraging trips a day. Hence, in **chapter 4** I investigate the cumulative impacts of wind farms as barriers to movement for a suite of breeding seabirds of differing morphologies, using an energetic modelling approach.

The radar data analysed in chapter 3 are fine-scale movements of birds around a wind farm and I realised that it might be possible to construct a model to quantitatively describe the movement process. Such a model could benefit the planning stages of wind farms and minimise the impacts on birds, as well as providing me with the opportunity to learn movement modelling techniques. This led to what is now affectionately known as 'the automatic duck model' or **chapter 5**.

Most wind farm developments in Scotland have been onshore projects therefore it became clear that SNH interests lay within terrestrial wind farms and the impacts on birds, much more than marine wind farms. This stance has now altered with increasing interest in marine renewable energy developments but **chapter 6** grew out of their original interest in terrestrial wind farms and the idea that rather than studying the individual impacts of a wind farm in isolation i.e. habitat loss, collision, or behavioural responses, we should be assessing these together. Therefore in chapter 6 I present a spatially-explicit individual-based model to conduct a population viability analysis for a population of hen harriers *Circus cyaneus* on Orkney in the presence of wind turbines. The hen harrier is a conservation priority species of interest to SNH and the population on Orkney is a well studied example. Therefore, being a relatively well-described system and an area where wind turbines are in operation and more have been approved for construction, it seemed an obvious place to start modelling the impacts of wind farms.

1.5 On the cutting room floor

The work presented in this thesis comprises projects that were completed; the pieces of work where the data and methods came together to provide results. However, there are others that did not come to fruition and are therefore absent. Here I outline two such projects which were started but subsequently abandoned.

1.5.1 Forestry as a proxy for wind farms

Among the birds highlighted as being particularly vulnerable to wind farm developments are the soaring raptors (Barrios and Rodriguez, 2004, Madders and Whitfield, 2006) and in the Scottish uplands these include the golden eagle. Golden eagles are territorial birds so may experience habitat loss and fragmentation reducing the area of home range suitable for foraging and potentially reducing breeding success. The loss of habitat expected with increasing wind farm developments in Scotland is similar to that seen from afforestation. There are limited data available regarding the interactions of raptors and wind farms in Scotland (Madders and Whitfield, 2006, Walker et al., 2005) however, extended monitoring of raptors by study groups and detailed forestry records can provide data to assess the impacts of forestry on raptors.

Previous studies have assessed the effects of forestry on golden eagles, analysing changes in breeding success in relation to forest cover and territory use (Whitfield et al., 2001, Whitfield et al., 2007a) but neither was able to quantify the effect due to high levels of variation in the results. A plausible explanation for their results could be that the sample size was small and had a restricted area with the studies analysing data from the southern half of mainland Argyll and Bute and the Isle of Mull respectively. Using a larger dataset may produce different trends. Therefore I proposed to closely follow the methods used in previous studies (Whitfield et al., 2007a, Whitfield et al., 2001) but use a larger dataset at the scale of the entire west of Scotland. Once I had analysed the relationship between area of afforestation and eagle breeding success, or change in area of afforestation and change in breeding success, I would use data on the behavioural avoidance of wind farms to assess the equivalent loss due to wind farms and the extent to which this could be mitigated by reduction in forest cover. In order to complete the project I required accurate data on the location of nest sites, and productivity of golden eagle pairs for the west of Scotland and also forest cover data. SNH had led me to believe that they were able to provide the data sets required for this analysis, but after some time indicated that I would need to request the data from each of the local Raptor Study Groups. Unfortunately, due to the sensitive nature of data on golden eagle nests, not all of the Scottish Raptor Study Groups would release the data and I could not follow this line of investigation any further.

1.5.2 Daily movements of common eiders

On completion of data analysis for chapter 3 it was apparent that the additional distances travelled by migrating common eider, and the associated loss in body mass, were trivial. It seemed likely that the impacts would be greater for birds passing a wind farm several times during a day. The original concept for chapter 4 was therefore to assess the daily movements of common eiders and in combination with the response data from chapter 3, hypothesise as to the potential impacts of wind farms for these birds. During winter, common eiders are present in the Clyde Sea Area and large flocks can be found by Rhu Narrows on Gare Loch. Therefore, using several co-ordinated observers, the aim was to visually track eiders flying within this study area to estimate the amount of time spent in flight during a day, and the distances flown. However, I had not anticipated that the birds simply would not fly! Within Gare Loch it appeared that the eiders preferred to stay on the water, even when disturbed, rather than to fly. Consequently, chapter 4 became an analysis of movement data extracted from the peer-reviewed literature.

2 Cumulative impact assessments and bird/wind farm interactions: developing a conceptual framework

Masden, E.A., Fox, A.D., Furness, R.W., Bullman, R., & Haydon, D.T. 2009 Cumulative impact assessments and bird/wind farm interactions: developing a conceptual framework. *Environmental Impact Assessment Review*. **30**: 1-7

2.1 Abstract

The wind power industry has grown rapidly in the UK to meet EU targets of sourcing 20% of energy from renewable sources by 2020. Although wind power is a renewable energy source, there are environmental concerns over increasing numbers of wind farm proposals and associated cumulative impacts. Individually, a wind farm, or indeed any action, may have minor effects on the environment, but collectively these may be significant, potentially greater than the sum of the individual parts acting alone. EU and UK legislation requires cumulative impact assessment (CIA) as part of Environmental Impact Assessments (EIA). However, in the absence of detailed guidance and definitions, such assessments within EIA are rarely adequate restricting the acquisition of basic knowledge about the cumulative impacts of wind farms on bird populations. Here we propose a conceptual framework to promote transparency in CIA through the explicit definition of impacts, actions and scales within an assessment. Our framework requires improved legislative guidance on the actions to include in assessments, and advice on the appropriate baselines against which to assess impacts. Cumulative impacts are currently considered on restricted scales (spatial and temporal) relating to individual development EIAs. We propose that benefits would be gained from elevating CIA to a strategic level, as a component of spatially explicit planning.

2.2 Introduction

Cumulative impacts originally gained status in the United States' National Environmental Policy Act (NEPA) and later incorporated into the Environmental Impact Assessment (EIA) Directive (85/337/EEC) of the European Community. Cumulative impact assessments provide information to inform the management of developments so resultant impacts do not exceed specified threshold levels (Canter and Kamath, 1995). Whilst all individual projects or actions affect their environment, the combined or cumulative effects of multiple actions can be greater than the sum of the individual parts (Canter and Kamath, 1995). Increasing numbers of proposed developments create greater pressures on the environment, making cumulative impacts a pressing issue. Such is the case for wind farms in the UK, where concerns have been raised over the negative impacts of increasing numbers of wind farms on bird populations (Stewart et al., 2007).

Although cumulative impacts are increasingly included within environmental impact assessments, the quality remains far from adequate (Piper, 2001). Most UK assessments fail to sufficiently incorporate cumulative impacts; only 48% of the statements reviewed by Cooper and Sheate (2002) mentioned the term 'cumulative impacts' and of those, only 18% provided a discussion on the topic. Explanations for the lack of consideration of cumulative impacts in EIA reports centre around the absence of guidance on the requirements of cumulative impact assessment and the lack of a comprehensive definition (Bérubé, 2007, Thatcher, 1990, Canter and Kamath, 1995).

Cumulative impact assessment remains a mystery to most EIA practitioners (Duinker and Greig, 2006, Smith, 2006) therefore changes are required in the way assessments are approached and delivered, if any utility is to be derived from the process. "We need revolution in how we undertake cumulative impact assessment, not evolution" (Duinker and Greig, 2006). One obvious barrier to effective assessment of cumulative impacts is the lack of clarity in discourse between the relevant parties i.e. developers, statutory bodies, non-governmental organisations (NGOs) and scientists. Practitioners remain uncertain about the legislative requirements and also the data required for assessments. Such confusion is not a problem unique to cumulative impact assessment. For example, in a review of ecosystem stability, Grimm and Wissel (1997) emphasised that although “human concepts can be signposts through the confusing complexity of nature” these concepts themselves can cause confusion. In this paper, we propose a conceptual framework to promote a more transparent and efficient cumulative impact assessment process, to further understanding of the impacts of wind farms on bird populations.

2.3 Legislative Background

Consideration of cumulative impacts is required under the EC Directive (85/337/EEC) on EIA, implemented in the UK *inter alia* under the Town and Country Planning (Environmental Impact Assessment) Regulations 1999. The regulations refer to cumulative impacts in the screening of projects (Schedule 3) “the characteristics of development must be considered having regard, in particular, …the cumulation with other developments” and in the inclusion of information in environmental statements (Schedule 4) “a description of the likely significant effects of the development on the environment, which should cover the cumulative effects of the development…” (Town and Country Planning Regulations Assessment of Environmental Effects Schedule 4 Part 1). These regulations in themselves create confusion: Schedule 3 defines cumulative impacts as an accumulation of impacts across developments, Schedule 4 refers to cumulative effects as effects that accumulate within a development over time.

The Habitats Directive (92/43/EEC) also refers to cumulative impacts and is implemented in the UK through the Conservation (Natural Habitats & c.) Regulations. The regulations state that where an Appropriate Assessment should be undertaken, “The effects considered should be those of the plan or project, either alone or in combination with other plans or projects already carried out or proposed, on the habitats and species of international importance…”. Cumulative effects also appear in the Strategic Environmental Assessment (SEA) Directive (2001/42/EC) on the assessment of certain plans and programmes on the environment. The Directive requires information to be provided on “the likely significant effects…including cumulative and synergistic effects…on the environment.”

Despite the Directives requiring assessment of cumulative effects, no appropriate definition of cumulative effects, or indeed guidance on methods of assessments, are provided in the legislation, creating an uncertain regulatory environment for practitioners. Documents discussing cumulative impacts or explicitly defining the term ‘cumulative impact’ are also rare (RPS, 2007) and the only available definition is found within the EU “Guidelines for the Assessment of Indirect and Cumulative Impacts as well as Impact Interactions” (Hyder, 1999). Cumulative impacts are defined as “Impacts that result from incremental changes caused by other past, present or reasonably foreseeable actions together with the project” (Hyder, 1999).

2.4 Deconstructing cumulative impacts

How can we improve on the current process of cumulative impact assessment? In this section we provide insight on the concepts (impact, actions, and scale) within Hyder’s (1999) cumulative impact definition and suggest how these ideas can direct data collection and analysis for a cumulative impact assessment.

2.4.1 Impact

A cumulative impact assessment is intended to estimate the impact of a planned action on a receptor, in combination with other actions. We define an environmental receptor as any ecological or other feature that is sensitive to, or has the potential to be affected by, an action. Of primary importance is the identification of environmental receptors at risk from the proposed action. For example, which guilds, species and/or individuals are to be considered in the assessment and why?

2.4.1.1 Species or guilds

An action could affect any species occurring in the impact area. When confronted with the decision to approve a wind farm location it may be necessary to consider the variation in effect dependent on species. Red-throated divers *Gavia stellata* and common scoter *Melanitta nigra* were found within the Horns Rev offshore wind farm area pre-construction but were almost totally absent post-construction (Petersen et al., 2006), although common scoters have since begun to forage between the turbines. In contrast, some long-tailed ducks *Clangula hyemalis* at Nysted offshore wind farm have occurred between the turbines, but at lower densities than prior to construction (Petersen et al., 2006). The extent and nature of effective habitat loss may therefore differ between species. Not only the behaviour, but also the ability to withstand effects may differ between species. Short-lived migratory species may show high reproductive capacity that buffers them, under normal circumstances, against the very high rates of mortality experienced during their annual migrations. Such species may be relatively robust to enhanced mortality levels in a way that is not the case for long-lived species, where even small increases in death rate will rapidly impact on population size because of low reproductive output. It is also important to consider ecosystem functioning and the trophic relationships between species, i.e. the processes and interactions that occur within an ecosystem. The presence or absence of a species, especially a top predator, may affect the abundance of their prey and ultimately the composition of the ecosystem (Mills et al., 1993).

Which species should be considered? Ideally, a broad range of species would be included in a cumulative impact assessment but rarely is it logistically or financially viable to consider all species occurring within a region. Species selection requires value to be placed on the environment and the receptors within. Value can be assigned to species by various methods (Turner et al., 2003, Bandara and Tisdell, 2005, Ekins, 2003, Patterson, 2002). However, the EU Directive on the Conservation of Wild Bird Species (EC Birds Directive 79/409/EEC) already assigns value to bird species via its species Annexes, e.g. Annex I lists critical species subject to special conservation measures whilst species in Annex II and III can be hunted. All species on the Birds Directive Annex I must be included in a cumulative impact assessment, as they are considered of particular value and awarded the greatest level of protection. We recommend that the list of species should not end with Annex I. Other species that practitioners should include in assessments are those for which the area is important for a specific life stage, whose characteristics make them especially vulnerable i.e. flying at turbine height, are named in the citation of adjacent protected areas or have low reproductive output (King et al., 2009).

2.4.1.2 Individuals or stages

An action can potentially affect a single individual or an entire population, dependent on the ecology of the species. For example golden eagles (*Aquila chrysaetos*) are largely sedentary, territorial birds so a development may only affect the pair whose territory encompasses the wind farm. Alternatively, a development may affect an entire population. Between 200 000 and 300 000 migrating common eiders *Somateria mollissima* (breeding in Estonia, Finland and eastern Sweden) may interact with the Nysted wind farm off the Danish coast during passage to and from their wintering grounds in the Wadden Sea (Petersen et al., 2006). Individuals of the same species may also represent different values; death of a territorial breeding adult of high quality may have a greater direct impact than the loss of a sub-adult that lacks the capacity to breed in a territorial population. Between individuals, the level of the effect may also vary as a function of state (starving versus satiated) (Kaiser et al., 2006) and personality (risk averse versus risk prone).

For a comprehensive assessment, all individuals within a population, at all stages within the lifecycle should be considered. However, resources available for assessments are often limiting so a comprehensive assessment is not always possible. In these situations we recommend that only the stages and individuals most likely affected should be included. To make consistent decisions on the stages/individuals to include requires a repeatable design protocol, which practitioners can follow but such protocols are not available. Therefore we suggest that consistency and repeatability would be enhanced by the development of a standard design protocol for deciding appropriately representative receptors to include in cumulative impact assessments.

2.4.1.3 Processes

Impact is often assumed to be synonymous with effect but the two have distinct meanings. An impact is the ultimate change due to an effect, with the effect being the proximate response of an individual to an action. Fox et al. (2006) highlighted the ways in which processes such as habitat use, can be affected in the wind farm context. Birds colliding with turbines represent a direct impact on population size (through additional mortality) but what of other effects? For example, birds may avoid the immediate vicinity of a wind farm post construction, where the effect is displacement from feeding habitat, resulting in effective habitat loss. The impact of the wind farm may thus be a reduction in local abundance by displacement of individuals to other areas. However, these birds may be displaced to already occupied or otherwise unsuitable habitat elsewhere, and this displacement may cause loss of condition amongst these individuals, reductions in reproductive output or even reductions in survival. Equally, a wind farm may be perceived by a bird as a barrier, necessitating additional flight to avoid the obstacle, thus causing the bird to expend excess energy, again potentially affecting its breeding success and survival. In this way an effect (avoidance response) ultimately contributes to an impact (reduced population size), suggesting changes in population abundance as a potential common currency or metric for impact assessment. The challenge is to assess these indirect effects along with the direct impacts and the difficulty lies in translating an effect, or cumulative effects, into their ultimate impacts. But this is a difficulty pervasive to almost all environmental science: how does physical environment influence population abundance? Our quantitative understanding of this link is often poor, and while we lack a currency to compare what are essentially chalk and cheese, combining impacts and effects in realistic cumulative impact assessments will remain a serious problem.

Theoretically abundance is an ideal metric but it is also often difficult to measure with sufficient accuracy to detect statistically significant changes before there is a real probability of a substantial ecological change. Consequently, it may be more practical to measure the effects of an action on a process. Effects are more easily detected and quantified than impacts, but it requires an understanding of how the processes are ultimately linked if the impact is to be estimated through the application of population modelling. For all species, the causal linkages between actions such as wind farms, population processes (effects) and changes in abundance (impacts) are currently unknown and a better basic understanding of these links remains a fundamental challenge for ecologists.

In contrast, estimates of collision risk can be quite robust. In this respect, the Baltic/Wadden Sea population of common eider provides an example of cumulative processes. Out of 235 000 passing eiders at Nysted, modelling showed with 95% certainty that 0.018-0.020% would collide with the turbines (less than 1 bird/turbine/year) (Desholm, 2006). Therefore the wind farm has an impact by directly adding to mortality rate, to a degree which can be predicted with confidence and verified by remote sensing. However, eiders are known to avoid wind farms, incurring an additional energetic cost to navigating around the turbines (Desholm and Kahlert, 2005, Masden et al., 2009) that may affect breeding condition and consequently affect the reproductive output of the population. Both the direct mortality and the results of indirect energetic costs will impact the population, but in this case, whilst the former is well documented, the population impacts of additional energy costs incurred by avoidance remain unknown. With such unknowns, the problem of assessing cumulative impacts as defined by Hyder (1999) is intractable and would require individual-based modelling at the scale of the flyway population along with knowledge of avian ecology, energetics and food resources that is not available at present, nor likely to be in the near future, in order to link the effects with population impacts. This problem is not one solely of cumulative impact assessment but of ecology in general, however, it does mean that there is a discrepancy between the data and knowledge required for a comprehensive cumulative impact assessment, and that which is available.

One solution would be to concentrate on a restricted number of processes. Different processes influence changes in population size to a greater or lesser extent, and the identification of these processes is important for effective population management (Benton and Grant, 1999). It is possible to predict the processes most likely affected by an action, based on the ecology of a species, for example eiders generally fly below rotor height and strongly avoid wind farms (Desholm and Kahlert, 2005) so are less likely to be affected by collision than by increased energetic costs. For some species the importance of different processes has already been established (RPS, 2007).

Predictions about the future impacts of wind farms on birds requires prior knowledge of the effects on processes, but this evidence-based approach is generally absent from studies which are often methodologically weak with few long term impact assessments (Stewart et al., 2007). We recommend the Before-After Control-Impact (BACI) design as an ideal framework upon which to base data collection before and after the construction of a development in order to understand the effect of an action upon a receptor. Although we recognise that the BACI design is not flawless, with potential for dispute over the comparability of control and impact sites, it is nonetheless the best method currently available, and a considerable improvement on many current practices. We also suggest that the assessment of cumulative impacts would benefit from increased availability of post-construction monitoring data, therefore more stringent regulations are required on the collection and dissemination of such data (Langston et al., 2006).

2.4.2 Actions

An action is any event that perturbs a receptor with a resultant effect. A cumulative impact can therefore be thought of as the impact associated with increasing numbers of actions and their resultant effects. According to Hyder's (1999) definition, a thorough cumulative impact assessment should be exhaustive and include all actions affecting a receptor. For example, when assessing the additional mortality incurred by a population of small passerines due to a wind farm, the list of other actions to be considered would include overhead power lines, tall buildings, windows, cars, cats, storms, etc. The actions may be homotypic or heterotypic (Irving et al., 1986) and may, or may not, have a specific consenting process (RPS, 2007). Actions such as climate change have no specific consenting process, but impact on a receptor. Such actions might then be considered background sources of impact nonetheless. Although the effects and impacts of these actions may be more difficult to assess due to the lack of a definite location of the action, it remains important to include them in assessments as they contribute to cumulative impacts, according to Hyder (1999). Inclusion of climate change in an assessment also allows the impact associated with other actions i.e. wind farms, to be viewed in the context of climate change (Stewart et al., 2007). Therefore, with the inclusion of all actions, it is possible to make comparisons between the impacts of different actions. For example, after a comparison of different actions affecting seabird populations, Wilcox and Donlan (2007) suggested that the removal of invasive predators from breeding islands would be a more effective means of increasing seabird population growth rate per dollar than fisheries closures and by catch reduction strategies. It may therefore be possible to compare the relative impacts of actions, for example a comparison between the relative impacts of hunting mortality, that of a wind farm, and of climate change on the Baltic/Wadden Sea population of common eiders.

Which actions should be included in a cumulative impact assessment? A comprehensive assessment should include all actions, past, present and future, with future being defined as those actions in planning when considering consented projects, and reasonable projections for non-consented actions such as fishing activity or climate change. Climate change is an action often excluded from assessments on the basis that it is impossible to disentangle the effects of human actions against those of climate change due to the variability and uncertainty linked with climate change. However, if climate change itself is considered an action, then the associated variability can be explained. For example, Rolland et al. (2008) assessed the combined effects of fisheries and climate on the endangered black-browed albatross, *Thalassarche melanophris*, concluding that the population dynamics were affected both by climatic conditions and fisheries. Although the Hyder (1999) definition dictates that all actions potentially affecting a receptor should be exhaustively included within a cumulative impact assessment, such an assessment is often logistically impractical. In these situations, the question of which actions to include in an assessment remains unanswered and decisions have to be made on the basis of expert opinion. Due to the necessity of expert evaluation and with the aim of consistent decision making, we propose that cumulative assessments would be better tackled once at a strategic level, rather than many times by different practitioners for individual project-based EIAs.

2.4.3 Scale

Not only is it necessary for types of actions to be defined within an assessment but also, the specific identity of actions. Therefore, it is crucial that the boundaries of space and time be defined so that actions can be identified and any scale effects can be ascertained (Canter and Kamath, 1995, Burris and Canter, 1997, João, 2002, Stewart et al., 2007).

2.4.3.1 Space

It is fundamental to determine the area to be included in an analysis and it must be large enough to cover the processes likely affected (Krebs, 2002). If an action affects a whole population, including only a sub-sample of the population in the assessment will not estimate the true effect. For example, post-construction of a wind farm, mortality of a receptor may increase due to collisions with the turbines. Consequently, the global population may be reduced but if new individuals move into the area due to a released constraint of density dependence, the local population may appear the same with the local area acting as a population sink. Considering only the local population, in this case, would underestimate the extent of the impact. Conversely, local sub-populations may be affected by different actions and this should be allowed for in the assessment of impact at the global scale. For example, a widespread species such as the chaffinch *Fringilla coelebs* migrates in a broad front rather than on a specific route. A single wind farm will therefore only affect a restricted portion of the population, and multiple wind farms will affect a different set of birds in turn. Contrast this with the same set of wind farms but located along a migration corridor; all wind farms now affect the same set of birds. Space use and the spatial scale at which the receptor is considered (local population or global population) are vital to the accurate assessment of impacts.

Another consideration is that although a receptor may not be present in the immediate vicinity of the action year-round, it may be linked to the action during discrete life stages. An example of this is the interaction of eiders with the Danish wind farm, Nysted. If the effects of the wind farm are only assessed in Danish waters then the receptor will be defined as the Danish population of eiders. However, the Wadden Sea/Baltic population migrate through the area of the wind farm twice a year, and therefore actions that affect the population along the flyway should also be included in the assessment. When considering larger spatial scales it may however be problematic, because species often move across international boundaries; it therefore requires cooperation to assess all of the actions that affect these populations.

If a species of concern is using the area around an action for any period of time then the ideal spatial scale for an assessment would be the area used by the global population of the species of interest. Thus the extent would include all actions that the species would interact with during all stages of a life cycle. Although the spatial extent of assessment may be the global range of a species, the main areas of interest are those of past, present or future actions.

Therefore, during data collection, field effort will be concentrated around these areas. Spatial boundaries therefore need to be defined at these smaller action-based scales, for example, when assessing collision mortality. If a bird collides with a wind turbine it can either be killed instantaneously or injured. If killed, it will drop to the ground in the vicinity of the turbine, however, if injured the bird may die some distance from the turbine. Consequently, the estimated mortality rate will change with the area included in the corpse search around the turbine. The greater the area included in the search, the more birds are likely to be found, however, the cause of death becomes less certain and difficult to verify as the distance from the turbine increases. Accuracy of assessments at the local action-based scale is vital to the accuracy of cumulative impact assessments at the global scale. For a given cumulative impact assessment, it may be sufficient to consider only the current range of a species however under certain circumstances additional areas may need to be incorporated. For example, the extent to which habitat loss will impact a species is dependent on the availability of suitable, but currently unutilised habitat. If a golden eagle territory is bounded by another, the ability of the eagle to expand its range in the face of reduced habitat will be constrained (McGrady et al., 1997, Whitfield et al., 2007a).

The ideal spatial scale of assessment may be the global range of a population but if large-scale data collection and analysis proves impossible, we recommend the use of smaller bio-geographic units. One such unit, if considering terrestrial birds in Scotland would be the natural heritage zones used by Scottish Natural Heritage (Whitfield et al., 2007b). A similar unit for the marine habitat of the North Atlantic could be ICES sea areas. Although often arbitrary, such units are already well established and may have associated data archives. However, as mentioned in Section 2.4.1.2, there is a lack of standard protocols for decision making on matters such as selecting appropriate scales of assessment therefore we highlight this as a target for effort in the future.

2.4.3.2 Time

The temporal boundaries of a cumulative impact assessment must be appropriate both for the processes likely to be affected and also the species ecology. Temporal scale should be considered because the effect of an action may show a temporal trend with a population more susceptible at specific times of the year, for example during the breeding season or the over-wintering period. This can be seen in the Nysted wind farm example of 3.3.1 with eiders being affected by collision mortality from Nysted only during their annual migration, i.e. twice a year. Another reason to consider the temporal scale of assessment is that effects are not always realised immediately, leading to delayed temporal variation. One potential cause of such a lag is breeding biology and the age of maturity of individuals with effects only being realised once individuals have entered the breeding population. For example, golden eagles do not secure a mate and enter the breeding population until 4 or 5 years old (Watson, 1997) therefore effects may be unobserved for at least 5 years. Serious consideration should be given to the inclusion of such lags when measuring effects because there is the potential for impacts to remain undetected (RPS, 2007).

‘The identification of the effects of past actions is critical to understanding the environmental condition of the area. ” (US Council on Environmental Quality, 1997). When setting the baseline against which to assess impacts a practitioner should consider whether the current condition is an adequate representation of the non-effected environment and if not, what data should be included to allow the differentiation between noise within the system and an impact due to the action. Assuming data availability, McCold and Saulsbury (1996) advocate, “The appropriate baseline for considering the significance of cumulative impacts is the time when the valued environmental component was most abundant”, though this may not always be true. The population of northern fulmar, *Fulmarus glacialis* around the British Isles provides one such example. This population has increased both its range and abundance since the mid-18th century (Mitchell et al., 2004). Availability of offal and discards from commercial fisheries has been implicated as a contributing factor to the growth in numbers and distribution, therefore it may not be fitting to use these elevated abundances as a baseline value because the increase is a function of anthropogenic activities (Mitchell et al., 2004). Another view is that the baseline should be the most recent state of the receptor. Although this may not be the true and naturally occurring state of the receptor, it is argued that it is the most feasible to assess. The problem arises when no standard baseline measure exists for a receptor, but rather the baseline is considered the state of the receptor at the time of individual assessments (IEEM, 2008). This lack of historical data integration is known as the “shifting baselines” syndrome (Pauly, 1995) and over time can lead to the degradation of a receptor. However, very rarely are sufficient time-series data available to adequately assign baselines. Therefore compromises have to be made, for instance the Ramsar Convention on Wetlands designations, adopts the last five years of reliable data to determine the benchmark/baseline.

Temporal variation may also occur over the lifetime of the action because the behaviour of the receptor changes in response to the action; birds may initially exhibit avoidance behaviour towards wind turbines but over time the response may change. At the Danish wind farm, Horns Rev, red-throated divers and common scoters were found within the wind farm area pre-construction but were almost totally absent immediately post-construction. Five years after construction, scoters have now moved back within the wind farm and occur at similar densities inside the wind farm area to outside, yet divers continue to be absent (Ib Krag Petersen pers. comm.). Similarly, habituation has been observed in pink-footed geese, *Anser brachyrhynchus* at terrestrial wind farms in Denmark (Madsen and Boertmann, 2008).

The temporal scale of an assessment should be defined in terms of the available baseline data, the species ecology, and also the lifetime of the actions of interest. The data collection and assessment should include seasons relevant to the environmental receptor of interest and the analysis of the data should allow for any potential time lags in effects. For example, an assessment for a wind farm (operational lifespan of 15 years) affecting golden eagles (breeding age of 5 years), may include a predictive model that has a temporal scale of 20 years. The concept of time lags is also important when designing post-construction monitoring for an action. If temporal lags in effects are expected then the scale of the monitoring program should encompass these lags and continue long enough to assess whether predicted impacts have been realised. The baseline data against which to compare these impacts should not be the state of the receptor at the time of assessment and data collection as over time, this will lead to shifting baseline syndrome (Pauly, 1995). Instead, it should include a series of data long enough to detect underlying variability in the system, against which to compare effects and impacts caused by the action. There is a lack of guidance available for practitioners on choosing appropriate baselines and so with each practitioner independently deciding on an appropriate baseline, the process is rather ad hoc. We therefore suggest that if all assessments are to be comparable and free from shifting baseline syndrome it requires a strategic decision to be made at the policy level about the value of species, appropriate baseline levels and acceptable target population sizes.

2.5 Formalising the framework

Throughout this paper we have described ways to consider the assessment of cumulative impacts and the concepts within the Hyder (1999) definition. In this section we present our discussion in terms of a formalised equation in an effort to further clarify thinking on the matter. Models and formulae are often useful as a tool to simplify concepts and identify the essential elements of a problem, in an effort to find solutions. Although simplistic, our formulation further highlights the different elements that should be incorporated into a cumulative assessment. As Box and Draper (1987) stated, “All models are wrong, but some are useful.”

2.5.1 The framework as a function

A function is a mathematical concept that describes the relationship between variables, such as abundance of golden eagles and number of wind farms. Therefore, the impact of the *i*th action (A_i), for example a wind farm, on the *j*th receptor (R_j), for example a golden eagle population, at location **x**, at time *t* can be defined by a function (I)

$$I\ (A_i, R_j, \mathbf{x}, t) \quad (1)$$

Having defined the impact of a specified action on a specified receptor at a particular space-time location, it is now possible to consider the cumulative impact of a set of actions (A), for example wind farms, forestry and persecution, on a set of receptors (R), for example adult and juvenile golden eagles, over a set of locations (Ω), accumulated over a defined time period comprising the past, present, and future. The cumulative impact (CI) can be expressed as a multiple integral of the impact function (I):

$$CI = \sum_{i \in \mathrm{A}} \sum_{j \in R} \int_{\mathbf{x} \in \Omega} \left(\int_{t_0}^{0} I(A_i, R_j, \mathbf{x}, t)dt + \int_{0}^{t_1} I(A_i, R_j, \mathbf{x}, t)dt \right) d\mathbf{x}$$

(2)

2.5.2 Defining the sets

In equation 2, actions (A_i) and receptors (R_j) are discrete values taken from sets, A and R, respectively. These sets should be selected, as discussed in section 2.4, to ensure inclusion of all relevant actions and receptors. Space ($\mathbf{x}$) is represented as a vector of locations (x, y) in a 2-dimentional plane ($\mathbf{R}^2$) within a bounded area or set (Ω), however space may also be represented as a set of discrete sampling points. Time (t) is divided into two periods, past (t_0) to present (0) and present to some defined point in the future (0 - t_1). Inclusion of past impacts prevents temporal creep from adopted baseline standards as discussed in section 2.3.3.2. The projection into the future should be made over appropriate time horizons based on the operational life-span of the action and the receptor ecology (section 2.4.3.2).

2.5.3 Assumptions

For clarity of presentation, a simplifying assumption of equation 2 is that the impacts are additive, with no interactions between receptors and/or actions. This is certainly violated in most situations. When considering the receptor for example, the behaviour of individual birds towards a wind farm may not be independent. Many species exhibit flocking behaviour, so the response of many individuals may be dependent on that of a few key individuals. It is also possible that for effects such as disturbance, the response is likely to be non-linear with threshold characteristics. A small disturbance may have limited impact but a more extensive or prolonged disturbance event may have a disproportionally large impact. Interactions are also possible between actions. It has been suggested that wind turbine structures in the marine environment may provide habitat for some life stages such as juvenile fishes. Furthermore, it is possible that no-take marine protected areas could be developed in association with the footprints of offshore wind farms, thus positively affecting species which are negatively affected by other actions elsewhere (Linley et al., 2007). Simplicity of presentation dictated the exclusion of interaction terms from equation 2 however these terms should be introduced where required.

2.6 Conclusion

"The natural world is in crisis; wild living resources are being depleted at increasing rates, the ecosystems upon which they depend are generally perturbed, and the consumption of resources by a growing human population generally increases" (Mangel et al., 1996). As human actions increasingly influence the environment, it is important to monitor and assess these anthropogenic-induced changes. Increasing numbers of wind farms seem to be inevitable given the international legal responsibility to reduce CO_2 emissions but there remains much concern over the impacts on bird populations. With increasing numbers of wind farms comes concern not only over isolated environmental effects but also the cumulative environmental impacts and despite awareness of the issue, there seems to be a lack of understanding and research in the area of cumulative impact assessment.

The cumulative impact assessment process is inadequate and unsatisfactory with few EIAs even considering cumulative impacts. Bad practice is not restricted to the UK, but widespread across Europe and North America (Burris and Canter, 1997, Wärnbäck and Hilding-Rydevik, 2008, Duinker and Greig, 2006). The absence of effective assessments of cumulative impacts is a function of the current lack of guidance (Cooper and Sheate, 2002), and particularly the absence of a comprehensive definition. Without a clear definition it is not possible to ensure an assessment that demonstrates adequate consideration of all aspects of the ecosystem including spatial and temporal scale. Therefore there is an urgent need for legislation and statutory authorities to offer clarity on the requirements of cumulative assessment. Similarly, without explicit statements of which components have been considered in a cumulative assessment, it is difficult to draw conclusions from the data. The framework we suggest provides a means by which to explicitly highlight and include actions, impacts and scales in any cumulative impact assessment. By explicitly stating the actions and receptors included (or more importantly, those not included) in an assessment, and the scales at which these have been considered, it is possible to reduce uncertainty surrounding the assessment. If data collection has to be compromised i.e. the spatial scale reduced from global to local, due to limited financial resources, it can be identified using the framework we describe. However, practitioners are still lacking a means by which to make consistent decisions on the reduced sets of actions and receptors to include, and the scales at which to consider them, in a cumulative impact assessment. Until a standard method is devised it is unlikely that cumulative impact assessments will provide any more value than at present.

A comprehensive cumulative impact assessment relies on the availability of data for actions. In a competitive business such as energy supply, acquiring information from other developers about potential actions, sufficient to conduct a thorough cumulative assessment is difficult, if not impossible. Ludwig et al. (2001) suggested, “Wicked problems, such as the planning of wind farms, require innovative policy but also innovative methods of arriving at the policy”. We propose that the innovation required is the elevation of cumulative impact assessment from the individual project to the strategic level. Under the EIA Directive, cumulative impact assessments are conducted at a project level by developers; elevating the process to a more strategic level may relieve some of the problems of data availability and confidentiality, with an assessment being the responsibility of a regulatory body rather than the individual project developer. With a more strategic approach, greater data acquisition would also be possible, as resources would be pooled for one assessment rather than for many. Strategic assessments already occur within the EU in the form of the strategic environmental assessment (SEA) and for offshore wind farm developments, the SEA is intended to inform cumulative impact assessments. Therefore, the infrastructure is more readily available and would only need modification. It has been suggested that when capability and resources for assessing cumulative impacts are limited, a greater proportion of effort should be assigned to minimise the impacts of single actions (MacDonald, 2000). The recommended shift in policy would see cumulative impact assessment integrated into strategic planning levels as part of the process of spatially explicit planning, making available the resources of developers to minimise the impacts of single actions through environmental impact assessments.

3 Barriers to movement: Impacts of wind farms on migrating birds

Masden, E.A., Haydon, D.T., Fox, A.D, Furness, R.W., Bullman, R. & Desholm, M. 2009. Barriers to Movement: Impacts of wind farms on migrating birds. *ICES Journal of Marine Science*. **66**: 746-753

3.1 Abstract

Advances in technology and engineering are enhancing the contribution that wind power makes to renewable energy generation. Wind farms, both operational and in planning, can be expected to negatively impact on wildlife populations, particularly birds. We propose a novel approach to assess the impacts through energetic costs of avoidance behaviour for a long-distance migratory seaduck. Flight trajectories were recorded using surveillance radar at a Danish offshore wind farm with emphasis placed on the 200 000+ migrating common eiders that pass through the area annually. Minimum distance to wind farm and curvature of trajectories were compared pre- and post-construction. Additional costs of the avoidance response were estimated using an avian energetics model. The curvature of eider trajectories was greatest post-construction and within 500 m of the wind farm, with a median curvature significantly greater than pre-construction suggesting birds adjusted their flight paths in the presence of the wind farm. Additional distance travelled due to the wind farm was c.400 m and trivial compared to the total costs of a 1400 km migration episode. However, construction of further wind farms along the migration route could have cumulative effects on the population, especially when considered in combination with other human actions.

3.2 Introduction

To curb climate change, governments are seeking to enhance the proportion of energy generated from renewable resources. Advances in technology and engineering realistically enable wind energy to form a significant proportion of this contribution (Larsson, 1994). More than 13 000 offshore wind turbines have been proposed in European waters (Desholm and Kahlert, 2005) with the UK government recently announcing an expansion of their wind energy programme, proposing 7 000 turbines to be built off the UK coast by 2020. However, wind farm developments are likely to impact negatively on the distribution and abundance of wildlife populations, particularly birds. Potential impacts of wind farms on bird populations can be categorised into three types: direct mortality of individuals due to collision with turbines and infrastructure; physical habitat modification due to the footprint of the turbines and associated structures; and avoidance responses of birds to turbines (Fielding et al., 2006, Fox et al., 2006). The latter includes both displacement from habitat and extended flights, where wind farms act as barriers to movement.

Studies have concentrated on collision mortality (Barrios and Rodriguez, 2004, Hötker et al., 2006) and habitat loss, either direct (Bright et al., 2006, Fielding et al., 2006) or effective, through avoidance behaviour (Larsen and Guillemette, 2007). Although the problem has been identified, researchers have not yet evaluated wind farms as barriers to movement (Madders and Whitfield, 2006, Fox et al., 2006, Langston and Pullan, 2003) and there is no standard methodology to tackle this issue. Animals often respond to spatial heterogeneity by altering their movement patterns (Frair et al., 2005) particularly in relation to novel objects (Jander, 1975). Sea ducks, particularly common eiders (*Somateria mollissima* L.) exhibit behavioural avoidance responses to wind farms (Desholm and Kahlert, 2005, Larsen and Guillemette, 2007); hence construction of wind farms along the flyway is likely to affect eider populations by increasing the distances travelled and the energy required to detour around these barriers. In many bird species, reproductive success is related to body condition at the time of breeding (Wendeln and Becker, 1999) especially amongst eiders because of the high investment of female body stores in reproduction (Meijer and Drent, 1999, Parker and Holm, 1990). Any reduction in mass due to increased flight requirements could be detrimental and directly impact the breeding output.

Common eiders are abundant throughout the Baltic, but the population is adversely affected by many human actions (e.g. fishing, eutrophication, and hunting). The Baltic Sea population of eiders decreased by 30-40% between 1991 and 2000 (Desholm et al., 2002) and this sea duck has also been highlighted as a species sensitive to climate change (Huntley et al., 2007). The Birds Directive and other international agreements require states to maintain bird populations and this necessitates understanding of the processes and pressures acting on a population. The cumulative impact of all pressures on a population may be negative but the challenge is to understand the impact of each pressure in isolation.

This study develops an approach to evaluate barrier effects associated with wind farms and uses this to assess the impact of the Nysted wind farm on the common eider. The following questions were addressed:

1. Do common eiders avoid the Nysted wind farm and at what distance?
2. Do common eiders increase their migration distance in the presence of the wind farm?
3. What is the cost of additional flight in the context of common eider seasonal migration and from the likely construction of many more marine wind farms?

3.3 Methods

3.3.1 Study site and species

The study area covered the Nysted offshore wind farm, in the western Baltic south of Denmark, comprising 72 turbines placed in eight north-south oriented rows, 850 metres apart at 480 metre intervals east-west, covering an area of c.60 km^2 (Figure 3.1). Flight trajectories of migrating waterbirds were collected between September 2000 and October 2005 using surveillance radar mounted on an observation tower located northeast of the wind farm (Petersen et al., 2006). Echoes from fixed targets were not displaced between the sweeps of the radar, and so it was concluded that the spatial movements of birds had been monitored precisely without displacement. Each flock of birds entering the detection area created an echo on the radar monitor, so by monitoring the movement of echoes, the migration trajectory of any given flock could be monitored. During daylight hours, trajectories were identified to species level, out to a range of c.11 km (Desholm and Kahlert, 2005, Desholm, 2003). All species trajectories were recorded but here we focus on common eider and make comparisons with all other trajectories gathered for waterbirds collectively. We present only data from the autumn migration as these were of the greatest quality but assume that the response to the wind farm and the associated energetic cost will be comparable during the spring migration. The Finnish population of common eider is likely to be affected by the Nysted wind farm because their migration route takes them from wintering areas in the Wadden Sea, to breeding areas in the Finnish Baltic, via southern Denmark. Between 200 000 to 300 000 common eiders pass the study site each spring and autumn (Alerstam et al., 1974, Petersen et al., 2006).

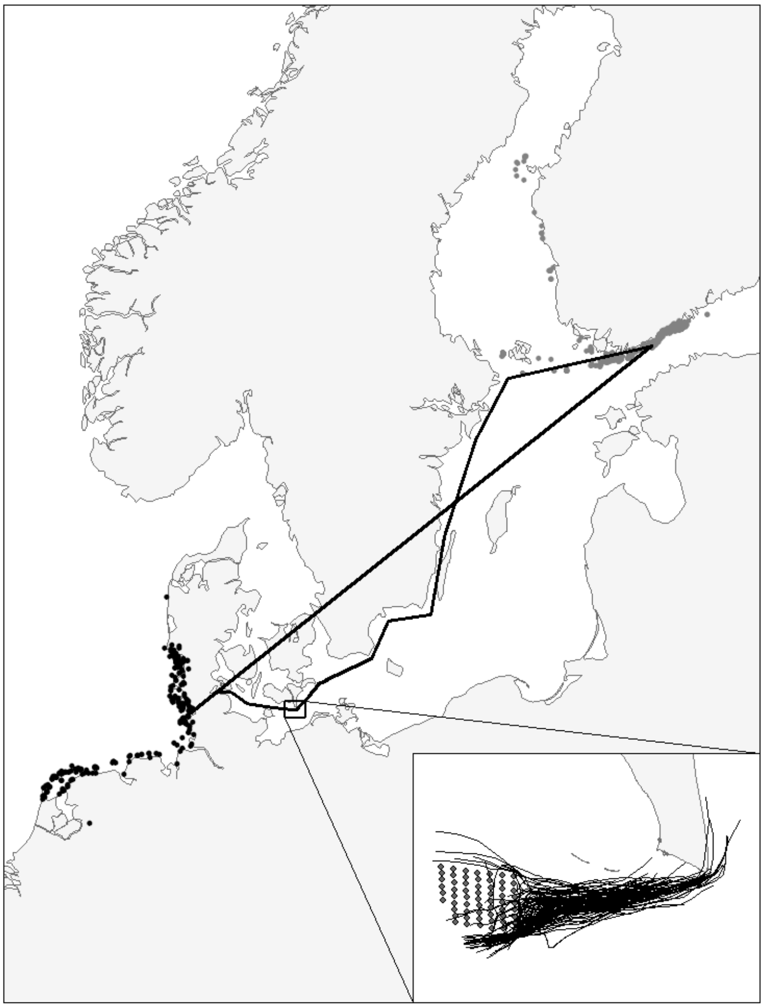

Figure 3.1 Estimated migration routes taken by eiders in comparison to the corresponding straight line distance. Grey circles represent capture sites of breeding adult females in Finland, and black circles correspond to the recovery sites of these Finnish-marked birds in winter in the Wadden Sea. Insert denotes study site and an example of eider trajectories (post-near).

3.3.2 Data Analyses

Deviation from a straight line trajectory, or curvature, was estimated to assess the additional distance travelled by individuals due to the presence of the wind farm. The measure of curvature is similar to the modified index of straightness (Batschelet, 1981), the difference being that the modified index takes a value between zero and one (Batschelet, 1981), whereas curvature can be any value greater than or equal to one.

Curvature = Length of trajectory / Euclidian distance from start to end point

Desholm (2003) used a similar method to assess how small changes in flight direction affected migration distance. For each trajectory, curvature was calculated from the beginning to the end of the trajectory. The minimum distance to the wind farm area was also estimated for each trajectory as a measure of the avoidance response.

Trajectories were categorised into those recorded pre- and post-construction and then further categorised as near or far from the wind farm; 500 metres was considered an appropriate distance threshold to differentiate between near and far because the distance between turbines in a row was approximately 500 metres. Larsen and Guillemette (2007) reported avoidance of eiders at 200 metres therefore it was reasonable to set the threshold greater than this. Visual examination of the data suggested that the curvature of trajectories did not vary greatly beyond 500 metres from any one turbine (Figure 3.2).

Curvature data were not normally distributed so non-parametric Kruskal-Wallis analysis was used to test for differences in curvature of trajectories between different categories. We also used a multiple comparisons test to identify the categories that were significantly different (Siegel and Castellan, 1988). Distributions of space use around the wind farm area were produced using a quartic kernel interpolation in the ArcGIS Spatial Analyst module. A quartic kernel was used as it is a good approximation of the Gaussian and in the absence of a more specific model of movement, a model approximating simple diffusion was considered a suitable estimation (H. Beyer pers. comm.). All data analyses were conducted using ArcGIS (version 9.2) with the additional package Hawth's Analysis Tools for ArcGIS (Beyer, 2004), and R (version 2.7.0).

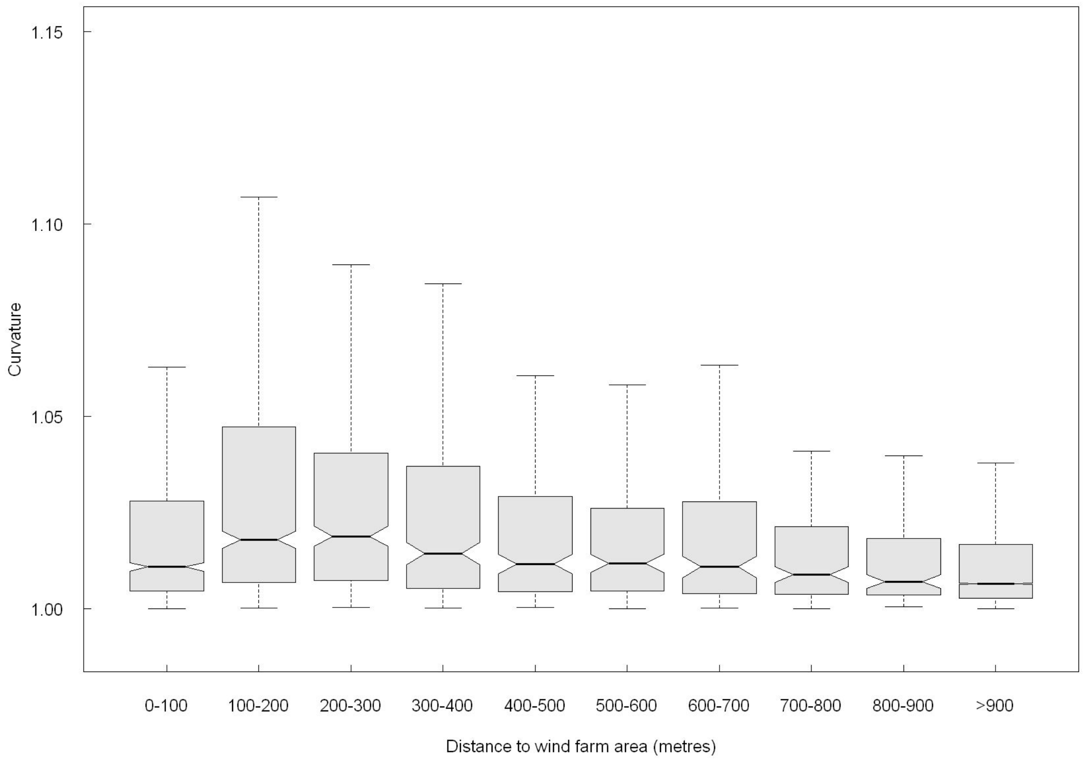

Figure 3.2 Curvature for all trajectories categorised by nearest distance to the wind farm area. Kruskal-Wallis chi-squared = 642.0206, df = 9, p-value < 0.05. Boxes represent the lower quartile, median and upper quartile values. Whiskers connect adjacent values within 1.5 times the interquartile range from the ends of the box.

3.3.3 Migration Scenarios

To assess the additional cost associated with the presence of the Nysted wind farm we first estimated the additional distance travelled by eiders post-construction within the study area.

Distance = $(C_{pre} - C_{post})$ x Median trajectory length

C_{pre} and C_{post} are curvature pre- and post-construction. Trajectory length was measured in metres. Satellite tracking data are not available for common eider, so the precise migration distances remain unknown. Previous estimates (Alerstam, 2001) were used in combination with location data from ringing recoveries of breeding and wintering female eiders from the Finnish population (Figure 3.1).

The overall cost of migration and the additional costs incurred due to the wind farm were estimated using the modelling software Flight 1.18 (Pennycuick, 2007). The model was used to estimate the cost of flight using aerodynamic principles and hence measure the cost of avoidance of the Nysted wind farm. We investigated different scenarios associated with the construction of several additional wind farms based on multiples of the response observed at Nysted. Also included was a comparison with the straight-line distance between breeding and wintering grounds as eiders already extend their annual migratory distance travelled over that of the shortest distance by avoiding flying over land. Model input parameters are shown in Table 3.1. The wingspan and wing area data were recorded from female adult eiders collected from Kalø Vig and Ebeltoft Vig, Denmark. The wing measurements were taken from 14 specimens following (Pennycuick, 1989). Fat mass was estimated by comparing the empty mass with the mass of lean females immediately after breeding (Christensen, 2008).

Table 3.1 Input values to the migration modelling software Flight 1.18 (Pennycuick, 2007).

Variable	Value	Reference Source
Empty Mass	2500 g	Henning Noer, DMU pers. comm.
Wing Span	0.9045 m	See methods
Wing Area	0.1192 m^2	See methods
Altitude	0	Sea level
Fat mass	1040 g	See methods
Distance to destination	To be determined	-
Cruising altitude	10.9 m	(Desholm, 2003)

3.4 Results

The data comprised 13 323 trajectories of which 2 593 were recorded pre-construction and 10 730 were post-construction of the wind farm; 806 trajectories were identified as eider, 245 pre-construction and 561 post-construction.

The median curvature for all trajectories was 1.0079 compared to 1.0174 for the records of eider. The trajectories post-near had greater curvature and variance than the other categories amongst all trajectory data (Figure 3.3). Kruskal-Wallis and multiple comparisons tests suggested that all categories were significantly different (Kruskal-Wallis: Chi-squared = 664.78, df = 3, $p < 0.05$) although comparisons including post-near had an effect size an order of magnitude greater than all other comparisons. A similar pattern was evident amongst the eider records (Figure 3.4). The median curvature of trajectories post-near was significantly greater than the curvature amongst the other categories (Kruskal-Wallis: Chi-squared = 89.77, df = 3, p < 0.05) and the variation in curvature was greater for the post-near category.

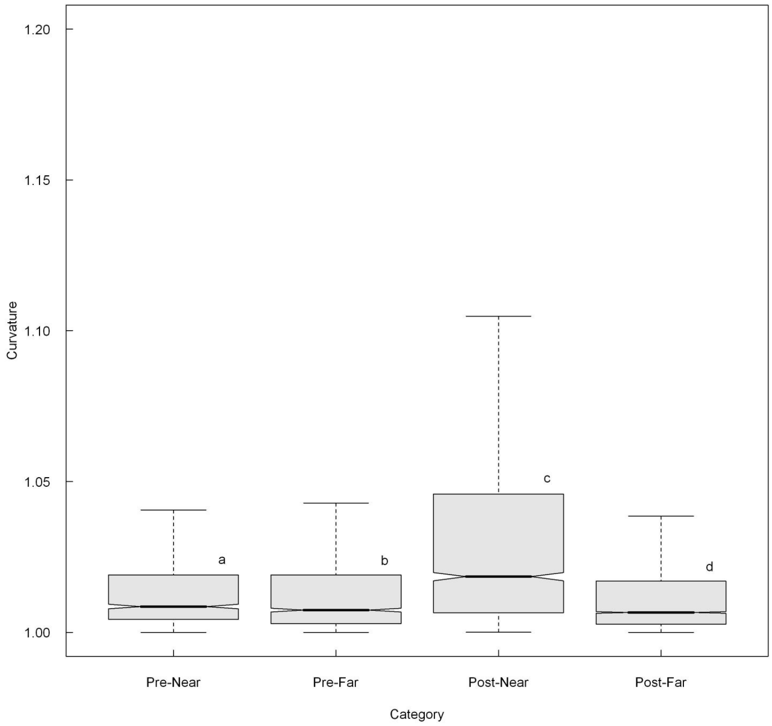

Figure 3.3 Curvature for all tracks both near (<500m to turbines) and far (>500m to turbines), pre and post construction of the Nysted wind farm. Kruskal-Wallis Chi-squared = 664.7844, df = 3, p-value < 0.05. Letters denote significant differences (multiple comparisons test, p = 0.05).

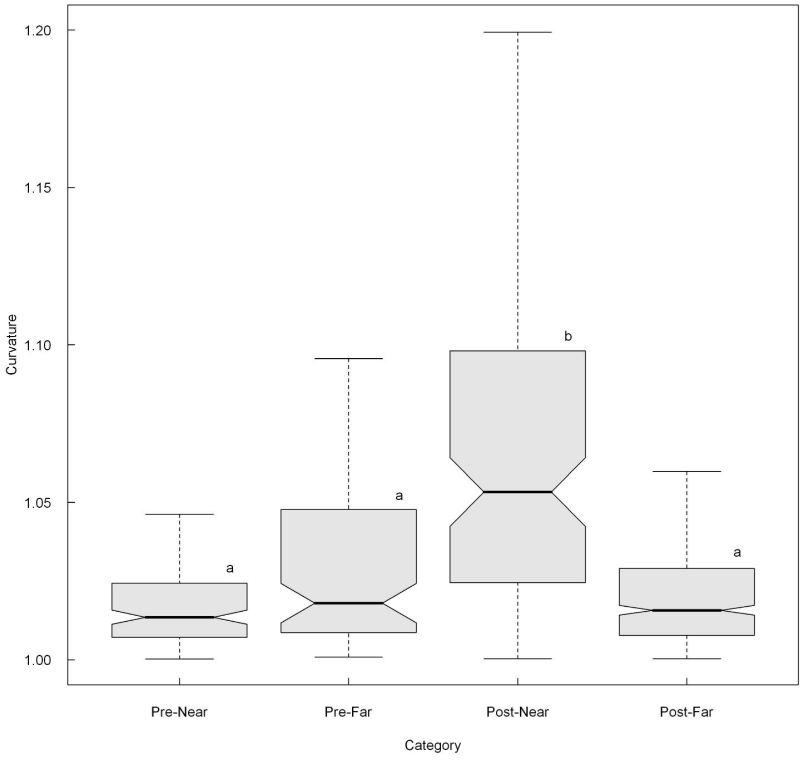

Figure 3.4 Curvature for eider tracks both near (<500m to turbines) and far (>500m to turbines), pre and post construction of Nysted wind farm. Kruskal-Wallis Chi-squared = 89.7699, df = 3, p-value < 0.05. Letters denote significant differences (multiple comparisons test, p = 0.05).

Minimum distance to the nearest wind turbine varied with category and species. Pre-construction, the trajectories for both eider and all the data were not significantly different (Figure 3.5). Post-construction, the median distance to the wind farm area increased significantly by 104 metres, from 56 metres to 160 metres, for all trajectories. Eiders exhibited a greater response, the median minimum distance to wind farm area increased from 50 metres to 224 metres, a displacement of 174 metres. The response of eider to the wind farm and the differences in space use are illustrated in Figure 3.6. Post-construction, the space used by eiders was reduced in the area of the wind farm when compared to that pre-construction with a corresponding increase in the use of surrounding areas, particularly to the south.

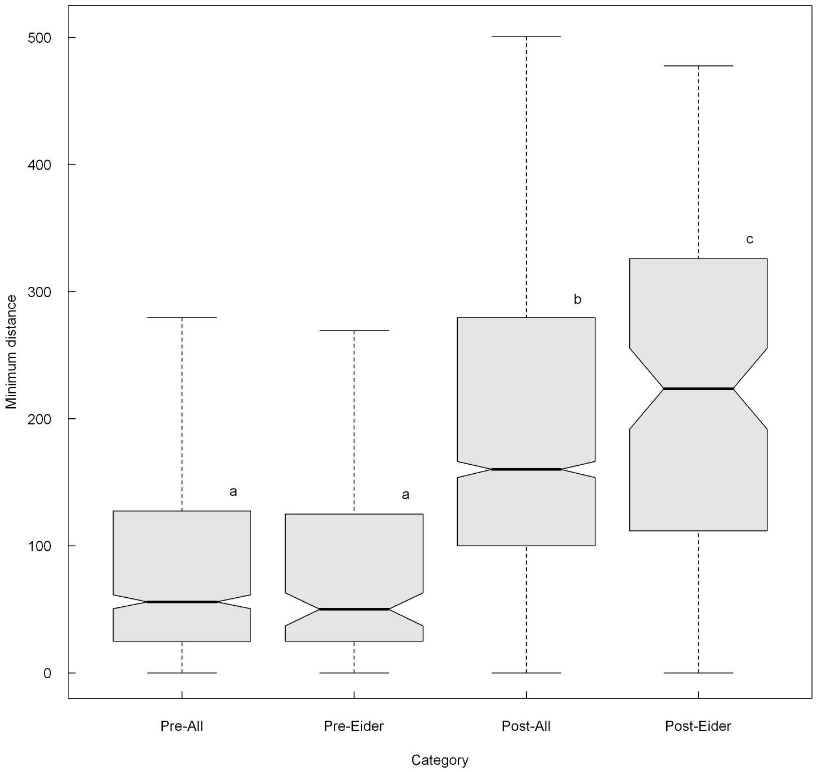

Figure 3.5 Minimum distance to wind turbines for all tracks and only eider that were <500m from the turbines, pre- and post-construction. Letters denote significant differences (multiple comparisons test, p = 0.05).

The median eider trajectory was 10.21 kilometres. The estimated curvature of eider trajectory pre-construction was 1.0135, increasing to 1.0533 post-construction. The additional distance incurred in the presence of the wind farm was therefore c.400 metres. The straight-line (great circle) distance between breeding and wintering grounds was approximately 1200 kilometres, requiring an estimated energetic expenditure of 13 300 kJ for eiders to fly the distance. The estimated distance of the likely migration route taken by eiders was 1400 kilometres equating to flight costs of 15200kJ. This difference between the two routes equates to a reduction in eider body mass of 0.06kg. Increasing the distance travelled by 1 kilometre (equivalent to 2.5 times the distance associated with the Nysted wind farm) had no detectable energetic cost and extra loss of mass. Only when the distance was increased to 1440 kilometres (equivalent to 100 Nysted wind farms) did the further reduction in mass of the bird exceed 0.5% (Table 3.2).

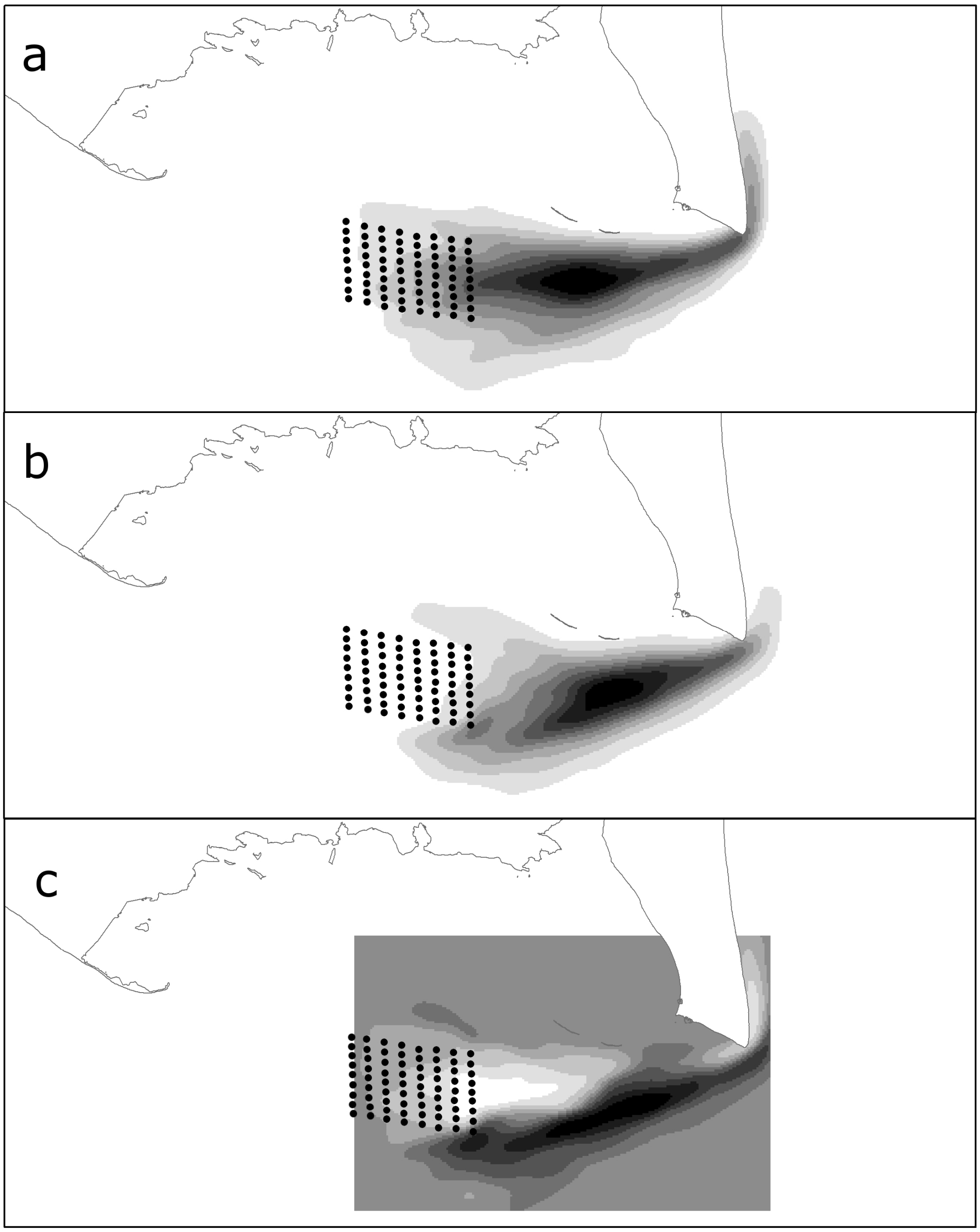

Figure 3.6 Kernels of space use by eider across the study area a) pre-construction b) post-construction and c) the difference in space use between a) and b). Darker colour represents greater use. Circles denote wind turbines.

Table 3.2. Estimated cost of flight associated with increasing distance travelled due to the avoidance response of common eider to wind farms (Pennycuick, 1989).

Windfarm Factor	Distance Travelled (km)	Cost (kJ)	Fat burnt (kJ)	Mass (kg)
0	1200	13300	12700	2.06
0	1400	15200	14400	2.00
2.5	1401	15200	14400	2.00
5	1402	15200	14400	2.00
10	1404	15200	14400	2.00
100	1440	15500	14700	1.99
1000	1800	18600	17700	1.88
4000	3000	27600	26300	1.59
5000	3400	30200	28700	1.50

3.5 Discussion

Little is known about the effects of wind farms on bird populations due to lack of pre- and post-construction comparative studies and Stewart et al. (2007) highlighted the weak methods and short duration of existing studies. Application of the BACI (Before-After-Control-Impact) method is advocated as the gold standard for study design in the context of wind farms but is rarely feasible due to time or monetary constraints and a lack of legislative necessity. This is the only data set recording bird movements both before and after the construction of an offshore wind farm, in an area of dense migratory movements, enabling us to answer questions not previously addressed.

Birds show avoidance responses to wind farms, but these vary within and between species (Hötker et al., 2006). Comparison of the pre- and post-construction data from Nysted showed individuals adjusted their flight trajectories to avoid the wind farm area post-construction, especially evident amongst common eiders (Figure 3.6). This species predominantly flew east to west pre-construction, shifting northeast to southwest post-construction and generally avoiding the area within the wind farm. Few trajectories passed between the turbines and the majority flew to the south of the wind farm. The variation in trajectories recorded may be due to differences in the distance at which birds show avoidance, with some reacting to the wind farm at several kilometres distance and others at close range. The differences observed in the route taken around the wind farm might also be due to differences in the prevailing wind direction and a risk aversion strategy to prevent being blown into the turbines. Only six trajectories navigated to the north, all during the prevailing southerly winds; since common eiders generally avoid flying over land, an alternative explanation of the data may be that birds avoided travelling to the north to avoid proximity to land.

Studies suggest that birds avoid wind farms but few have quantified avoidance rates or distances and these measurements are vital to understanding bird-wind farm interactions. Common eiders at Tunø Knob, Denmark showed avoidance at c.200 metres from that wind farm (Larsen and Guillemette, 2007). This was similar to the median minimum distance of 224 metres observed amongst eider at Nysed post-construction, representing a displacement of 174 metres from the pre-construction state. Other species flew closer to the wind farm but post-construction data also showed significant displacement. Hence, all birds respond to the wind farm but common eiders showed a greater avoidance response. One explanation for this could be that eiders are more risk averse than other species in the study.

Fox et al. (2006) highlighted barriers to movements as one of the effects of wind farms on bird populations. Our study showed that birds, common eider in particular, avoided the Nysted wind farm and flew around it, rather than between the turbines. The extent to which avoidance is considered an impact depends on the species, the size of the wind farm, the spatial arrangement of the turbines, the type of movement i.e. local movements between feeding, nesting and roosting areas or annual migrations, and the incurred energetic cost (Fox et al., 2006). The Nysted wind farm comprises 72 turbines occupying an area of approximately 60 km^2 therefore the extra distance required to fly around the wind farm is likely trivial for common eider migrating 1400 km or more. Trivial or not, the expectation was that curvature would differ significantly between trajectories recorded pre- and post construction due to an avoidance response. However, we predicted there to be no difference in curvature between trajectories far from the wind farm area, pre- and post-construction, as these birds were travelling at distances great enough to require no change in flight path. Figures 3.3 and 3.4 illustrate the differences in curvature pre- and post-construction for all bird species and common eider. The results for both analyses indicated that birds near to the wind farm area flew further post-construction. Amongst eider, the curvature was significantly greater for trajectories recorded near and post-construction equating to an additional 400 metres travelled whilst traversing the study area. When all trajectories were analysed, all categories were significantly different. This result is likely due to high statistical power resulting from the analysis of over 13 000 trajectories. The result is therefore statistically significant but it may not be biologically significant or relevant and graphically (Figure 3.3) it would seem that there was little difference between the categories pre-near, pre-far, and post-far.

General migration routes are known for many species, but knowledge of the fine-scale movements of birds on migration is limited. The Baltic/Wadden Sea population of common eiders mainly winter in the western Baltic and Wadden Sea and make the journey back to the Baltic Sea to breed. Individuals of this population therefore pass through the area of the Nysted wind farm and are potentially impacted by the wind farm. In the extreme, the energetic costs of avoidance behaviour and increased distance travelled would reduce the mass and condition of an individual to the point of adversely affecting breeding success. The estimated increase in distance travelled by common eiders in the presence of the Nysted wind farm was c.400 metres, 0.04% of the estimated distance travelled between wintering and breeding grounds. The cost of the additional distance travelled to avoid the wind farm was undetectable and a response similar to that of passing one hundred similar wind farms was required to achieve a loss in body mass (Table 3.2). The energetic cost for a single journey avoiding one wind farm is therefore insignificant compared to factors such as strong or unfavourable wind conditions (Hedenström and Alerstam, 1995, Pennycuick, 1978). However, if numerous wind farms were constructed along a migration flyway it may give cause for concern. Common eiders avoid flying over land and navigate around southern Sweden as shown in Figure 3.1 (Alerstam, 2001), yet these same individuals fly over mainland Denmark to reach the Wadden Sea. The associated energetic cost of this behaviour to avoid land is also greater than that of navigating around the Nysted wind farm (Table 3.2). In a larger context, the effect of Nysted is just one of many ways in which human activities impact on bird populations, others being collisions with buildings, predation by domestic animals, climate change, and hunting (Erickson et al., 2005, Kurle et al., 2008, Veltri and Klem, 2005, Woods et al., 2003, Huntley et al., 2007). For example, the annual Danish hunting bag for common eiders is 30 000 to 70 000 birds (Christensen, 2008).

This study is based on several assumptions which should be tested. It was assumed that each journey was an independent event and that individuals could compensate for the extra energetic costs by increased feeding rates in between events. If this is not the case then the impacts may be cumulative over time (Kalmbach et al., 2004). We considered a population undertaking a “one-off” annual migration and in this situation the cost of avoidance was trivial. However, if the population were commuting daily, the cumulative energetic costs of frequently avoiding a wind farm would be greater (Fox et al., 2006). For example Common Scoter (*Melanitta nigra* L.) or Long-tailed Ducks (*Clangula hyemalis* L.) moving between marine feeding and roosting areas daily during winter, or breeding terns moving frequently between marine foraging grounds and terrestrial nest sites. Furthermore, we only considered the displacement of individuals in latitude and longitude but not altitude. Desholm and Kahlert (2005) reported that at night, birds increased their flying altitude but the eider trajectories in this analysis were recorded during daylight hours. Therefore, in this analysis, altitudinal displacement was not considered but may add to the impact in other scenarios.

In conclusion, the additional distance travelled by common eiders due to the Nysted wind farm is unlikely to impact the population as the increased distance and associated energetic costs appear trivial. However, the cumulative effects of many similar wind farms built along a migration route would give cause for concern. Also, if other actions, such as habitat degradation, were to impact the population, then presently small effects of the wind farm may become important. Finally, we have considered a migratory scenario, however, it is possible that some species interact with wind farms daily and the effects may be greatly increased for these individuals.

4 Barriers to movement: modelling energetic costs of avoiding marine wind farms amongst breeding seabirds

Masden, E.A., Haydon, D.T., Fox, A.D., and Furness, R.W 2010 Barriers to movement: modelling energetic costs of avoiding marine wind farms amongst breeding seabirds. *Marine Pollution Bulletin. In Press*

4.1 Abstract

Proposals for wind farms in areas of known importance for breeding seabirds highlight the need to understand the impacts of these structures. Using an energetic modelling approach, we examine the effects of wind farms as barriers to movement on seabirds of differing morphology. Additional costs, expressed in relation to typical daily energetic expenditures, were highest per unit flight for seabirds with high wing loadings, such as cormorants. Taking species-specific differences into account, costs were relatively higher in terns, due to the high daily frequency of foraging flights. For all species, costs of extra flight to avoid a wind farm appear much less than those imposed by low food abundance or adverse weather, although such costs will be additive to these. We conclude that adopting a species-specific approach is essential when assessing the impacts of wind farms on breeding seabird populations, to fully anticipate the effects of avoidance flights.

4.2 Introduction

In response to climate change, the EU has set targets to achieve 20% of energy from renewable sources by 2020 (House of Lords European Committee 2008). Consequently, the UK wind power industry has grown rapidly, with associated concerns over adverse effects of wind farms on wildlife populations, particularly birds, and our ability to assess these impacts (Masden et al., 2010). The potential impacts of wind farms on bird populations can be grouped into three major types: direct mortality due to collision with turbines/infrastructure; physical habitat modification and/or loss due to the footprint of turbines and associated structures; and avoidance responses of birds to turbines (Fox et al., 2006, Masden et al., 2009, Fielding et al., 2006).

Birds exhibit avoidance responses to wind farms; whilst these vary within and between species (Hötker et al., 2006), concern remains over the extent and impact of these responses. Wind farms may act as barriers to movement, increasing distances travelled and so increasing energy expenditure. Reproductive success is often related to parental body condition at the time of breeding amongst provisioning birds (Wendeln and Becker, 1999), so any reduction in mass due to increased flight costs may be detrimental and may impact reproductive output. Masden et al. (2009) showed that for common eiders *Somateria mollissima* migrating over 1 400 km, the additional energy required to divert around the Nysted wind farm off southern Denmark (a medium-sized wind farm: 72 turbines covering an area of approx. 60 km^2) was trivial; responses equivalent to avoiding 100 such wind farms would be necessary to cause detectable reductions in bird body mass. However, breeding season impacts may be different and greater for other species. Seabirds typically have altricial offspring and parents commute daily between breeding colonies and foraging sites to provide food for their offspring. Therefore, seabirds could potentially interact with a wind farm located within their foraging range several times a day throughout much of the breeding season.

To date, there has been no consideration of the potential energetic costs to such birds that are forced to commute around offshore wind farms on a regular basis. This is particularly relevant for the breeding seabirds in the Firth of Forth, Scotland. The Scottish Offshore Wind Exclusivity agreements have identified four areas in the Firth of Forth for possible development and close by there is also another offshore site outside 12 nautical miles (The Crown Estate, 2009). All of these areas are within the foraging range of seabirds breeding on Bass Rock (of international importance for northern gannets *Morus bassanus*) and the Isle of May National Nature Reserve. Therefore it is quite possible that a breeding seabird may interact with, and be affected by, one or more wind farms.

This study is the first of its kind to model the likely impact of wind farms on a range of breeding seabirds due to the birds' avoidance responses. Using an energetic model parameterised with values from the peer-reviewed literature, we examine the extent to which wind farms of differing sizes impact different species of seabirds through increases in energy expenditure. We hypothesise that species will be differentially sensitive to wind farms due to their contrasting morphologies (body mass, wing span and wing area) and foraging characteristics (foraging distance and trips per day). Although set in the context of the Firth of Forth wind power developments, the concepts are globally applicable to the renewable energy sector more generally, as for example, in the case of movements of large marine predators around tidal turbines.

4.3 Methods

4.3.1 Species

Nine species of seabird were considered in this analysis (shag *Phalacrocorax aristotelis*, great cormorant *Phalacrocorax carbo*, Atlantic puffin *Fratercula arctica*, common guillemot *Uria aalge*, black-legged kittiwake *Rissa tridactyla*, lesser black-backed gull *Larus fuscus*, common tern *Sterna hirundo*, northern fulmar *Fulmarus glacialis*, and northern gannet). The choice was motivated by the need to capture a full range of contrasting morphologies, flight characteristics and foraging ecologies (see references and Table 4.1) when evaluating the potential impacts on different species, but balanced by the availability of empirical data.

Table 4.1 Input parameters for energetic model. Superscripts denote references. * highlights references that provided only maximum foraging range. [$] indicates where DEE was calculated using the 'All Seabirds' equation within Ellis and Gabrielsen (2002). [n] indicates the number of observations within the reference where data were provided. 1. Wanless and Harris (1992) [15]; 2. Pearson (1968)*; 3. Wanless et al. (1991) [31]; 4. Enstipp et al (2006); 5. Grémillet et al. (2004) [29]; 6. Grémillet (1997) [14]; 7. Grémillet et al. (1999) [18]; 8. Corkhill (1973); 9. Wanless et al. (1990) [14]; 10. Ellis and Gabrielsen (2002) [9]; 11. Hatchwell (1991); 12. Wanless et al. (1988) ; 13. Monaghan et al. (1994) [38]; 14. Hamer et al. (1993) [99]; 15. Daunt et al. (2002)* [9]; 16. Golet et al. (2000); 17. Becker et al. (1993) [91]; 18. Klaassen et al. (1992) [7]; 19. Phillips and Hamer (2000); 20. Hamer et al. (1997)* [168]; 21. Furness and Bryant (1996) [14]; 22. Hamer et al. (2001) [14]; 23. Birt-Friesen et al. (1989).

Species	Scientific Name	Mean mass (g)	Mean trips per day	Mean distance to foraging area (km)	DEE ($kJ.d^{-1}$)
Shag	*Phalacrocorax aristotelis*	1860	2 [1, 2]	12 [2, 3, 4]	2249 [2]
Great Cormorant	*Phalacrocorax carbo*	2560	3 [5, 6]	8 [5, 7]	2762 [$]
Atlantic Puffin	*Fratercula arctica*	387	3 [2, 8]	58 [2, 8, 9]	848 [10]
Common Guillemot	*Uria aalge*	891	2 [2, 11, 12]	12 [4, 13]	1641 [4]
Black-legged Kittiwake	*Rissa tridactyla*	368	5 [2, 14]	45 [2, 4, 15]	786 [16]
Lesser Black Backed Gull	*Larus fuscus*	831	2 [2]	42 [2, 3]	1328 [$]
Common Tern	*Sterna hirundo*	128	12 [2, 17]	19 [2, 17]	343 [18]
Northern Fulmar	*Fulmarus glacialis*	772	1 [19, 20]	184 [20]	1444 [21]
Northern Gannet	*Morus bassanus*	3000	1 [22]	160 [22]	4856 [23]

4.3.2 Calculating energy requirements: the intrinsic cost of flight

The energy required for flight was estimated using aerodynamic principles and the modelling software Flight 1.19 (http://www.bio.bristol.ac.uk/people/pennycuick.htm). Model input parameters are presented in Table 4.1. An extensive literature review was conducted and parameter values were sourced from the peer-reviewed literature; mean values of those reported in the literature were used to parameterise the model. Values for body mass were sourced from the British Trust for Ornithology (http://www.bto.org/birdfacts/) except for northern gannet (Cramp and Simmons, 1977). When considering foraging distance, some studies only provided maximum foraging distance; these references have been highlighted (Table 4.1). Variation in distance to foraging site and foraging trips per day are shown in Figure 4.1. We assumed that birds would fly close to V_{mp} (minimum power speed) (C. Pennycuick pers. comm.) and then estimated the power required to achieve this speed. It was assumed that ground speed was equal to air speed to make the analyses more general, as predominant wind direction and speed will undoubtedly be case-specific.

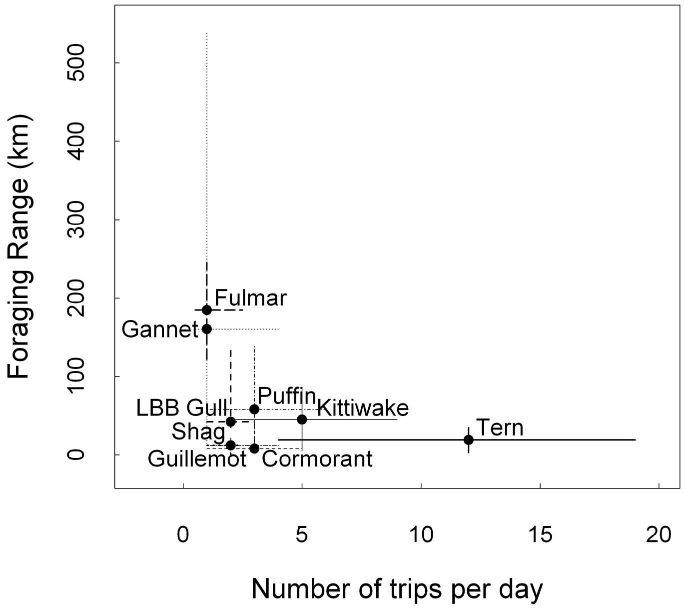

Figure 4.1 Variation in the number of foraging trips and distance to foraging site for different seabird species (Shag = shag, Cormorant = great cormorant, Tern = common tern, Guillemot = common guillemot, Puffin = Atlantic puffin, Kittiwake = black-legged kittiwake, LBB Gull = lesser black backed gull, Fulmar = northern fulmar, Gannet = northern gannet). Dots represent the parameter values used in the simulations. Lines denote the range of values reported in the cited literature. See Table 4.1 for data sources and samples sizes.

4.3.3 The energetic cost of barriers to movement

To estimate the cost associated with wind farms (when perceived by birds as complete barriers to movement) we increased the distance the bird had to fly to reach its foraging area. The additional distance (*Δd*) we considered ranged from 100 – 10 000 metres as a consequence of the potential different combinations of sizes, shapes and numbers of wind farms that might be avoided. This range of distances would incorporate a very minor shift in orientation, to completely circumventing a large wind farm. For example, the worst-case scenario for birds travelling around the Danish Nysted wind farm would be an additional 4 000 m. The energy required for flight (*Δe*) was calculated on a daily basis as follows:

$$\frac{\left(2\times\left(\text{foraging range}+\Delta d\right)\right)\times\text{number of trips}}{\text{speed}}\times\text{energy}$$

Where foraging range and *Δd* were measured in metres, speed was measured in $m.s^{-1}$, and energy was measured in Watts ($J.s^{-1}$). The calculations were completed for each species and *Δe* reported as the percentage of the daily energetic expenditure (DEE) for each species.

Model simulations were divided into three sections:

1.) To compare differences in *Δe* between species (due to morphology) we used constant values for foraging range (20 km) and number of foraging trips (4 per day). These values were chosen because they lie centrally within the possible ranges for all species.

2.) To examine variation in *Δe* within species due to foraging characteristics, we varied the number of foraging trips and *Δd* for each of the nine seabird species. Foraging trips ranged from 1 to 20 trips per day and *Δd* ranged from 100 to 10 000m.

3.) To assess the importance of species-specific analyses we compared differences in *Δe* due to the overall ecology of species (both morphology and foraging characteristics) and used species-specific values for foraging range and number of foraging trips (Table 4.1).

Data analyses were conducted using R (Version 2.8.1) and SigmaPlot 2001.

4.4 Results

4.4.1 Constant values

The increase in energy required (*Δe*) by different species under the same conditions (20 km, 4 times a day) is shown in Figure 4.2. *Δe* increases linearly as *Δd* increases, although the rate of increase differs between species; the rate of increase in *Δe* (slope of *Δd* vs. *Δe*) required by shag (when represented as a percentage of DEE) is 0.003% m^{-1} compared with a rate of 0.0009% m^{-1} for common tern. It can also be seen from Figure 4.2 that the largest difference in *Δe* for different species occurs when *Δd* is greatest. The difference in *Δe* between shag and common tern is 37 kJ when flying an extra 500 m but increases to 745 kJ when flying an extra 10 000 m.

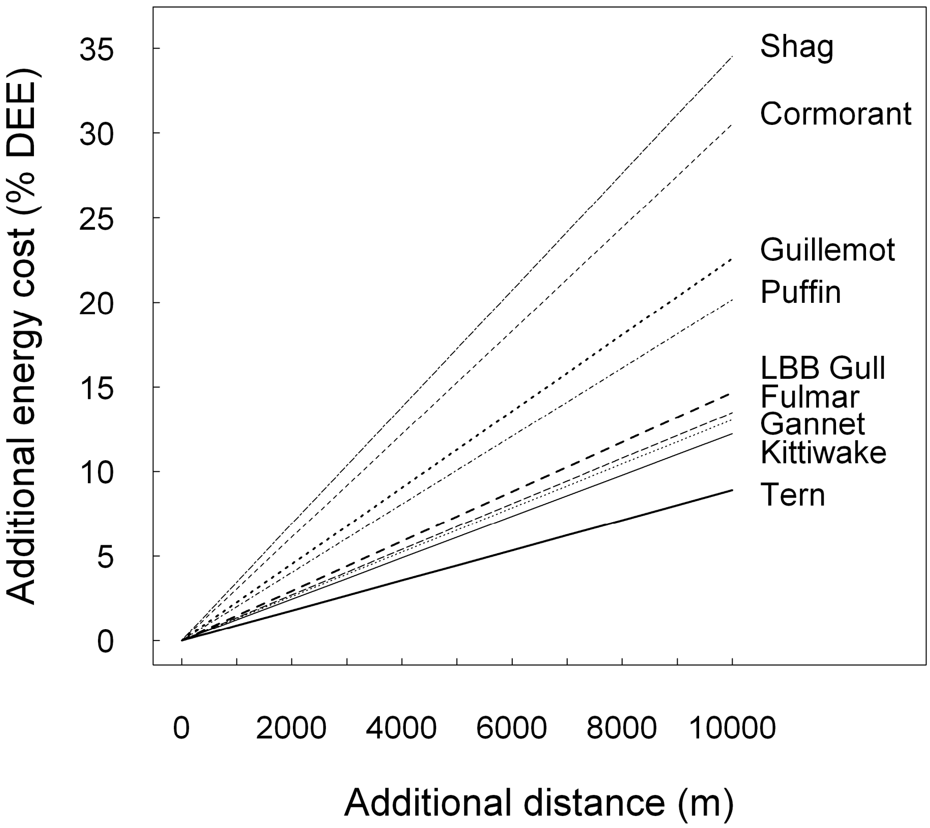

Figure 4.2 The increased energy expenditure (expressed as % DEE) associated with increasing additional distance (Δd) for birds of different species foraging 4 times per day, at a distance of 20 km from their breeding site. Lines represent different species. For complete species names see Figure 4.1.

4.4.2 Inter and intra-specific variation

Figure 4.3 represents the change in Δe for different species when foraging bouts vary in distance and frequency (as reported in Table 4.1). Great cormorant and shag had high rates of increase in Δe (0.0023% m^{-1} and 0.0018% m^{-1} respectively), but the highest (0.0027% m^{-1}) was for common tern (Figure 4.3). Northern fulmar and northern gannet had the lowest rates of increase. Greatest differences in Δe between species were again when Δd was 10 000 m.

Considering the overall foraging flight costs (*i.e.* the complete foraging trip), rather than just Δe (the cost of travelling Δd), there was large variation in the percentage of DEE that was required for flight (Figure 4.4). Although common tern, cormorant and shag, had the greatest rates of energy increase, black-legged kittiwake and Atlantic puffin used a larger percentage of their overall DEE on flight. An Atlantic puffin travelling an extra 10 000 m would require 103% of its normal DEE to fuel this extended flight.

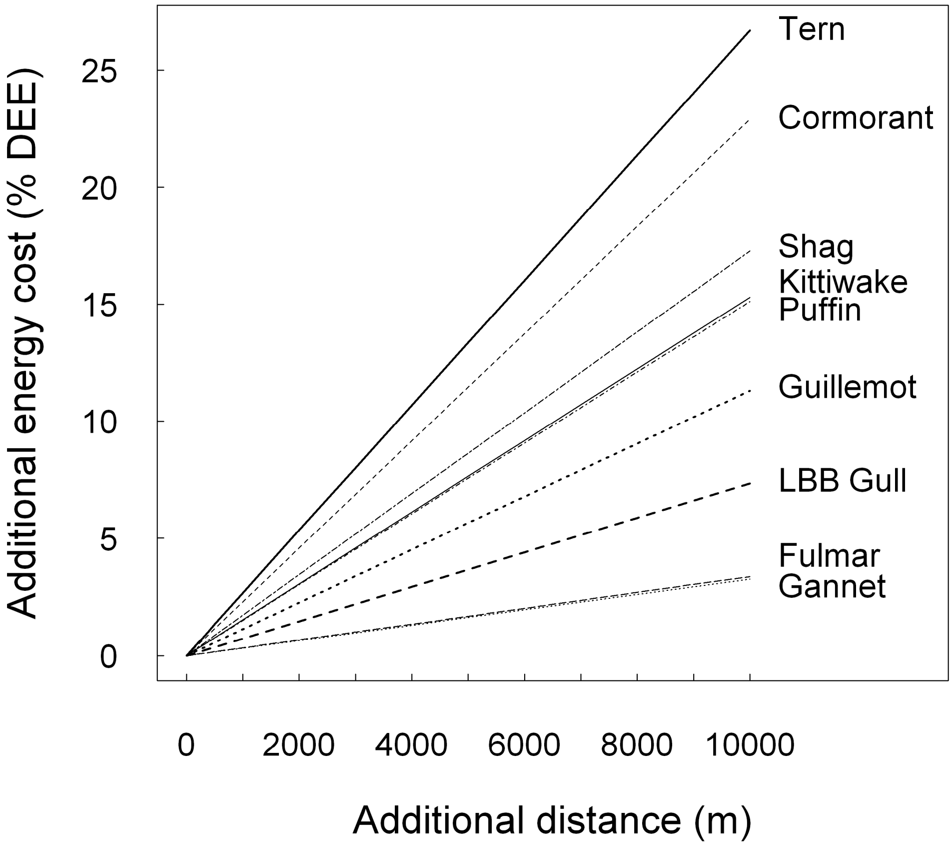

Figure 4.3 The increased energy expenditure (% DEE) associated with increasing additional distance (Δd) for birds foraging based on an "average" individual for each species (for parameter values used see Methods and Table 4.1). Lines represent different species. For complete species names see Figure 4.1.

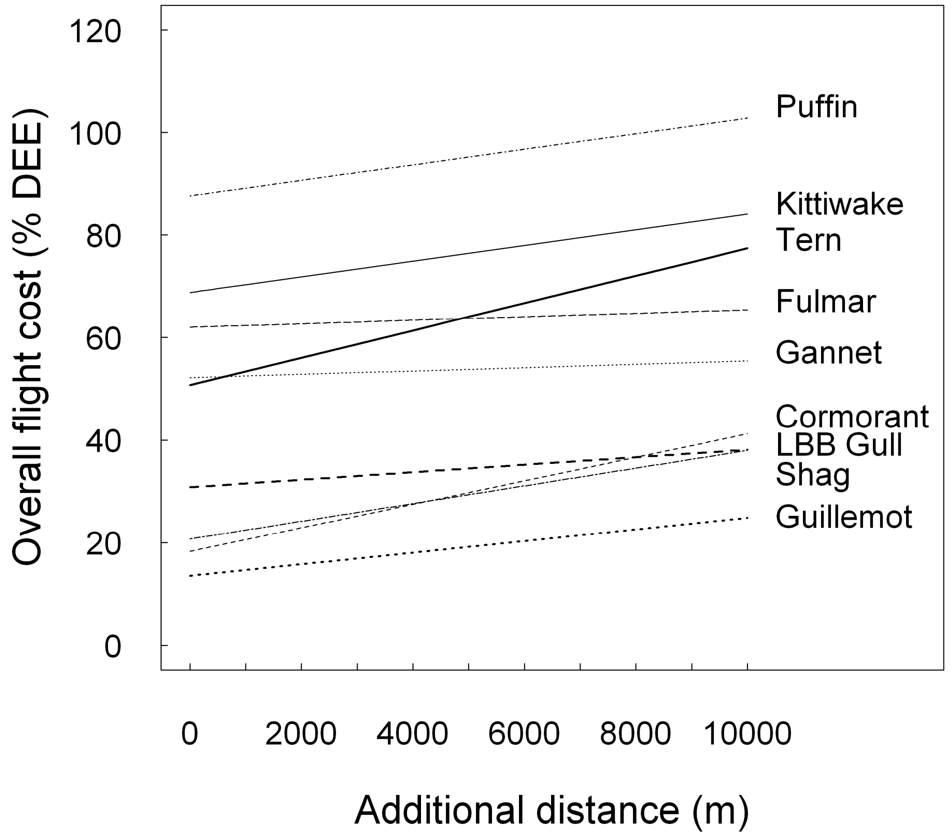

Figure 4.4 The overall energy expenditure (expressed as %DEE) of foraging flight for different seabird species. Lines represent different species. For complete species names see Figure 4.1.

There is variation in foraging range and trips within species (see Figure 4.1); the contour plots in Figure 4.5 show *Δe* (% DEE) expended by different species making varying numbers of foraging trips over a range of *Δd*. We assume that costs remained constant over time, therefore the results are relevant across varying foraging ranges. For all species, *Δe* required for flight increased as both *Δd* and number of foraging trips increased. However, the greatest increases in *Δe* (up to 170% DEE) were for shag, great cormorant, common guillemot and Atlantic puffin (Figure 4.5).

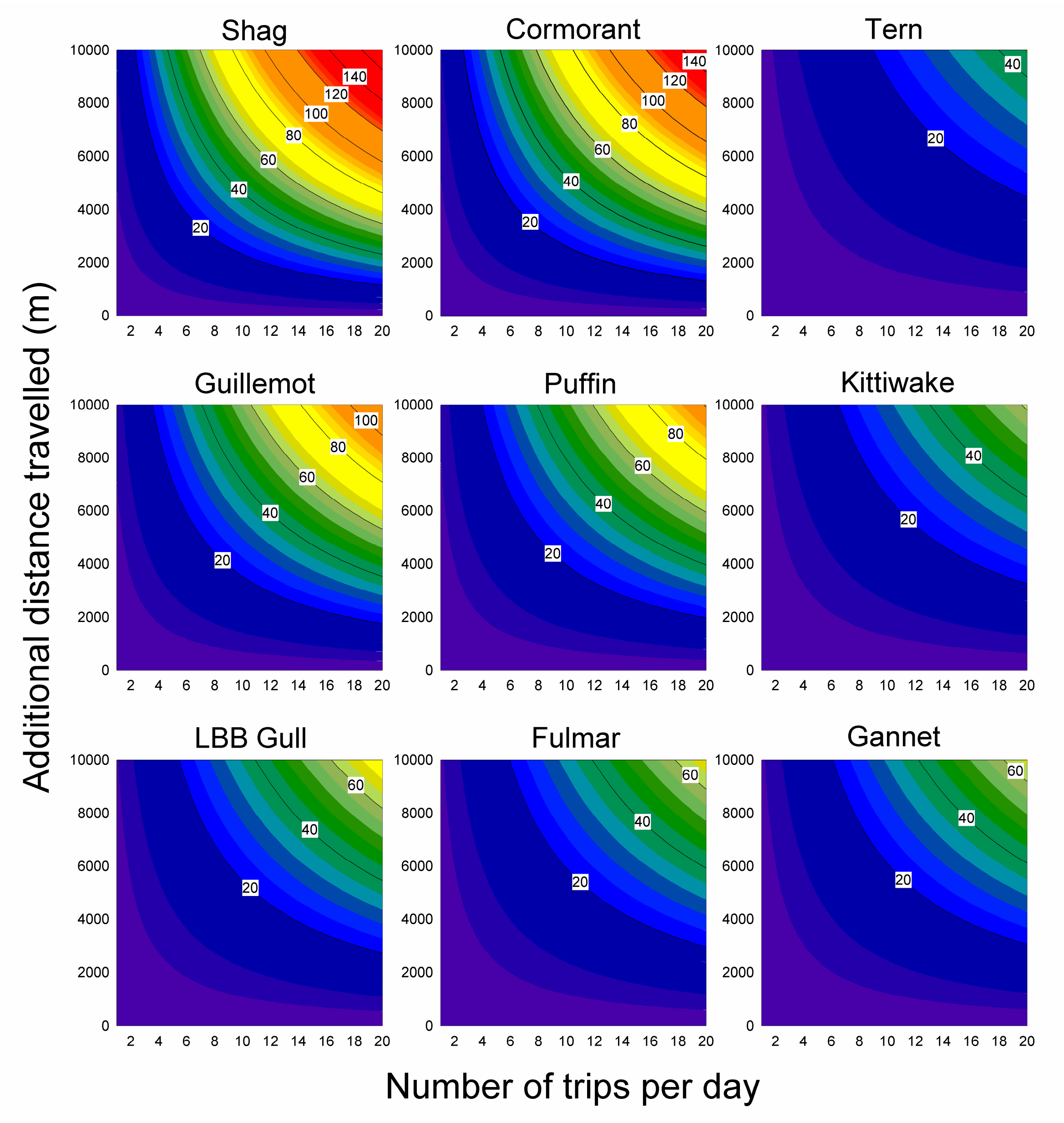

Figure 4.5 Energy requirement for flight according to number of trips and additional distance (Δ*d*) travelled for different seabird species. Colours and contours represent energy increase as %DEE. For complete species names see Figure 4.1.

4.5 Discussion

With an increase in the number of wind farm proposals, it is vital that developers consider the impacts and consequences of the construction of these potential barriers to animal movements. In this study, we assessed the potential cumulative impacts of wind farms on birds through the energetic costs of additional flight incurred during regular provisioning flights between nesting sites and feeding areas. With increasing numbers of wind turbines in the environment, and increasing observations that many species of birds exhibit avoidance behaviour towards wind turbines (Desholm and Kahlert, 2005), it is likely that individuals will have to fly increasing distances in order to reach their foraging grounds. The increased flight has energetic consequences that may impact upon the health of the population. For example, an increased energetic requirement that could not be balanced could lead to a reduction in the condition of a breeding bird, to a reduction in the fitness of its offspring, or both.

Bird species differ intrinsically in their morphology (e.g. variations in wing span and wing area in relation to body mass) so flight costs can be expected to be species-dependent. Variations in the additional costs of flying distances (Δd) beyond a constant 20 km, 4 times a day were species-specific, with shag and great cormorant requiring the most additional energy, followed by the auks (common guillemot and Atlantic puffin, Figure 4.2). The high cost of flight for these species is likely associated with large body mass and relatively small wing area, resulting in a high wing loading and hence relatively high cost of faster flapping flight (Benowitz-Fredericks et al., 2007, Calder, 1984, Pennycuick, 2008).

For a given distance, species have different levels of basic energy expenditure for flight and thus different species suffer proportionally more or less energetic penalties for each extra kilometre of flight caused by avoidance of objects such as wind turbines (Figure 4.5). Cormorants and auks undertake a few short provisioning flights (Figure 4.1), and hence experienced the greatest additional costs when performing many foraging trips per day, and travelling large additional distances as would be expected when avoiding wind farms. In contrast, northern fulmar and northern gannet undertake few but long foraging trips and are adapted to using efficient gliding flight, so the extra costs of additional distance are relatively small, although both species may have difficulty provisioning chicks in low-wind or strong head-wind conditions due to the high energetic cost of flapping flight (Furness and Bryant, 1996). Gulls (lesser black-backed and black-legged kittiwake) also show similar patterns, since they too use gliding flight, despite their shorter and more frequent provisioning trips. Finally, although common tern required the least energy when ecology and foraging characteristics were assumed constant across all species, it was the species most affected by the additional distance when foraging ecology was considered species-specific. A common tern typically completes 12 foraging trips per day and therefore would interact with the wind farm and incur the additional distance, 24 times per day. If the additional distance were 500m then the increase in energy requirement would be 1% of their DEE. However, 1% DEE may be insignificant when compared to unsuitable wind conditions or changes in prey density (Furness and Bryant, 1996, Hamer et al., 1993). For example, Furness and Bryant (1996) found that breeding northern fulmars more than doubled their wing-beat frequency and increased at-sea metabolic rate by 100% when mean wind speed decreased from 8 $m.s^{-1}$ to 3 $m.s^{-1}$.

Although the cost of flying *Δd* may be small, the overall cost of foraging flights should be considered, and not simply the additional cost incurred due to wind farms. The sensitivity of a bird to any incurred additional energetic costs is likely dependent on how close it is operating to its physiological limit. Figure 4.4 shows how the overall energetic cost of flight increases for each species and it can be seen that although common tern, great cormorant and shag have the greatest rates of increase in energy requirement, it is Atlantic puffin and black-legged kittiwake that incur the greatest energy costs relative to their DEE. If an Atlantic puffin were to travel an additional 10 000 m due to the presence of wind farms then it would expend 103% of its DEE on the extended flight activity alone. Should an individual be unable to acquire this additional energy without extra costs, it would soon be in deficit and the condition of the bird would decrease, affecting the fitness of its offspring.

Species show variation in their foraging characteristics as demonstrated in Figure 4.1. Some of this variation may be associated with environmental conditions, with birds having to travel further in bad years due to low food availability.

Hamer et al. (1993) reported black-legged kittiwakes travelling 5 km to foraging areas in a good year but 40 km in a bad year, with the frequency of foraging trips decreasing from 9 to 2 times a day. Their study indicated that variations in prey abundance from year to year can have a marked effect on seabird energy expenditure and breeding success; it is against this background of strong variation in natural conditions that we must assess the impact of extra flight by seabirds commuting past and around wind farms at sea. In this situation, based on additional energy requirements of flying greater distances, birds in good years would be impacted more by the development of wind farms and the consequent increase in foraging trip distance (Figure 4.5). Although, on the whole, individuals during a bad year would experience a greater impact despite lower absolute energy costs, because they would be closer to their physiological capacity in terms of energy balance if prey availability is low. Individuals may also forage further from a breeding colony as colony size and therefore competition for food resources increases. Such competition may deplete food around a colony (Ashmole's Halo: Birt et al., 1987, Gaston et al., 2007, Ashmole, 1963) forcing birds to travel further to meet their foraging demands, especially later in the breeding season. Additional distance and energetic costs associated with increasing numbers of wind farms may therefore impact individuals in smaller colonies to a greater extent than those in larger colonies. This is because these birds will be travelling shorter distances to forage and potentially passing the wind farm more times each day, therefore the additional cost will be a larger proportion of their daily energetic budget. Although it is possible that the consequences may be more severe for seabirds in large colonies if competition is already requiring them to work at their physiological limit (Lewis et al., 2001).

In this analysis we have only assessed the impact in terms of energy costs; however, it may be the case that there would be additional impacts associated with changing foraging ecology. For example, a bird travelling further to a foraging location, if travelling at a constant speed, will be away from the nest for longer. Therefore, during any given day, the time available for nest defence and pair-bonding between the two adult birds will be reduced and this may impact upon the success of the breeding attempt (Caldow and Furness, 2000). However, there are many factors which can lead to variation in the proportion of time when both adults are present at the nest, for example poor weather conditions (Finney et al., 1999). If birds are travelling further and there is no change in the speed of flight, there will be a reduction in the number of foraging trips that can be made within a day. If an individual is limited to carrying single prey items, then the amount of food brought back to the nest will be reduced. Compensation for this may involve travelling with more or bearing larger prey items which may also carry an energetic penalty; both of these are mechanisms which may also contribute to deterioration in the condition of the offspring. Birds can increase flight speed to compensate for travelling the additional distance, but only with an associated energy cost, which if not recovered, may lead to a decrease in adult body condition and ultimately fitness.

The results generated by this study are based on several assumptions that should be considered in relation to our conclusions. It was assumed that the cost of flight remained constant over time. We included neither variation in the mass of the bird during foraging trips nor varying wind conditions. The mass of a bird and the associated energetic cost of flight will increase when carrying prey items and probably decrease with flight activity. Therefore it was assumed that this variation would balance over the period of a day. The effects of wind were excluded from calculations of energy expenditure since wind speed and direction will inevitably vary in space and time. Generally, during a foraging trip and over the course of a day an individual will experience both head winds (increased energy expenditure) and tail winds (decreased energy expenditure) so net energy expenditure due to wind is likely to be low. Another assumption made was that birds fly at minimum power speed (V_{mp}), the speed at which less power is needed to fly than at faster or slower speeds (Pennycuick, 1989). If this assumption is not met, then the absolute values for overall energy expenditure will vary from those reported within this study. However, the relationships between energy expenditure, additional distance and number of trips, and between species would remain the same. Similarly, if another method other than aerodynamic theory was used for energy calculation, for example allometry (Castro and Myers, 1988), then the absolute values for energy expenditure would probably have varied slightly from those reported, although the general relationships would hold true. Finally, we only considered foraging in the day time period because the majority of studies report only data collected during daylight hours. If birds continue to forage during the night then the number of trips per day will increase and there will be an associated increase in energy expenditure (Figure 4.5).

4.6 Conclusions

The energetic costs of flying around one wind farm may be insignificant for the range of breeding seabirds considered in this study. Nevertheless, as the number of wind farms increases, so too will the cost of reaching foraging grounds as birds will have to fly further on each journey. This also confirms the need to study the potential mitigating effects of varying wind farm geometric design and inter-turbine distances as a means of reducing such potential additions to breeding seabird energetic expenditure. The results clearly show that it cannot be assumed that the effects will be similar across seabird species. Due to the differences in ecology of seabirds there is variability in the effects of wind farms and therefore a species-specific approach should be taken when assessing the barrier effects of wind farms on birds.

5 Modelling bird movements in response to marine wind farms: a Bayesian approach

5.1 Abstract

EU renewable energy targets have stimulated the rapid growth of the wind power sector, but the associated environmental impact assessment (EIA) process has not kept pace with developments. Wind farms contribute to carbon emission reductions but there is a need to ensure that the wind farms themselves do not adversely impact the environment, particularly birds. We developed Bayesian models based on observed avian avoidance responses to wind farms that can predict such impacts and contribute potential mitigation measures. Flight trajectory data collected post-construction of the Danish Nysted offshore wind farm were used to parameterise four competing models to describe bird movements, based on the premise that individuals show avoidance behaviour to turbines. The model most closely resembling the observed data incorporated individual variation in the minimum distance at which birds responded to the turbines. We show how such models can contribute to the planning process by assessing the effects of wind farm size, turbine spacing and configurations on the probability of birds passing between the turbines. Avian movement models can make new contributions to EIAs of wind farm development and reduce avian impacts at the planning stage, but a lack of available post-construction data currently limits progress.

5.2 Introduction

Many countries are increasing their use of renewable energy, in particular wind energy, in an effort to curb the effects of climate change. Increasing numbers of wind farms are being developed both onshore and offshore, with potentially negative effects on wildlife, especially birds. When birds exhibit avoidance behaviour towards turbines, wind farms may act as barriers to movement (Desholm and Kahlert, 2005, Masden et al., 2009), increasing flight distances and so elevating energy expenditure. Lack of avoidance behaviour puts birds at risk from mortality through collision with the structures (Erickson et al., 2005, Langston et al., 2006). Wind farms may also affect birds through habitat loss, either directly as a consequence of the turbine ‘footprints’ or indirectly through avian avoidance responses to turbines (West and Caldow, 2006, Pearce-Higgins et al., 2008, Madders and Whitfield, 2006).

When planning a wind farm it would be beneficial to be able to predict how individual birds respond to a range of different wind turbine locations and configurations. For example, under what circumstances are individuals more likely to fly around or through an array of turbines? This would enable informed judgements to be made about where to develop wind turbines, in what densities and in which configurations to minimise barrier effects and/or collision risk. Until recently, the only types of movement data available regarding bird and wind farm interactions were i) observational watches recorded during environmental impact assessments (EIAs) consisting mainly of information on flying heights in the immediate vicinity of the wind farm; or ii) long distance movements from bird ring recoveries that provide general information on movements, from which it may be deduced, assuming the most direct route, whether a bird could have interacted with a wind farm. Therefore until recently it was not possible to describe in detail the movements of birds in response to wind turbines; however, there are now technologies such as surveillance radar and satellite tracking devices that can provide data at a greater spatial and temporal resolution (Bevanger et al., 2008, Gauthreaux and Belser, 2009, Kelly et al., 2009).

Despite being an important factor in determining animal distributions, animal movement often remains poorly understood (Turchin, 1998, Nathan et al., 2008).

However, the response of animals to landscape features can be quantitatively described and movement paths characterised by mathematical models (Morales et al., 2004). Movement models can be separated into two types: i) statistical models that describe emergent properties of the data such as sinuosity, first passage time and fractal dimension (Bailey and Thompson, 2006, Benhamou, 2004, Schick et al., 2008); and ii) mechanistic models that aim to describe the underlying movement process, often using modified correlated random walks or diffusion processes (Codling et al., 2008, Benhamou, 2006). Only the latter can link movement processes mechanistically to covariates such as habitat type (Morales et al., 2004) and therefore have the capacity to predict an animal's movement patterns. For that reason, only these mechanistic models can be used to contemplate the consequences of landscape change. Technological advances in tracking methods have increased the feasibility of data collection to parameterise such mechanistic models whilst improvements in computing power have now made it possible to numerically fit these often complex models to data (Patterson et al., 2008, Rutz and Hays, 2009).

The aim of this study is to illustrate how data collected in the EIA process could be used more efficiently to aid planning and development of the wind power industry, and minimise the impacts on wildlife. This study is the first of its kind to apply current methods from animal movement analysis, to radar data collected during the post-construction assessment of an offshore wind farm, and quantitatively describe the movements of birds around a wind farm. Fitting complex models to data is often limited by classical estimation techniques, therefore we used Bayesian methods of analysis and performed inference with JAGS (Just Another Gibbs Sampler, Plummer (2009)). We give two examples of how such a model can be used to improve the assessment of the impacts of wind farms on birds: (i) the effect of wind farm dimensions on the number of birds passing between turbines; and (ii) the effect of different configurations of turbines on the permeability of a wind farm.

5.3 Methods

5.3.1 Data collection and processing

Data were collected from the Nysted offshore wind farm which comprises 72 wind turbines in eight north-south oriented rows, 850 metres apart at 480 metre intervals east-west, covering an area of c.60 km^2 in the western Baltic Sea south of Denmark. Flight trajectories of autumn migrating common eider *Somateria mollissima* were recorded using surveillance radar mounted on an observation tower near the wind farm (Petersen et al., 2006). Birds entering the detection area created an echo on the radar monitor, and by observing the echoes, the migration trajectory could be determined (see Desholm and Kahlert (2005) for data collection methods). Only east-west trajectories were used in this study due to the position of the radar in relation to the wind farm. We also used only those trajectories that came within 500m of a wind turbine as birds showed very little response to the wind farm at distances greater than 500m (Masden et al., 2009). The selected trajectories were converted from continuous lines to discrete points at 100 metre intervals using ArcGIS (version 9.3) and Hawth's Analysis Tools for GIS (Beyer, 2004). The final data set contained 89 individual trajectories comprising 70 to 230 data points (median = 127).

5.3.2 Models

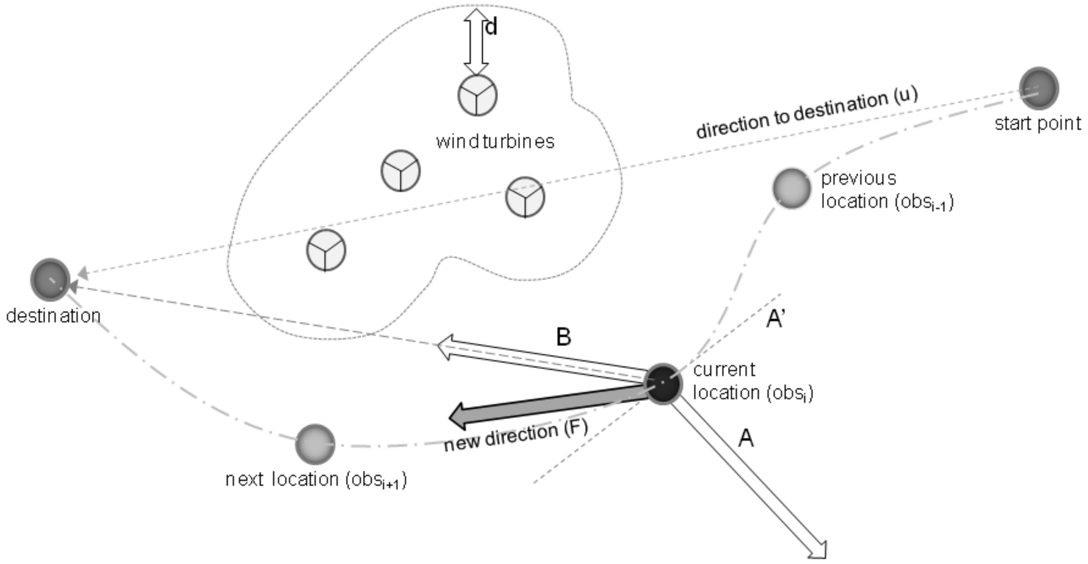

Figure 5.1 Diagram of the general principles of the model showing an example movement trajectory (dot-dash line) and variables within the model.

Here we present four models, each designed to describe the movements of birds in response to wind turbines. We assume that individual birds travel directly from a starting location towards a final destination and exhibit avoidance behaviour towards a wind farm. For each observation (*obs*$_i$), the models estimate the direction of movement to the next observation (*obs*$_{i+1}$) by resolving the forces attracting a bird to its final destination and repelling it away from a wind farm, the proportion of each depending on the distance between the bird and the wind farm (see Figure 5.1 for a diagram), and the method of resolution differing between models. The direction in radians (Φ) between each pair of observations is assumed to be independently drawn from a wrapped Cauchy distribution with parameters μ (the mean direction) and ρ (the cosine of the angular distribution). The wrapped Cauchy was considered suitable to describe direction of movement because it is a circular distribution and was previously used in a study by Morales et al. (2004) to model turning angles of random walks. The likelihood function is:

$$\mathrm{L}\left(\phi_{11}\cdots\phi_{Nn_N}\left|\mu_{11}\cdots\mu_{Nn_N},\rho\right.\right)=\prod_{i=1}^{N}\prod_{j=1}^{n_i-1}\mathrm{C}\left(\phi_{ij},\mu_{ij},\rho\right)$$

where N is the number of trajectories i.e. 89, n_i is the total number of observations of trajectory *i*, ϕ_{ij} is the observed direction of the next point in the trajectory from point *j*, μ_{ij} is the predicted mean direction, and C denotes the wrapped Cauchy distribution (Fisher, 1993) with density function:

$$\mathrm{C}(\phi,\mu,\rho)=\frac{1}{2\pi}\frac{1-\rho^2}{1+\rho^2-2\rho\cos(\phi-\mu)}$$

$$0\le\mu<2\pi \quad 0\le\rho\le1$$

Model 1: A model that assumes the direction of travel is simply the sum of the attractive force and the repellent force (adjusted by a scaling factor). The repelling force exerted by each turbine in the wind farm is described with an inverse power law with power *p*-1.

$$V_{ijk}=\frac{1}{l_{ijk}^{\;p}}\begin{pmatrix}x_{ijk}\\ y_{ijk}\end{pmatrix}$$

Where $\begin{pmatrix} x_{ijk} \\ y_{ijk} \end{pmatrix}$ is the vector from the wind turbine k to the j^{th} observed location of bird i and l_{ijk} is the length of this vector. **A**$_{ij}$ is the sum of these forces summed over all turbines (T) and is the overall repulsion exerted on a bird at a given location by the wind farm.

$$\boldsymbol{A}_{ij} = \sum_k V_{ijk}$$

The attraction towards the final destination is represented by the vector **B**, where *u* is the bearing to the final destination. However, the distance between the start and the destination is sufficiently great that **B** does not significantly change direction over the course of the trajectory.

$$B = \begin{pmatrix} \cos u \\ \sin u \end{pmatrix}$$

The resultant unit vector (**F**$_{ij}$) describing the direction of travel is thus:

$$\begin{pmatrix} \cos \mu_{ij} \\ \sin \mu_{ij} \end{pmatrix} = \boldsymbol{F}_{ij} = \frac{\boldsymbol{B} + c\boldsymbol{A}_{ij}}{\sqrt{1+c^2}}$$

where *c* is a scaling factor and from which we can derive the bearing μ_{ij}. The bird then travels in this direction.

Model 2: A model constrained to contour round the turbines. Vectors **A**$_{ij}$ and **B** are estimated as for model 1 but **F**$_{ij}$ is now a weighted sum of $\mathbf{A}^{\mathbf{T}}_{ij}$ (perpendicular to **A**$_{ij}$) and **B**, each scaled according to the dot product of **A**$_{ij}$ and **B** (**A**$_{ij}$•**B**).

$$p_{b_{ij}} = \operatorname{logit}^{-1} k_b \left(\boldsymbol{A}_{ij} \cdot \boldsymbol{B} + d^{1-p} \right)$$

$$\boldsymbol{F}'_{ij} = p_{b_{ij}} \boldsymbol{B} + \left(1 - p_{b_{ij}}\right) \frac{\boldsymbol{A}^{\boldsymbol{T}}_{ij}}{\left|\boldsymbol{A}^{\boldsymbol{T}}_{ij}\right|} \qquad \boldsymbol{F}_{ij} = \frac{\boldsymbol{F}'_{ij}}{\left|\boldsymbol{F}'_{ij}\right|}$$

$$\text{where } \boldsymbol{A}' = \begin{pmatrix} -a_y \\ a_x \end{pmatrix} \text{ and } \qquad \boldsymbol{A}^{\boldsymbol{T}} = \begin{cases} \boldsymbol{A}' & \text{with probability } \operatorname{logit}^{-1} k_c \boldsymbol{A}' \cdot \boldsymbol{B} \\ -\boldsymbol{A}' & \text{otherwise} \end{cases}$$

At each movement step, a bird must choose whether to fly directly towards its destination (**B**) or to turn away in response to the turbines. If vectors **A**$_{ij}$ and **B** are acting in the same direction then **A**$_{ij}$•**B** i.e. |**A**$_{ij}$| x the cosine of the angle between the vectors, will be large and positive. Consequently, p_{bij} will be close to 1 and the bird will continue directly to its final destination, however if $\mathbf{A}_{ij}$ and $\mathbf{B}$ are opposing then the dot product will be negative, p_{bij} will be closer to 0 and the bird will turn more towards $\mathbf{A}^T{}_i$. In the absence of strong repulsion, the bird will follow $\mathbf{B}$. The decision on how far to turn towards $\mathbf{A}^T{}_i$ depends on the distance to the turbines: the parameter d is the distance from a single turbine at which a bird would turn exactly half-way from $\mathbf{B}$ to $\mathbf{A}^T{}_{ij}$, and k_b is a scaling factor that determines how quickly the bird moves from $\mathbf{B}$ to $\mathbf{A}^T{}_{ij}$ as it approaches the turbines. For example, a high value of k_b will make a bird turn away suddenly at d, whereas a low value will make it start turning away slowly earlier. However, when the bird has approached closer to the turbines, it will follow a contour $\mathbf{A}^T{}_{ij}$ which keeps the magnitude of the repulsion constant until it can get round them.

As well as deciding how much to turn away, the bird must also choose which direction to turn. If $\mathbf{A}_{ij}$ and $\mathbf{B}$ are in exactly opposite directions then the bird will randomly choose either left or right, as neither choice will make it reach its destination quicker. Otherwise the bird will tend to turn from $\mathbf{B}$ in the direction in which $\mathbf{A}_{ij}$ is closer, which should correspond to the shorter route round the turbines. Whether the bird is to turn to the right or to the left is determined by a Bernoulli random variable. The scaling factor k_c determines how frequently the bird will turn in the correct direction, with high absolute values of k_c indicating that it will always choose the shorter route to its destination whilst a zero value for k_c would indicate a 50:50 chance of going either way around the wind turbine array.

Model 3: In model 2 we assumed that parameters were constant across all trajectories. In model 3 we relaxed this assumption and fitted parameter d_i separately for each of the 89 trajectories to include individual variation in the distance at which birds responded to the wind turbines. The d_i values were taken from a gamma distribution because d_i had to be positive and the gamma distribution takes only real and positive values.

$d_i \sim Gamma(shape, shape/d)$

Model 4: In model 3 we fitted parameter *d_i* separately for all trajectories but assumed that *u* was constant across all trajectories. In model 4 as well as fitting *d_i* separately we also fitted parameter *u_i* separately for each of the 89 trajectories to include individual variation in the bearing to the final destination. The *u_i* values were taken from a normal distribution because it was known that all birds were heading in the same overall direction, but that there would be some variation around this mean direction. However, the variation was unlikely to span 2π and 0, meaning a circular distribution was not required.

$u_i \sim Normal\ (u, \tau)$ where $\tau = \sigma^{-2}$

5.3.3 Model Parameterisation

Models were fitted using Monte Carlo Markov Chain techniques as implemented in JAGS (Plummer, 2009). For prior distributions see Table 5.1. For each model we ran three MCMC chains for 100 000 iterations and examined convergence and autocorrelation for the model parameters. Convergence was assessed using the Gelman-Rubin convergence statistic (Gelman and Rubin, 1992) which compares variance between and within Markov chains. Values close to 1 indicate convergence.

5.3.4 Goodness of fit

To compare the fit of the 4 competing models we used posterior predictive checks (Gelman et al., 2004). We assessed whether movement trajectories produced by the models had characteristics similar to those observed in the data. The characteristic we used was the number of trajectories that passed between the five central turbines on the eastern boundary of the wind farm i.e. trajectories that entered the middle of wind farm. This was a feature of the data that was not used to fit the models and was used because it quantified the number of individuals entering and moving through the central area of the wind farm and not just the periphery. For 89 trajectories we sampled from the posterior distributions of model parameters. Movement trajectories were then simulated using these sampled parameters. Starting locations were selected from the original data. This simulation process was repeated 50 times and the number of trajectories that entered the wind farm were recorded and compared against the original data. The model producing tracks that were most representative of the original data was chosen for the remainder of the study.

Table 5.1 Prior distributions for the parameters in the models

Parameter	Prior
u	uniform (3.1,3.6)
ρ	uniform(0.7,1)
c	uniform(0, 100)
d	uniform(0.2,5)
p	uniform(1,5)
k_b	uniform(0,20)
k_c	uniform(-20,0)
shape	gamma(0.01,0.01)
τ	uniform(1,100)

5.3.5 Simulations

Using the parameter estimates from the chosen model, we simulated movement trajectories of birds through areas with wind turbines. We ran simulations to investigate:

1. The effect of wind farm dimensions on the number of birds passing between turbines
2. The effect of different configurations of turbines on the permeability of a wind farm.

5.3.5.1 The effect of wind farm dimensions on the number of birds passing between turbines

If a species has a high risk of collision due to its behaviour it may be beneficial to design wind farms that ensure the birds do not fly through the array in order to reduce the risk of collision mortality. Therefore it is important to be able to predict the number of birds likely to pass between turbines at varying turbine spacing. A wind farm comprises horizontal rows of turbines and vertical columns. Ignoring potential constraints on turbine spacing due to the effects on turbine efficiency, we varied the distance between rows of turbines (from 200 to 1 000 metres at intervals of 20 metres) and also the number of columns in an array (from 1 to 8), using the Nysted wind turbine array as a template. We simulated 100 trajectories for each combination of inter-turbine distance and number of columns, and recorded the number of trajectories that entered the wind farm through the central five turbines on the eastern boundary of the wind farm. To account for any possible differences due to approach angle, the trajectories were started from 10 different locations.

5.3.5.2 The effect of different configurations of turbines on the permeability of a wind farm

Some species may be more sensitive to increased energy costs due to wind farms acting as barriers to movement, rather than having a high risk of collision mortality. If a region is known to contain these species it may be more important to have permeability through the wind farm. We define permeability as the capacity of a delimited area to be infiltrated by birds. Permeability was assessed by computing the difference between the actual distance travelled between start and end points i.e. length of simulated trajectory, to the Euclidian distance between these two points. If an area was completely permeable then the distance measures would be the same and the permeability index would be zero however the greater the disparity, the less permeable the area and the more negative the measure of permeability. We investigated the permeability of a 100 km^2 area containing 100 turbines in different configurations. This average turbine density (1 turbine km^{-1}) is similar to that of the Nysted wind farm (1.2 turbines km^{-1}) yet still allowed plausible scenarios to be explored. The scenarios were:

i) Equal spacing across the whole area (inter-row distance = 1 000m; inter-column distance = 1 000m)
ii) Equal spacing within the central 25 km^2 (inter-row distance = 500m; inter-column distance = 500m)
iii) Four blocks containing 25 turbines with equal spacing (inter-row distance = 500m; inter-column distance = 500m)
iv) Random

For each scenario we simulated 100 trajectories. To account for any possible differences due to approach angle, this was repeated from 10 different start locations on an arc 20 km from the centre of the 100 km^2 area, giving a total of 1 000 simulated trajectories. 20 km was considered a suitable distance as this corresponded to the maximum distances from the centre of the Nysted wind farm to start points of the observed data used to parameterise the model. The trajectories were targeted through the centre of the 100 km^2 area, therefore in the absence of the wind farm, all trajectories would cross at the centre point.

5.4 Results

5.4.1 Parameters

We generated 600 000 samples from the posterior distributions of all parameters using 3 chains, a burn-in period of 100 000, and an initial thinning rate of 1 in 100. For all parameters, chains were considered to have converged with Gelman-Rubin convergence statistic values <1.2 however autocorrelation between posterior samples of the parameters d and p was detected. We therefore thinned these samples further by a rate of 1 in 6 to give a final sample size of 500.

A summary of parameter estimates is presented in Table 5.2 and density and trace plots for the parameters are included in Appendix 1. Models 2, 3 and 4 estimated u to be 3.29 radians putting the destination point in a south-westerly direction. Models produced similar estimates for most parameters with overlapping credible intervals. For example, the mean estimate of d was 0.266 (95% CI = 0.253, 0.278) for model 2, 0.239 (95% CI = 0.221, 0.256) for model 3 and 0.244 (95% CI = 0.228, 0.260) for model 4, therefore, models 3 and 4 responded later to the wind turbines than model 2. The mean estimate for k_b was also less for model 2 than for models 3 and 4 therefore models 3 and 4 were responding more suddenly at distance d to the turbines rather than turning away earlier. Parameter p and so the power p-1, was greater for model 2 than for models 3 and 4, therefore the repelling kernel extended further from the turbines for model 2. The shape parameter present in models 3 and 4 was estimated at 25.24 (95% CI = 18.08, 33.75) and 25.22 (95% CI = 18.07, 34.11) respectively. In model 4, τ was estimated to be 81.11. Therefore the individual d_i parameters were distributed with a mean of d and a standard deviation of 0.05 whilst the individual u_i parameters were distributed with a mean of u and a standard deviation of 0.01.

Table 5.2 Mean estimates of parameters within the models (lower and upper bounds of 95% credible intervals). Grey shading indicates where parameters were not included in models.

Parameter	Model 1	Model 2	Model 3	Model 4
u	3.246 (3.240, 3.252)	3.291 (3.285, 3.296)	3.296 (3.292, 3.300)	3.291 (3.269, 3.312)
ρ	0.855 (0.852, 0.859)	0.884 (0.881, 0.887)	0.899 (0.896, 0.901)	0.920 (0.918, 0.923)
c	0.013 (0.011, 0.015)			
d		0.266 (0.253, 0.278)	0.239 (0.221, 0.256)	0.244 (0.228, 0.260)
p	1.482 (1.372, 1.608)	2.701 (2.651, 2.743)	2.599 (2.558, 2.639)	2.621 (2.579, 2.664)
k_b		0.319 (0.306, 0.332)	0.387 (0.374, 0.400)	0.380 (0.369, 0.392)
k_c		-1.350 (-1.524,-1.177)	-1.323 (-1.499,-1.155)	-0.803 (-0.883,-0.727)
shape			25.24 (18.08, 33.75)	25.22 (18.07, 34.11)
τ				81.11 (58.36, 97.63)

5.4.2 Model selection

For 50 replicates of 445 simulated tracks, the mean percentage of tracks entering the wind farm was 42% for model 1 with all 50 replicates producing the same outcome, 0% for model 2, 5% (range = 4 to 5%) for model 3, and 5% (range = 4 to 6%) for model 4. Five of the original data tracks (6%) entered the wind farm. Model 4 included more individual variation than the alternative models and simulated more tracks that were representative of the observed data (Figure 5.2). It produced similar numbers of tracks that entered the wind farm, so model 4 was chosen to simulate tracks for the remainder of the study.

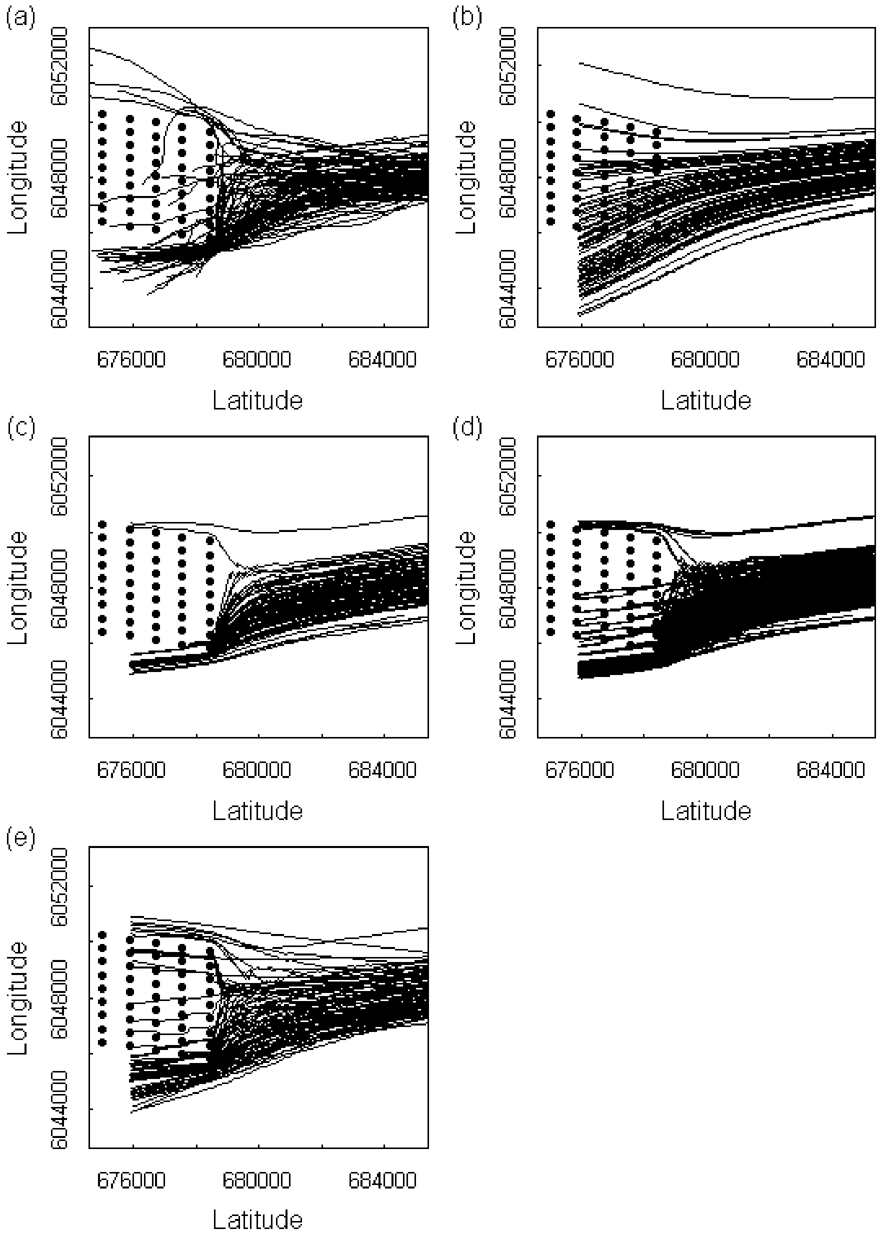

Figure 5.2 Example movement trajectories (a) 89 observed tracks (b) 89 tracks simulated using parameters from model 1 (c) 89 tracks simulated using parameter estimates from model 2 (d) 89 tracks simulated using parameter estimates from model 3 and (e) 89 tracks simulated using parameter estimates from model 4. Black dots denote wind turbines.

5.4.3 Simulations

5.4.3.1 The effect of wind farm dimensions on the number of birds passing between turbines

As the distance between turbines increased so did the proportion of birds travelling between turbines (Figure 5.3). With eight columns of turbines at 200 m spacing, 0% of birds passed between turbines. Increasing the inter-turbine distance to 500 m increased the percentage of birds to 21% whilst a spacing of 1 000 m increased this further to 98%. Increasing the number of columns in a wind farm decreased the number of birds entering the wind farm. A distance of 500 m between turbine rows caused 99% of birds to enter the wind farm when there was only one column of turbines. Increasing the size of the wind farm to two columns decreased this to 83% whilst a further decrease to 21% was seen for 8 columns. The distance between rows of turbines at which over 50% of birds entered the wind farm was 360 m for a wind farm comprising one column of turbines; this increased to 440 m for 2 columns, 520 m for 4 columns and 620 m for 8 columns.

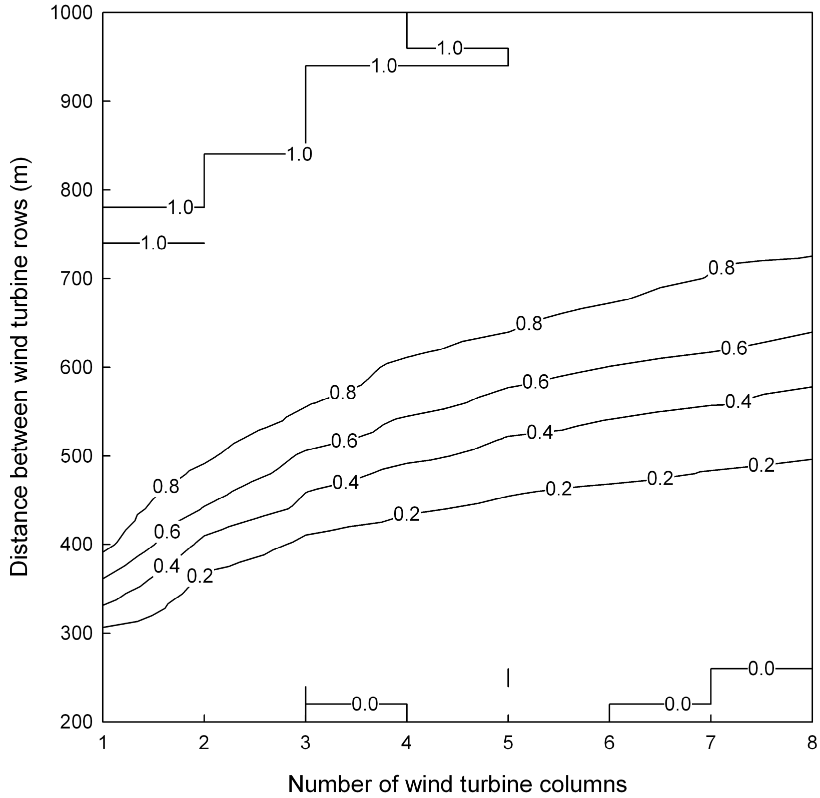

Figure 5.3 Contour plot of the proportion of simulated trajectories entering a wind farm through the central five turbines on the eastern boundary. Wind farms comprised varying numbers of columns, and distances between the turbine rows varied.

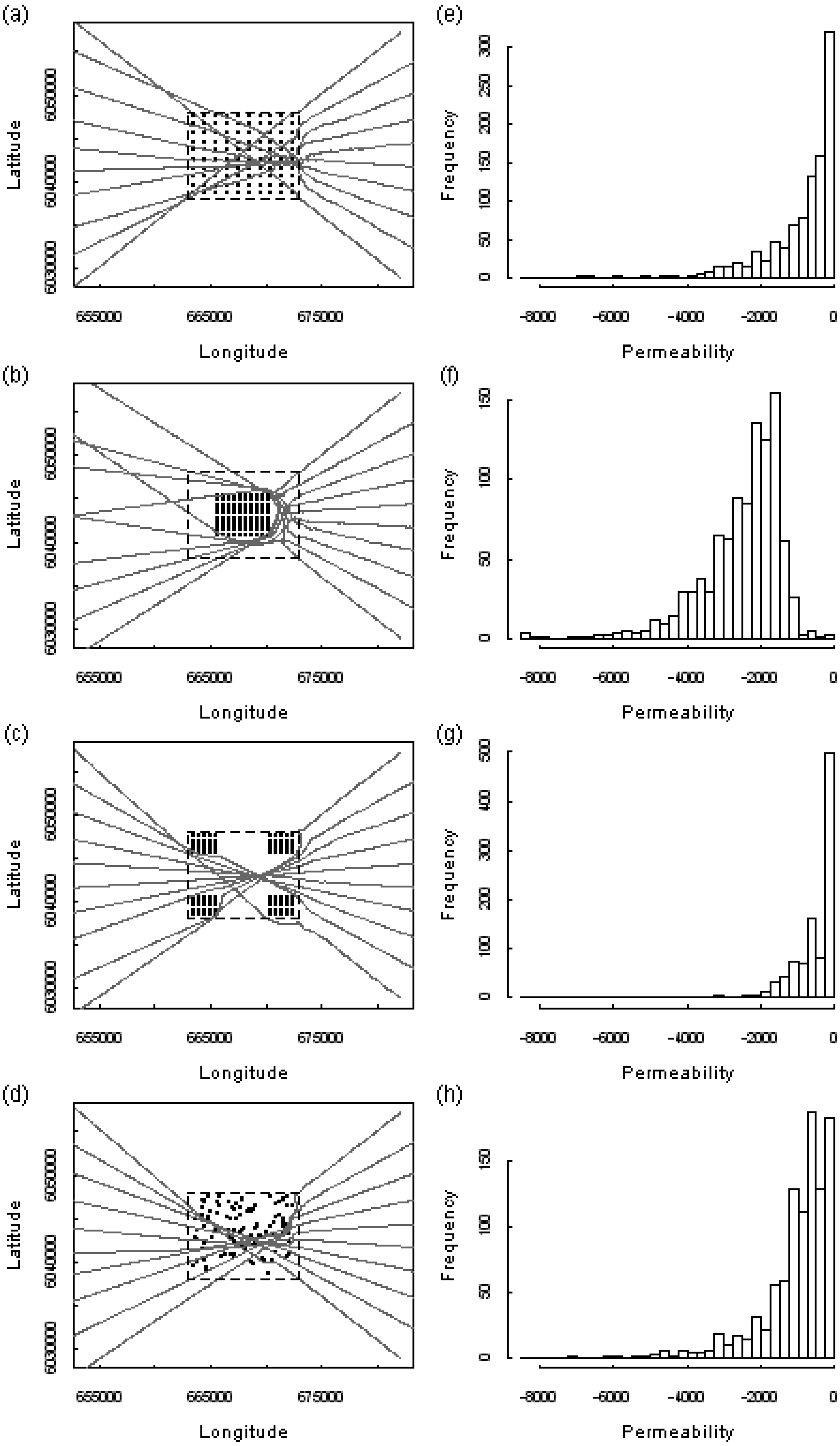

Figure 5.4 Results for different wind farm scenarios. Figures (a-d) Plots of example trajectories. Grey lines represent simulated trajectories. Black dots represent wind turbines. Dashed box represents the 100 km^2 wind farm area. Figures (e-h) Histograms of permeability assuming wind farm scenarios 1-4.

5.4.3.2 The effect of different configurations of turbines on the permeability of an area

The permeability of the area differed for each of the turbine scenarios. Permeability ranged from −8400 to −100 across the scenarios. Scenario 3 (4 blocks of turbines) had the greatest permeability (mean = −510, range = −3500 to −100) whilst scenario 2 (central block of turbines) had the lowest permeability (mean = −2480, range = −8400 to −100) with few trajectories passing between turbines (Figure 5.4).

5.5 Discussion

We demonstrate how data collected on bird movements, post-construction of a wind farm can be used to parameterise avian movement models. This has practical applications in environmental impact assessments of wind farm developments and associated implications for planning. Such models are increasingly vital, since the EU has set targets to produce 20% of energy from renewable sources by 2020 (House of Lords European Committee, 2008) and hence there has been a rapid increase in numbers of proposed wind farm developments. More wind farms leads to greater concerns over the potential adverse effects and their cumulative impacts on wildlife populations, in particular birds. Despite increasing numbers of avian studies on the effects of wind farms, there remains a lack of understanding of the interactions between birds and wind turbines for many species, limiting the ability to predict the likely effects of future wind farms.

Wind farm EIAs invariably record bird movement data in and around the area of the wind farm. The types of data recorded range from visual observations i.e. vantage point watches (Walker et al., 2005) to radar and telemetry data (Desholm and Kahlert, 2005, Bevanger et al., 2008) with the latter becoming more readily available in recent years. This increase in available data associated with individual birds at greater resolution presents an opportunity to investigate the impacts of wind farms on birds using techniques not previously used in this area of research. To date, the majority of data analyses regarding movements of birds around wind farms have been qualitative, e.g. describing species-specific flight heights and abundance, although some studies have taken a more quantitative approach using statistical models for example, to assess golden eagles *Aquila chrysaetos* home ranges and space use (Fielding et al., 2006). One obvious exception is the Band model (Band et al., 2007) which is a mechanistic model to estimate collision risk. The model presented here uses techniques from movement ecology and applies them to the problem of assessing the impacts of wind farms on birds.

Of the models presented, model 4 captured more of the variability in the observed data with simulated trajectories more closely resembling observed trajectories (Figure 5.2). Model 4 incorporated the most individual variation with variation in both the distance at which birds responded to the wind turbines and the bearing to the final destination, suggesting that individual behaviour is an important factor that should be included in movement models. As well as graphically exploring the data we assessed model fit using a test variable (the number of simulated trajectories to enter the wind farm through the central five turbines on the eastern boundary of the wind farm i.e. the middle of the wind farm). Although the models did not reproduce the results from the observed data, this could not have been expected as the test variable was an emergent property of the model, rather than a parameter explicitly modelled. A modification that could improve model fit would be to model turning angle between movement steps rather than bearing, as this would incorporate any autocorrelation between the movement steps.

In this study we provide two example uses of a model, to support the environmental assessment process of wind farms. The first example is where a species is vulnerable to collision. This is relevant in areas known to be hot spots for particular species, e.g. migration corridors or wintering/breeding areas. For such a species, it is beneficial to be able to predict the dimensions and spatial configuration of turbines that would reduce the probability that individuals would fly through the wind farm. By varying turbine row spacing, and column number, we influenced the number of birds entering the centre of the wind farm (Figure 5.3). Also, as the number of rows in an array increased, the greater the inter-turbine distance could be before birds flew between turbines. However, birds continued to pass between peripheral turbines for example, cutting off a corner rather than flying straight through the entire array, suggesting that designs eliminating corners (e.g. by creating rounded edges to wind farms) may be beneficial. Despite technological and engineering constraints on the minimum and optimal proximity of turbines and their placement (Crespo et al., 1999), such knowledge enables wind farm design to balance both engineering and environmental considerations.

For species known to be adversely affected by wind farms, turbines ultimately act as barriers to movements with the additional distance travelled as a result adding to normal energy requirements. This may especially be the case for breeding seabirds, which forage several times a day and may have to commute past wind farms (Masden et al., In Press). To explore the concept of permeability, we considered four different wind farm scenarios (Figure 5.4), and simulated trajectories of birds travelling through the developed area. Permeability was least when turbines were spaced equally within the central area (scenario 2), causing individuals to travel further to reach their destination (Figure 5.4f). Four blocks of turbines (scenario 3) had the greatest permeability and the least variation, suggesting for this example at least, that having several smaller wind farms may have advantages over one larger wind farm. Such a modelling approach provides extensive opportunities to explore different scenarios and the potential impacts on bird movements. This enables a more flexible approach to planning a development that can incorporate not only economic and engineering, but also environmental considerations in the optimal wind farm design.

The results generated by this study are based on several assumptions. We assume that avian avoidance behaviour is manifest at the level of the wind turbine, and although cumulative, the repulsion is not to the wind farm structure as a single entity. This is an assumption of the model and consequently the model predicts that a bird is more likely to avoid an array of wind turbines than to avoid a single row of turbines and this is unlikely for all species. The model was parameterised using data collected from a single species, common eider, and it is unlikely that all species exhibit the same behaviour. However, this is the first attempt at such a model, hence the novel value and with more data for different species, the model could be extended. The model presented describes only changes in movement in terms of latitude and longitude, because the data available were from surveillance radar but it is known that birds may also adjust their altitude in response a wind farm

(Desholm and Kahlert, 2005). Similarly, we model movements around a wind farm surrounded by sea, so topography will have no influence on bird movements, yet this would not be the case for onshore wind farms where birds are likely to respond to a variable landscape.

In conclusion, we demonstrate that avian movement models can be used in the planning of wind farm developments to reduce the negative effects of wind farms on birds. In the future, our ability to parameterise such models depends entirely on data availability. There is a lack of post-construction monitoring and associated data (Stewart et al., 2007, Langston et al., 2006) and it is fundamental that this shortfall is rectified if further progress is to be made in this area.

6 Assessing the cumulative impacts of wind farms on birds: an individual-based model of hen harriers in Orkney

6.1 Abstract

Wind farms are known to impact birds and with increasing numbers of turbines in upland areas of the UK, a species of conservation concern particularly likely to be affected is the hen harrier *Circus cyaneus*. A spatially-explicit individual-based model, including collision mortality rate, effects of direct habitat loss and displacement, was used to examine the cumulative impacts of increasing numbers of wind turbines for a population of hen harriers in Orkney. From an initial population of 70 adults, the model predicted a population decline to a mean of 32 adults (s.d.= 16) after 50 years of simulation, based on currently operational or consented turbines in Orkney, compared to a mean of 11 adults (s.d.= 12) with 16 times that number of turbines. The population response also varied according to turbine location, and the largest impacts resulted from turbines located within 1 km of hen harrier nest sites. Removal of collision mortality from the model showed that the majority of turbine impacts were associated with habitat loss (direct and indirect). Wind turbines impact hen harriers but it may be possible to reduce the effects by considering hen harrier ecology during the planning procedure and/or implementing mitigating measures such as rough grassland restoration.

6.2 Introduction

The EU has set targets to achieve 20% of energy from renewable sources by 2020 to reduce carbon emissions (House of Lords European Committee, 2008). European governments are turning to wind energy, particularly land-based wind turbines which are currently the most developed form of renewable energy. Rapid increases in the numbers of UK wind farms means that the total installed capacity of onshore wind power in Scotland is now greater than hydro power (SNH, 2009). There are also many more wind farm developments in planning and a large proportion of these are for sites in upland areas.

The UK uplands support many bird species of high conservation importance (Thompson et al., 1995). Wind turbines impact wildlife, particularly birds, and although the magnitude of the effects remains uncertain, they can be grouped into three major types: direct mortality due to collision with turbines/infrastructure; physical habitat modification and/or loss due to the footprint of turbines and associated structures; and avoidance responses of birds to turbines (Fielding et al., 2006, Fox et al., 2006, Masden et al., 2009).

One such upland bird species shown to be affected by wind turbines is the hen harrier *Circus cyaneus*, a species of conservation concern, listed both on Annex 1 of the European Community Birds Directive (79/409/EEC) and the Red List of birds in the UK (Eaton et al., 2009). Hen harriers have undergone large changes in abundance and range in the UK, declining to near extinction during the 19th century due to persecution (Sim et al., 2007). However, throughout this period a population of hen harriers persisted in Orkney and now Scotland holds the majority of the UK breeding population. More recently there have been declines in the hen harrier population in Orkney thought to be related to changing land use and an associated reduction in food supply for the birds (Amar et al., 2005, Amar et al., 2008, Amar and Redpath, 2002) and Orkney Mainland Moors has been designated as a special protection area (SPA) for hen harriers. Pearce-Higgins et al. (2009) reported that hen harriers significantly avoided turbines and reduced their flight activity within 250 m of their location. Whitfield and Madders (2005) also concluded that there may be displacement of hen harrier nesting attempts in a 300 m buffer around a turbine.

The aim of this study is to investigate the combined impacts of wind turbines through the effects of collision mortality, direct habitat loss and displacement on population change. The ornithological literature associated with wind farms has increased in recent years, as have the number of wind turbines (Drewitt and Langston, 2006) but the literature has concentrated on individual effects; this study is the first of its kind to assess the integrated cumulative impact of not only multiple effects but also multiple turbines. Here we present a spatially-explicit individual-based model to consider the effects of wind turbines on the dynamics of the Orkney hen harrier population. Including wildlife priorities in landscape management is often vital in species conservation (Larson et al., 2004) therefore we consider the effects of locating turbines according to different management rules.

6.3 Methods

6.3.1 Study area and data collection

Data for this study come from the hen harrier population on West Mainland, Orkney. The Orkney population has been well studied and so much ecological information is available about these birds. Mainland is the largest island in the Orkney archipelago and the majority of Orkney hen harriers nest on West Mainland. All areas of potential nesting habitat were surveyed for breeding harriers between 2001 and 2008, and all nest site locations recorded. Habitat data were obtained from the Land Cover Map of Great Britain 1990 (hereafter LCM) (Fuller et al., 1994). Although a more recent dataset was available (LCM 2000) we chose to use LCM 1990 as Arroyo et al. (2006) suggest that it provides a better representation of Orkney habitats. The habitat data comprised 25m x 25m grid cells, each categorised by the dominant habitat type. LCM has three different categories for rough grass habitat (grass heath, moorland grass and rough/marsh grass), however we followed the methods of Amar et al. (2008) and used only grass heath and moorland grass as a measure of rough grass habitat. The locations of operational and approved turbines were made available by Scottish Natural Heritage (SNH).

6.3.2 The matrix population model

We used an age-structured population model (Lebreton, 2005, Leslie, 1945, Leslie, 1948) to predict changes in the hen harrier population in Orkney. The model was female-based as population growth is ultimately a function of the number of females in a population. The model took the form:

$$\begin{pmatrix} N_{0,t} \\ N_{1,t} \\ N_{2,t} \end{pmatrix} = \begin{pmatrix} 0 & 0 & f \\ s_0 & 0 & 0 \\ 0 & s_1 & s_2 \end{pmatrix} \begin{pmatrix} N_{0,t-1} \\ N_{1,t-1} \\ N_{2,t-1} \end{pmatrix}$$

Where $N_{0,\,t}$ and $N_{1,\,t}$ represent numbers of juveniles 0-1 years and 1-2 years respectively, present in the population at time t, and $N_{2,\,t}$ represents numbers of adults at time t. Changes in population abundance were modelled as a function of s_0 the annual survival of juveniles from age 0 to 1, s_1 the annual survival of juveniles from age 1 to 2, s_2 the annual survival of adults and f, the annual productivity of adults i.e. number of chicks produced. Picozzi (1984b) provided mean annual survival and standard error for adults, and juvenile females aged 0-2; we used the Delta method (Oehlert, 1992) to obtain measures for s_0 and s_1 assuming identical survival in the two year classes and represented survivorship as approximately parameterised beta distributions. Adult productivity (f) was a Poisson random variable, with a mean value dependent on the percentage of rough grass habitat in a 2 km radius of a nest site (see Table 6.1). Parameter estimates were then sampled from these distributions (see Table 6.1).

The matrix population model was used to obtain estimates for the intrinsic population growth rate (λ). For each nest site we obtained an estimate for productivity alongside adult and juvenile survival and used matrix population model methodologies to calculate λ (Caswell, 2001). This was repeated 1 000 times to account for stochasticity in the random variables. Sensitivity analysis was performed to investigate how population growth rate varied in response to manipulation of the different components of the matrix population model. It is generally considered that, unless impractical, conservation management should concentrate on life cycle stages that cause the greatest change in population growth rate. Within the known range of values for juvenile survival (s_0 and s_1), adult survival or adult productivity, we varied two of these parameters whilst holding the third constant at its mean and monitored the change in λ.

Table 6.1. Overview of model input parameters, functions and stochastic processes

Parameter/variable description	Value/equation	Distribution	Reference
Adult female mortality	μ=0.9; σ=0.03	Beta (89, 10)	Picozzi (1984b)
Juvenile female mortality	μ=0.54; σ=0.04	Beta (83, 71)	Picozzi (1984b)
Sex ratio of chicks	50:50	Bernoulli	
Productivity	exp(-1.5+0.11*%RG)	Poisson	Amar et al. (2008)
Age of reproduction	2		
Breeding female foraging range	2-3km	Uniform	Watson (1977)
Breeding male foraging range	3-4km	Uniform	Watson (1977)
Collision risk avoidance rate	0.99		Whitfield & Madders (2005)
Flights at risk height (exc. display)	0.03		Whitfield and Madders (2006)
Display flights at risk height (%)	100		
Display activity	70 secs.hr^{-1}		Amar (unpublished data)
Turbine avoidance (in flight)	1/(1+(1/exp(-1.85*exp(-d/250))))*2		Pearce-Higgins et al. (2009)
Turbine avoidance (nesting)	300 m		Whitfield & Madders (2005)
Display period	30 days		
Incubating period	30 days		
Nestling period	60 days		
Winter period	245 days		
Prey capture rate	0.9 items.hr^{-1}		Redpath et al. (2002)
Weight per prey item	40g		SNH (2004)
Juvenile food requirement (wk 1)	40g		SNH (2004)
Juvenile food requirement (wk 2)	70g		SNH (2004)
Juvenile food requirement (wk 3)	140g		SNH (2004)
Juvenile food requirement (wk 4)	155g		SNH (2004)
Juvenile food requirement (wk 5+)	185g		SNH (2004)

6.3.3 The individual-based model

For clarity and transparency, the model description follows the ODD standard protocol for describing individual- and agent-based models (Grimm et al., 2006). Manipulation of habitat input data was performed using ArcGIS with Spatial Analyst and Hawth's Tools (Beyer, 2004). The model was implemented in Matlab R2009b with the additional Mapping Toolbox. Results reported exclude a burn-in period of five years to eliminate any dynamics attributable to initial conditions.

6.3.3.1 Purpose

The aim of this model is to assess the impact of differing numbers and spatial arrangement of wind turbines on a population of hen harriers, through changes in habitat-related productivity and collision mortality. From a management perspective, this model is intended to provide information to aid the planning of wind farm developments and reduce the occurrence of adverse bird-wind farm interactions.

6.3.3.2 Process overview and scheduling

The model proceeds in discrete time steps, corresponding to a biological year of a hen harrier. The model is female-based and the fate of every female in the population is recorded from birth to death. Male collision mortality was included in the model as a catastrophic event during courtship display and breeding, because the death of either member of a hen harrier breeding pair prior to chick fledging will cause abandonment and breeding failure. We assume one male per nest site. Each time step comprises the following processes: ageing, nest choice, courtship, breeding, and mortality. For an overview of the model see Figure 6.1 and for more information on these processes see Section 6.3.3.4.

6.3.3.3 Initialization

Between 67 and 74 breeding pairs of hen harriers were reported in Orkney in 2004 (Sim et al., 2007) therefore we initialised simulations with 70 individuals. The initial individuals were all female and breeding adults.

6.3.3.4 Sub-models

Ageing: The age of individuals in the model increased by one year with each iteration. Individuals aged 0-2 were considered juveniles and those >2 were considered adults.

Nest choice: At the beginning of each model year adult females were assigned to a nest. Nests were chosen from a set of 305 known harrier nest locations and according to a set of choice rules (Picozzi, 1984a): (1) a nest could not be within

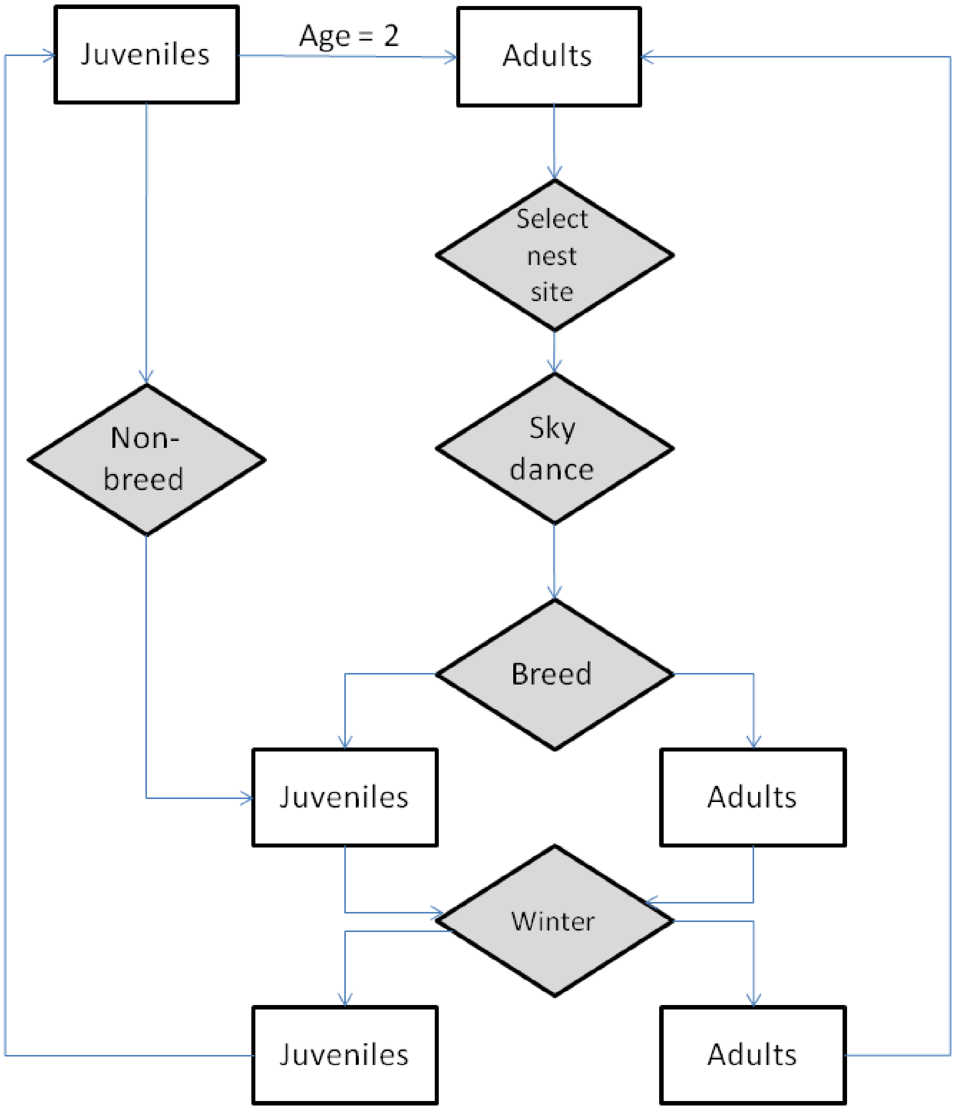

Figure 6.1 A flow diagram of the processes in the model. White rectangles represent life stages and grey diamonds represent processes which may be affected by the presence of wind farms.

200m of another occupied nest; (2) if a breeding attempt was successful the previous year then occupy the same nest with a probability of 0.7 and for those females that move, remain close to the previous nest (mean = 1.32 km, sd = 0.9); (3) if a breeding attempt failed the previous year then occupy the same nest with a probability of 0.3 and for females that move to a new nest, move further away (mean = 2.29 km, sd = 2.41); and (4) if a female is breeding for the first time, then choose a nest near to her natal nest (mean = 5.68 km, sd = 5.43). Nests within 300m of a turbine were not occupied (Whitfield and Madders, 2005). For each adult female, all available nest sites were assigned a probability of occupation according to the distance from the previous year's nest or natal nest and the choice rules. Due to a lack of information, the probability was estimated using a uniform probability density function with parameters mean ± sd, and the lower bound truncated at zero.

Breeding: Females could only reproduce when adult (2+ years) and reproduction was assumed not to be limited by the availability of males. The productivity of a nesting attempt was a Poisson random variable with a mean dependent on the percentage of rough grass (RG) in a 2 km radius of the nest (Table 6.1).

$$\text{Productivity} \sim Poisson\left(e^{-1.5+0.11\text{RG}}\right)$$

Although hen harriers nest in heather, it has been suggested that rough unmanaged grass is critical habitat for these birds probably due to higher abundance of prey (Amar et al., 2008). Hen harriers showed reduced use of habitat near wind turbines (Pearce-Higgins et al., 2009) so we adjusted the proportion of rough grass habitat in each 25m x 25m cell to account for this reduction in use according to distance to the nearest turbine (see Table 6.1). To account for direct habitat loss under a wind turbine we also removed rough grass habitat from cells that contained a turbine.

Annual Mortality: For each simulation year, individual females had a sex- and age-specific probability of survival (see section 6.3.2 and Table 6.1). We used the beta distribution to introduce stochasticity in survival rates and this was suitable as the distribution is bounded by 0 and 1. *Collision Mortality*: Individuals also had an additional probability of mortality through collision with wind turbines. Collision mortality was estimated as a function of time spent flying at risk height, and the number and size of wind turbines within the area of activity (Band et al., 2007) with a 0.99 correction factor for avoidance behaviour (Whitfield and Madders, 2005). Although turbines vary in shape and size, for the purpose of this model it was assumed that the turbines were identical to those in Band et al. (2007) with an ‘at risk height’ between 24 and 76 m. During display, birds spend more time at risk height than at other times of year due to the nature of the sky dancing display. Due to a lack of specific data, it was assumed that all display activity was at collision risk height. This behaviour is limited to within a 1 km radius of the nest site (Madders, 2004). All adult females were assumed to display. Individuals were also at risk from collision whilst foraging during the display period and it was assumed that females were foraging for themselves as males in Orkney rarely provide enough food items for the females (A. Amar pers.comm.). Only males were at risk from collision whilst the females were incubating on the nests. Adults provision their chicks whilst nestling, so the time spent hunting was dependent on food requirements of both adults and chicks. Of these provisioning flights, 3% were assumed to be at risk height (Whitfield and Madders, 2006) and that flight activity would be restricted to within 2-3 km (females) and 3-4 km (males) of the nest (Watson, 1977). Although more time could possibility be spent hunting close to the nest than at greater distances, we assumed uniform use of the whole range. Post-fledging, all birds were considered to be wintering i.e. foraging for themselves and free to range across the entirety of West Mainland. After wintering, juveniles were assumed to forage freely across West Mainland for another year before entering the breeding population at age 2.

6.3.4 Simulation experiments

This study evaluated the population dynamics of hen harriers in Orkney in response to 17 different configurations of wind turbines. These 17 scenarios started from the current situation (5 operational and 2 approved turbines as baseline scenario 1) and covered a range of turbine densities (7 to 105 additional turbines), including precautionary scenarios that intentionally avoid locating turbines in areas with harrier nests and others that targeted areas with harrier nests (Table 6.2). Scenarios comprised the seven original turbines plus additional turbines to make totals of 2, 4, 8, and 16 times the baseline. Additional turbines were sited independently as single turbines according to one of four different management rules: (1) in (25m x 25m) cells more than 2 km from a hen harrier nest; (2) in cells for which there is no more than 10% rough grass habitat within a 2 km radius; (3) in cells more than 2 km from a nest site and with no more than 10% rough grass habitat within a 2 km radius; or (4) within 1 km of a hen harrier nest. Turbines could not be located within 500 m of another turbine, reflecting inter-turbine distances in operational wind farms.

Within the model, wind turbines can affect harrier dynamics through (1) collision mortality and (2) reduction in habitat quality causing reduced fecundity. To disentangle these effects and assess the effect of collision on the population dynamics we simulated the worst-case scenario (scenario 17) in the absence of the collision component. In this simulation, the wind turbines could only affect the hen harrier population dynamics through habitat productivity.

Table 6.2 Summary of scenario ID numbers and management rule descriptions

	Number of turbines (multiple of 7 turbines i.e. current situation)			
Management rule	x2	x4	x8	x16
More than 2km from a nest site	2	3	4	5
No more than 10% rough grass habitat within a 2km radius	6	7	8	9
More than 2km from a nest site & in cells with no more than 10% rough grass habitat within a 2km radius	10	11	12	13
Within 1km of a hen harrier nest	14	15	16	17

6.4 Results

6.4.1 Matrix population model

The intrinsic population growth rate (λ) for the population matrix described in section 6.3.2 had a mean value of 0.98 (sd = 0.1). Varying productivity across its range had a greater effect on λ than did adult survival or juvenile survival (Figure 6.2). Increasing productivity had a greater effect on λ for higher values of juvenile survival.

6.4.2 Individual-based model

The mean number of adult female hen harriers alive in the population varied across the fifty years and with management scenario i.e. number and placement of wind turbines. Scenarios with more turbines had a greater decrease in population size (Figure 6.3a) however the growth rate of the adult population, measured as the change between one year and the next (N_{t+1}/N_t), did not appear to vary greatly between scenarios (Figure 6.3c).

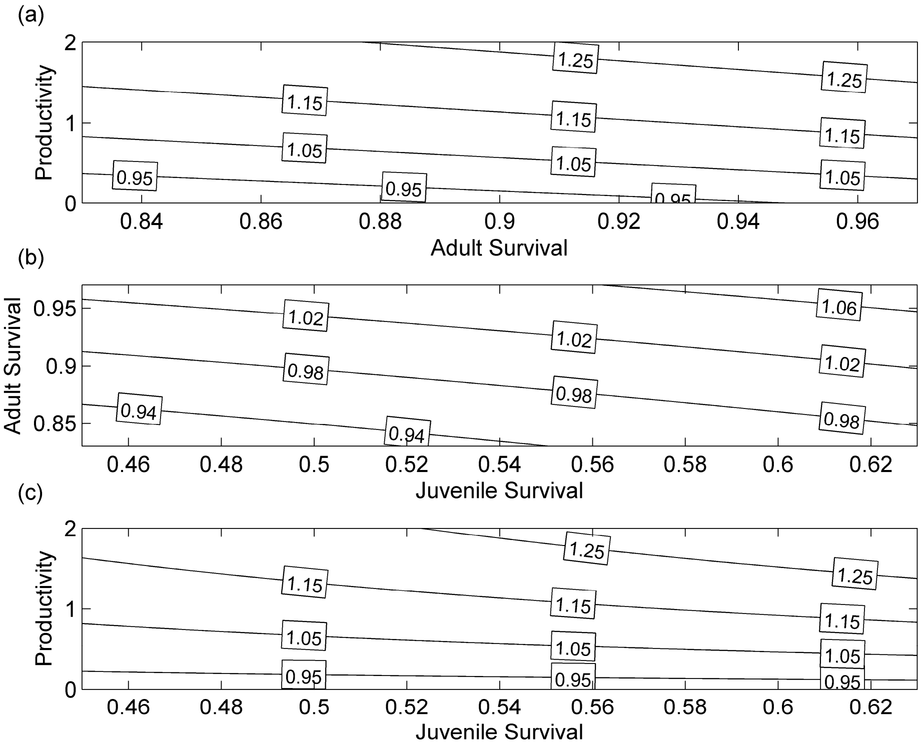

Figure 6.2 Contour plots of population growth rate (λ) as a function of the components of the hen harrier matrix population model. For each plot, the missing component of the matrix model is held constant at its mean value.

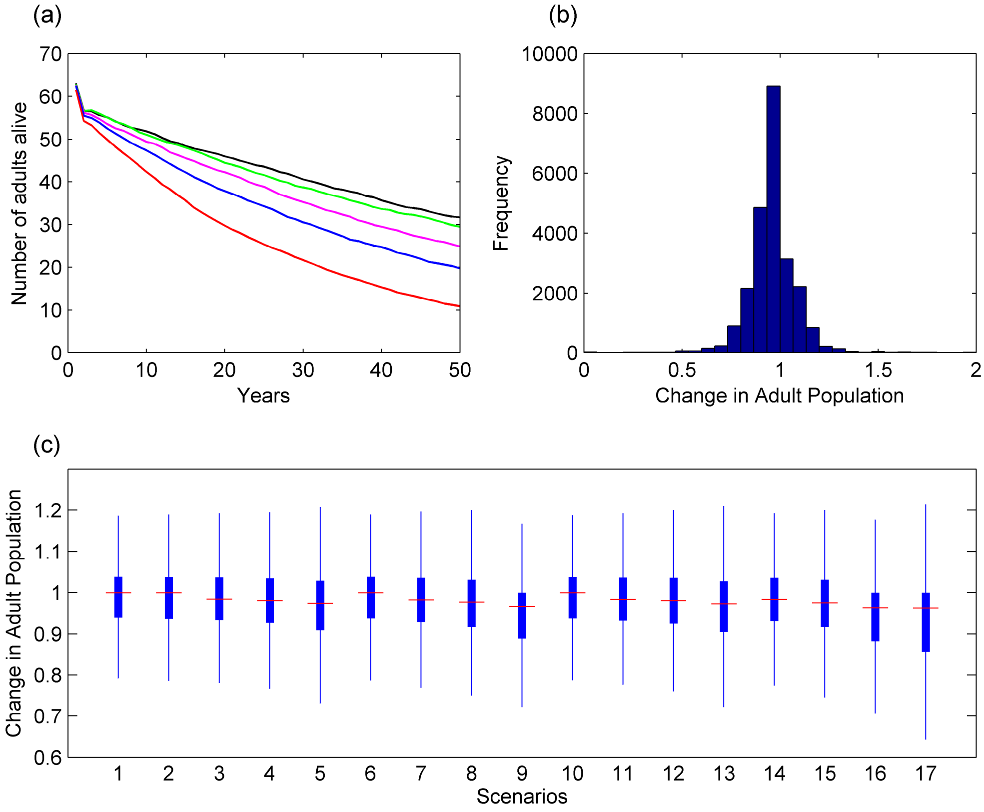

Figure 6.3 Changes in the hen harrier population under different management strategies. (a) Mean number of adults alive calculated from 500 simulations for a selection of scenarios. Scenario 1 (black); scenario 10 (green); scenario 11 (magenta); scenario 12 (blue); and scenario 13 (red). See table 6.2 for scenario ID information. (b) Histogram of the change in the adult hen harrier population (N_{t+1}/N_t) across 50 years for scenario 13. (c) Boxplots of the change in the adult hen harrier population (N_{t+1}/N_t) across 50 years for all scenarios. Horizontal red line denotes the median, blue boxes show the 25th and 75th percentiles and blue lines show 5th to 95th percentiles.

The number of simulations that decreased below 10% of the initial population size (i.e. 7 adults) differed considerably with the number of turbines. Very few simulations (maximum = 5%) declined below 7 adults for scenarios with twice as many turbines than at present (Figure 6.4a) however a larger proportion of simulations showed such declines with 16 times as many turbines (maximum = 98%; Figure 6.4d). With increasing numbers of turbines differences emerged between placement strategies, with the number of simulations with less than 7 adults being greater when turbines were placed within 1 km of nests.

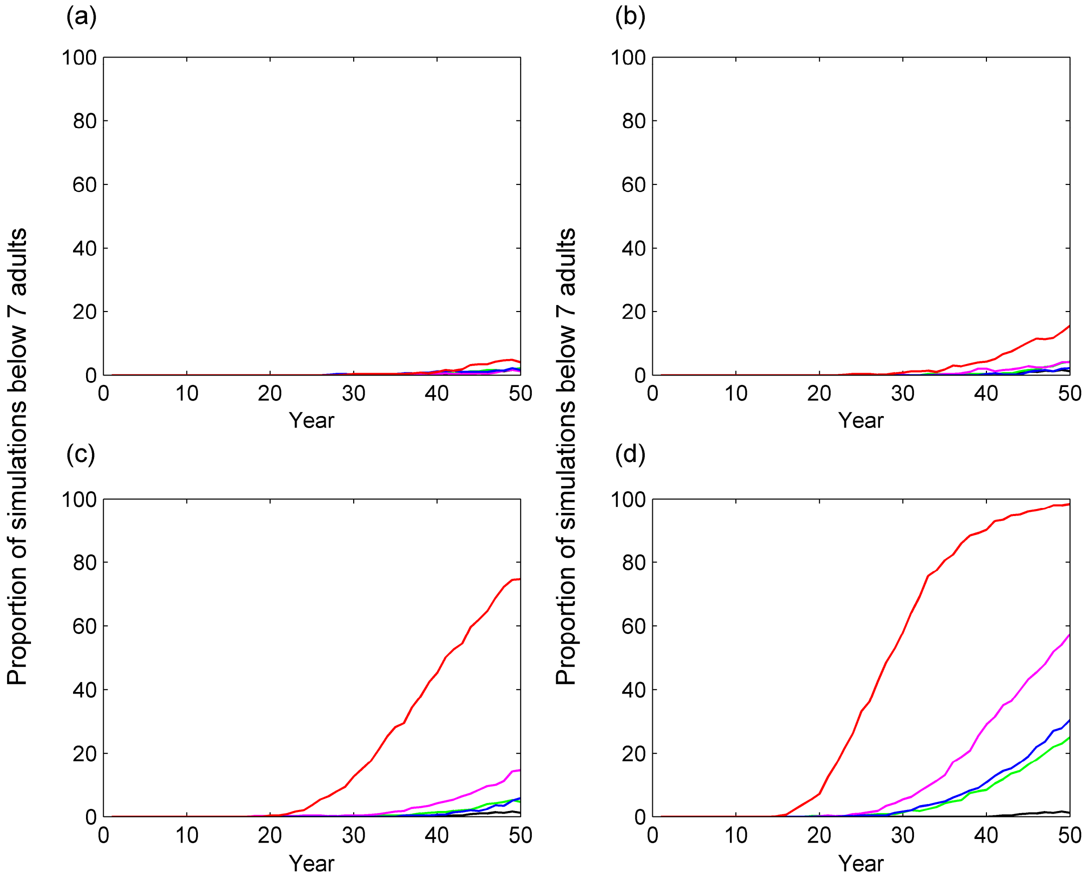

Figure 6.4 Proportion of 500 simulations for which the total number of adults alive was less than 7 i.e. 10% of initial population. (a) 2 times current number of turbines (b) 4 times current number of turbines (c) 8 times current number of turbines (d) 16 times current number of turbines. Line colour denotes management strategy: Black = control i.e. current turbines; green = turbines located >2 km from a nest site; magenta = no more than 10% rough grass habitat within a 2 km radius; blue = turbines located >2 km from a nest site and in areas with no more than 10% rough grass habitat within a 2 km radius; and red = within 1 km of a hen harrier nest.

The number of adult females (referred to as adults hereafter) alive after 50 years of simulation also differed with scenario. More adults were alive for scenarios with twice as many turbines than at present (scenario 2 mean = 28, s.d. = 14; Figure 6.5a) than for those with 16 times more (scenario 5 mean = 12, s.d. = 7; Figure 6.5d). For scenarios with twice as many turbines than at present, there was little difference between the counts of adults alive irrespective of turbine placement. These were also indistinguishable from the control (scenario 1 i.e. current situation). For 4 times, 8 times and 16 times the current number of turbines, the management strategy that placed turbines within 1 km of harrier nests caused the greatest reduction in adults, followed by placement in areas with no more than 10% rough grass habitat within a 2 km radius. The strategy of locating turbines further than 2 km from a harrier nest, and also the combination rule of further than 2 km from a nest site and in areas with no more than 10% rough grass habitat within a 2 km radius produced similar results (Figures 6.5b-d).

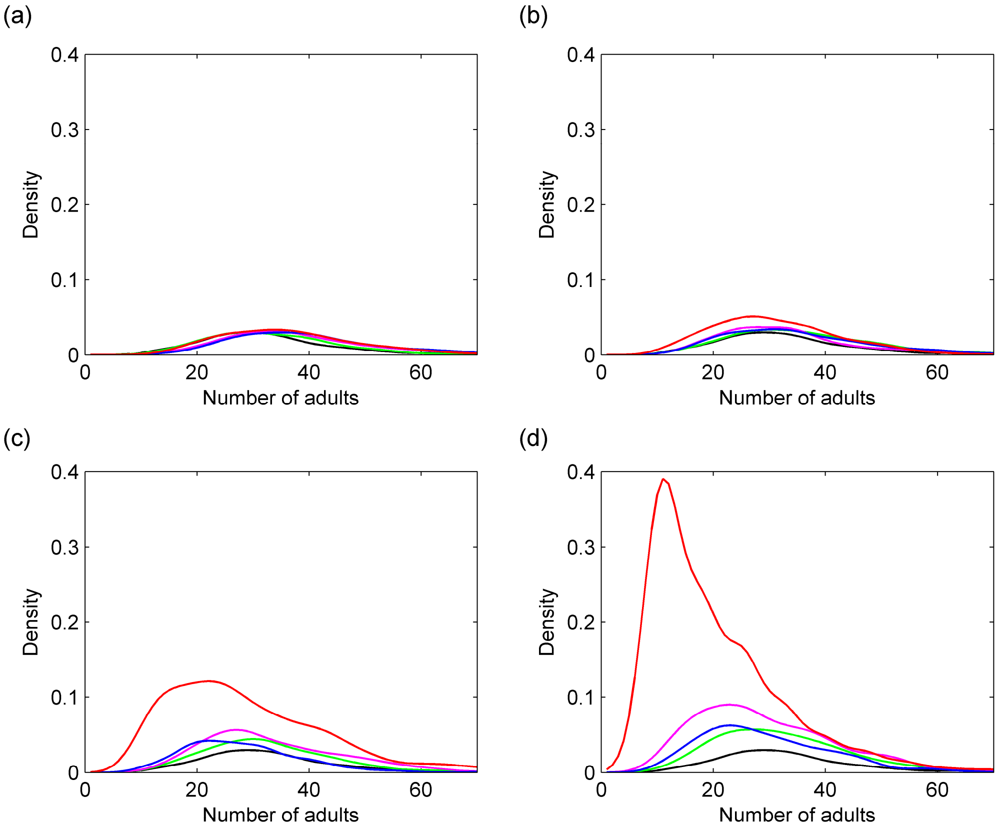

Figure 6.5 Counts of adult harriers alive at the end of simulations i.e. year = 50. (a) 2 times current number of turbines (b) 4 times current number of turbines (c) 8 times current number of turbines (d) 16 times current number of turbines. Line colour denotes management strategy: Black = control i.e. current turbines; green = turbines located >2km from a nest site; magenta = no more than 10% rough grass habitat within a 2km radius; blue = turbines located >2km from a nest site and in areas with no more than 10% rough grass habitat within a 2km radius; and red = within 1km of a hen harrier nest.

Figure 6.6 shows the effects of removing collision mortality from the model. For the control scenario 1 i.e. current situation, there was little effect of collision on the mean population trajectory (Figure 6.6a), population growth rate (Figure 6.6b), the number of simulations in which the number of adults decreased to 10% of the initial population (Figure 6.6c) or the number of adults alive at the end of 50 years of simulations (Figure 6.6d). However, there were differences when comparing scenario 17 (16 times the current number of turbines placed within 1 km of hen harrier nests) with and without the effect of collision mortality. The mean number of adult females alive at year 50 was 1.35 for scenario 17 with collision and 4.37 when collision mortality was excluded (Figures 6.6a and 6.6d). Also, the proportion of simulations that had less than 7 adults alive at year 50 was 98% for scenario 17 but 77% for scenario 17 without collision.

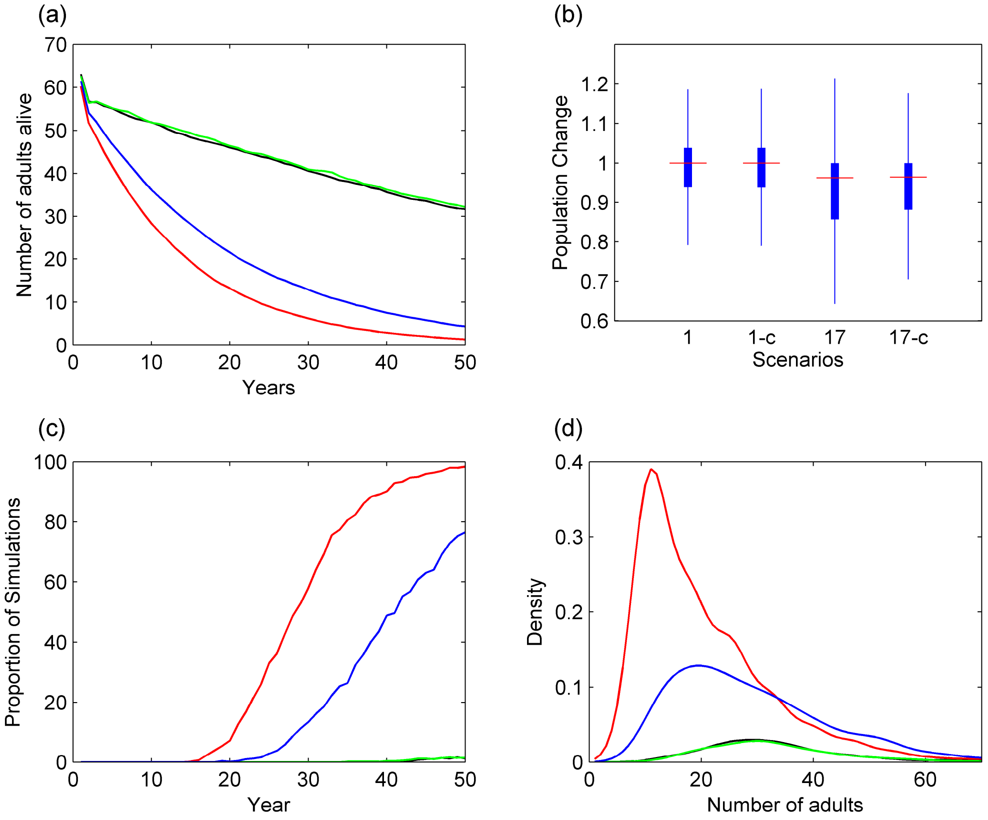

Figure 6.6 The effect of collision mortality. (a) Mean number of adults alive calculated from 500 simulations for a selection of scenarios. (b) Boxplots of the change in the adult hen harrier population (N_{t+1}/N_t) across 50 years for scenarios 1 and 17 including and excluding collision mortality (scenario –c). Horizontal red line denotes the median, blue boxes show the 25th and 75th percentiles and blue lines show 5th to 95th percentiles. (c) Proportion of 500 simulations for which the total number of adults alive was less than 7 i.e. 10% of initial population. (d) Counts of adult harriers alive at the end of simulations i.e. year = 50. Line colour denotes scenario: Black = control i.e. current turbines; green = control without collision mortality; red = 16 times current number of turbines placed within 1km of a hen harrier nest; and blue = 16 times current number of turbines placed within 1km of a hen harrier nest without collision mortality.

6.5 Discussion

The UK government has set a target to provide 20% of its electricity from renewable sources by 2020, and in Scotland there is a more ambitious target of 50%, of which a large proportion will come from onshore wind installations. With increasing numbers of wind farms comes concern over the impacts on birds and this has led to an increase in the ornithological literature associated with the topic. However, the majority of studies have concentrated on assessing single effects of a wind farm i.e. habitat loss, collision, or behavioural responses rather than an integrated cumulative assessment. In this study we developed a model to assess the summed effects of both collision mortality and habitat loss (direct and indirect) on a population of hen harriers in a way that incorporates both factors. We also assessed the cumulative effects of varying numbers of wind turbines on the population.

A matrix population model is one way to represent and describe the structure of a population. The hen harrier model predicted a mean population growth rate (λ) of 0.98, suggesting a declining population. Using a similar female-based population model, Fielding et al. (2009) estimated a declining Orkney population with a mean growth rate of 0.92 and Amar (2001) also reported λ to be less than 1. Sensitivity analysis of the matrix population model highlighted annual adult productivity as the most important parameter affecting population growth of the hen harrier population in Orkney (Figure 6.2). This suggests that the most effective management actions to contribute to maintaining a healthy population would concentrate on increasing productivity.

In this study we also investigated the effects of differing numbers of wind turbines on the hen harrier population using an individual-based model. In the model, wind turbines could affect the population through collision mortality (affecting survival), and direct and indirect habitat loss (affecting nest productivity i.e. number of chicks). On removal of collision mortality from the model, the estimated population change remained almost the same for scenario 1 (control) but there was a change when considering the worst-case scenario 17 (16 times the number of current turbines within 1 km of harrier nests) (Figure 6.6). The majority of the impact of wind turbines on the population therefore comes from the effect of habitat reduction on the number of chicks produced, rather than an effect on adult survival. Hence, management measures to mitigate the effects of wind turbines should concentrate on the link between habitat and productivity, for example habitat enrichment or restoration of rough grassland (Amar et al., 2008).

Increasing numbers of wind farms seem inevitable given the international commitment to reduce CO_2 emissions, therefore the challenge is to locate them in a manner that ensures the least environmental impact. We investigated the impacts on hen harriers of increasing the number of turbines in Orkney Mainland compared to the current situation of 5 turbines installed and 2 more that have received planning consent. Increasing numbers of turbines sped the hen harrier population declines but doubling present turbine numbers (to 14 turbines) produced results very similar to the control scenario (current). This implies that installing twice the number of turbines, if not more, in Orkney than at present is unlikely to adversely affect the hen harrier population providing the turbines are located in the least damaging places.

We also investigated the effect of different management strategies (rules by which to locate turbines). As expected, placing turbines within 1 km of hen harrier nests had the greatest population impact and this was more evident with increasing numbers of turbines. The combined strategy of locating turbines more than 2 km from a nest site and in areas with no more than 10% rough grass habitat within a 2 km radius, and the strategy of locating turbines more than 2 km from a nest site caused the least impact. The combined strategy had less impact than that of installing turbines in areas with no more than 10% rough grass habitat within a 2 km radius, perhaps because not all harrier nests were in high rough grass areas, despite the importance of rough grass for both foraging and breeding. Any strategy for planning turbine locations in Orkney should consider hen harrier ecology, as it will likely increase the number of turbines that can be installed without impacting the hen harrier population.

Despite differences in the scenarios i.e. number and placement of turbines, the annual change in λ, the adult population (N_{t+1}/N_t), was similar across scenarios with the mean value of λ never being less than 0.95 and rarely greater than 1.2 (Figure 6.3c). There was a trend for the mean value of λ to decrease and the variability in λ to generally increase with increasing wind turbine numbers.

However, when looking at the size of the adult female population after 50 years, there were obvious differences between the scenarios (Figures 6.4 and 6.5). This therefore highlights the fact that the time-scale of assessment is important when considering the impacts of wind turbines on birds, especially long-lived species (Masden et al., 2010) and the conclusions drawn could be very different if an annual or 3 year time-scale was used, as is typically the case in environmental impact assessments, rather than a 25 or 50 year period.

The model and results presented in this study are based on several assumptions that should be considered. One assumption was that a hen harrier would not extend its foraging range/area to compensate for any reduction in habitat. This is only likely if the population is large and at carrying capacity. An extension of foraging range may therefore alter the relationship between rough grass and productivity, for example the radius in which to measure the percentage rough grass may need to be extended to 3 km; this would differ from the 2 km suggested by Amar et al. (2008) reducing the true magnitude of the effects to less than reported here. Movements of wintering birds are not known so we assumed that over-wintering individuals were at liberty to move around the whole island. If the true area used is less, then the number of turbines encountered by an individual will vary accordingly, as will the collision risk. Similarly, we assumed that breeding birds foraged evenly across their foraging range and if this assumption was false then collision risk would vary. However, even if the majority of foraging time was spent in areas with few turbines, a bird may still interact with turbines on return to the nest. Another assumption of the model was that breeding pairs are monogamous; however in Orkney there is an unknown proportion of the population which is polygamous causing a greater affect of male collision mortality on the population (catastrophic events) since if one male dies, two breeding attempts may fail. The Orkney population also has a proportion of non-breeding females but due to a lack of information the model assumed that all females had the opportunity to breed. If a proportion of the females were non-breeding then they would probably forage across the entire study area, interacting with more turbines but over a larger area and this would change the collision risk for these females. The number of individuals in the hen harrier population in Orkney has been declining therefore intra-specific competition is unlikely limiting population growth. For this reason, density-dependence was not incorporated in the model. However, if density-dependence was acting on the population and reducing the per capita birth rate i.e. productivity, then it would act as a buffer against the impact of the wind turbines.

In conclusion, we presented an individual-based model to assess the cumulative impacts of wind turbines on a population of hen harriers in Orkney, through the effects of collision mortality, and direct and indirect habitat loss. Increasing the number of wind turbines reduced the population of hen harriers over a 50 year period, however the magnitude of the effects depended on where turbines were located. This suggests that the planning procedure for wind turbines should recognise and take account of the ecology of hen harriers to minimise impacts. The bird sensitivity map (Bright et al., 2008) was the first tool to provide wind farm developers in Scotland with ornithological information at the planning stages to reduce the impacts of wind farms on birds of conservation priority. The model presented here has the potential to further this and provide information at a greater spatial resolution. Habitat loss contributed more to modelled impacts on the population than collision mortality, suggesting that mitigation measures would be most successful if they included positive management for rough grass habitat.

7 General Discussion

Wind turbines are becoming a common feature in both terrestrial and marine environments. Although wind turbines have benefits in terms of providing renewable energy and reducing the national and global carbon footprint, the environmental impacts of these structures have not yet been fully investigated. In this thesis I have explored the topic of assessing the cumulative impacts of wind farms on birds.

7.1 Thesis overview

Cumulative impact assessment is a legislative requirement of EIA (Environmental Impact Assessment) but too frequently it has been tacked on to the end of assessments as an afterthought. Reasons for this are numerous but a recurring theme is the lack of definition and guidance regarding the context and contents of a cumulative impact assessment. In **chapter 2** we developed a conceptual framework to promote transparency within assessments. The core concept being that explicit definition of impacts, actions and scales of assessment are required to reduce uncertainty in the process of assessment and improve communication between stakeholders. Only when it is clear what has been included within a cumulative assessment, is it possible to start to make comparisons between developments, and assess the costs and benefits. Despite much concern over the impacts of wind farms on birds, and results from previous studies detecting impacts, a recurring theme of this thesis has been that for the instances examined, by and large, the impacts of wind farms appear slight. For example, in **chapter 3** we found that the additional distance travelled by migrating eider due to avoidance of the Nysted offshore wind farm was trivial in terms of predicted changes in body mass. To induce a detectable reduction in body mass required the birds to fly around the equivalent of one hundred Nysted wind farms. Using the data available on flight paths of eider around Nysted we also developed a model to describe the movement process in relation to the wind turbines (**chapter 5**) and therefore to predict the impacts of different numbers and configurations of turbines. For breeding seabirds i.e. daily commuting birds, the impact in terms of daily energetic expenditure (DEE) was species-specific due to differing morphologies and foraging characteristics (**chapter 4**). However, the impacts were not as pronounced as had been anticipated with the energetic cost of flight increasing by no more than 30% DEE for an additional 10 km travelled. For hen harriers on Orkney (**chapter 6**) a large number of additional wind turbines were also required before differences from the current baseline were observed. With only twice as many turbines than at present i.e. 14, there was no detectable impact on hen harriers.

7.2 The importance of the affects of wind farms on birds

Despite studies documenting the effects at the individual level, the scientific foundation for concern over population level effects remains weak. Some extreme and high profile problem cases such as Smøla in Norway and the Altamont Wind Resource Area in California have arisen where wind farms were poorly sited but as the ornithological literature associated with wind farm studies grows, there are more examples that show effects of wind farms on birds. However there are also studies that show variation in results and others that demonstrate no detrimental effects. For example Devereux et al. (2008) reported that turbine location did not affect the distribution of wintering farmland birds and Rothery et al. (2009) also found no effect of turbines on the numbers or behaviour of seabirds after the construction of turbines. These studies provide information to enable effective planning of future wind farms to avoid avian impacts.

Marris and Fairless (2007) suggested that the impact of wind farms on birds has been overstated, quoting facts such as “the annual death toll attributable to an average wind farm in the United States is 3% of a bird!” They also highlight the fact that “America’s birds die at a rate of 40 000 a year due to turbines but die in the hundreds of millions due to domestic cats.” As true as these statements may be, they mask the genuine concerns regarding wind farm and bird interactions but they do highlight the fact that effects and impacts should be set in the wider context. In the case of the common eider in chapter 3, the Nysted offshore wind farm is not the only way that human activities impact on the eider population, for example the annual Danish hunting bag is 30 000 – 70 000 birds (Christensen, 2008). Equally, seabirds such as kittiwakes (chapter 4) are impacted by fisheries and experience related population declines (Furness, 2003) whilst hen harriers (chapter 6) have been severely persecuted throughout history (Sim et al., 2007). However, in principle, all of these human activities or actions can be anticipated and integrated using the cumulative impact assessment framework presented in chapter 2.

Due to a lack of knowledge of the impacts of wind farms on birds and an inability to convert effects at the individual level into population impacts, the precautionary principle is often adopted and many wind farm applications have been rejected on ornithological grounds. However, with more studies and data becoming available and considering the need for renewable energy, it may be time to assess the true cumulative impact of wind farms and include not only the costs but also the benefits (Inger et al., 2009), particularly with reference to climate change. As presented in this thesis, there are data available on the costs i.e. impacts of wind farms, although more data would allow a much greater understanding; however much more research is needed to explore the potential benefits. For example, it has been suggested that marine wind farms may create artificial reefs and effective marine reserves or no-take zones which would benefit prey species and their seabird predators (Petersen and Malm, 2006). A complete cumulative impact assessment should also consider the impact of climate change on birds in the absence of the wind farms i.e. wind farms will mitigate the effects of climate change and benefit birds. There are inherent difficulties with such large scale assessments as they require large data sets such as time series data including both pre- and post-construction of a wind farm and an understanding of population variability in the absence of the development. However, it may be possible for a few sites such as the Firth of Forth where long term data are available, for example from the Isle of May seabird studies.

The results from the research presented in this thesis suggest that the impacts of wind farms on birds may not be as pronounced as originally suspected, however it is possible that I was not able to capture the entire cumulative impact. Both in chapter 3 and chapter 4 I assumed that the individuals were healthy and in good condition, with an average body mass. To date, no studies have assessed the cumulative impact of multiple actions i.e. fishing, climate change, and a wind farm. If a population is already impacted by a fishery then the individuals may not be in good condition, due to a lack of prey items, and therefore any loss in body mass attributed to an increase in energy requirement to fly around a wind farm may have a greater effect. It is likely to be in these populations already stressed or at their physiological limits where we will observe the greatest impacts. Also, most studies that have attempted to assess the impacts of wind farms on birds have done so over short temporal scales.

However, as stated in chapter 2, it is critical that assessments are conducted over appropriate time scales both for the processes likely affected and the ecology of the species at risk, as highlighted in chapter 6. For the population of hen harriers in Orkney, the impacts of the wind turbines were most obvious when assessing the changes over a 50 year period rather than observing annual changes. By using modelling techniques, it is possible to predict the impacts of wind farms on bird populations over these longer temporal scales, however the validity of such models depends on the availability of data, and there remains a general lack of post-construction monitoring data.

7.3 Future research

Although there are increasing numbers of studies available on the effects of wind farms on birds, few have linked effects at the individual level to population impacts. Much of this stems from a fundamental mismatch of the scales at which wind farm developers, authorities and researchers operate. To date, developers have been responsible for collecting the majority of data available on the effects of wind farms and in few cases has there been an obligation to collect post-construction data. Consequently the data available are limited to short time scales. Although it is the responsibility of the developer to determine the impacts of their wind farm proposal, it is now clear that the data available from these environmental impact assessments are not enough to determine population level effects over longer time scales. Therefore there is a need for a framework to be developed and funding to be sourced to begin research and data collection to answer these population-level questions.

To understand the longer term, population effects we need to understand the details of how exactly birds interact with wind turbines. Much research has been devoted to this topic, especially monitoring and predicting collision mortality risk of birds colliding with turbines, although a great deal remains unknown. Models such as the Band collision model (Band et al., 2007) can provide estimates of the likely collision risk to a bird from a wind farm but there has been no adequate validation in the field of such models, mainly due to the fact that collision events are so rare. This requires urgent attention if collision risk is to be included in population models, as in chapter 6, because errors may propagate throughout the simulations with the risk of producing results that are incorrect by orders of magnitude. If it is not possible to validate these collision models, an alternative approach may be to pose the question, “How many birds have to die through collision before there is a population impact?” and assess the likelihood of such a mortality event. For example, if it required 500 guillemots to be killed per day to produce a population impact and only 50 birds per day were observed at a site, a population risk could be inferred. However, such an approach requires a definition of ‘population’ which is clear and unambiguous, and this is rarely the case.

Definitions of populations usually refer to a group of individuals of the same species living together in a particular area but the area is often loosely and/or arbitrarily defined (Berryman, 2002). A more formal definition of population states that ‘the area should be sufficient to permit normal dispersal and migration and in which numerical changes are largely determined by birth and death processes’ (Berryman, 2002). This definition gives rise to smaller areas connected through migration and dispersal being recognised as local or sub-populations and the larger, overall area as a metapopulation. The metapopulation concept (Levins, 1969, Hanski, 1991) is currently absent from assessments of the impacts of wind farms on seabirds and needs to be incorporated. At present, a seabird colony or SPA is often considered a population in its own right when in fact it may be a sub-population of a larger metapopulation. With movements of birds between colonies, wind farms will not impact colonies in isolation; therefore assessments should concentrate on the impacts and status of the metapopulation rather than single colonies.

Before we can fully assess the population impacts of a wind farm, there is a requirement for data on the movements of birds and habitat use so the connectivity between a wind farm and breeding, roosting, and/or feeding sites can be determined. Without this movement information it is not possible to assign an individual at a wind farm to a specific population and therefore link the effects on individual birds at a wind farm site, to changes and impacts on a population. This is viewed as a particularly important question by authorities such as Scottish Natural Heritage because they are responsible for the maintenance of, for example, seabird populations in Special Protection Areas (SPAs), which must remain in favourable conservation status i.e. populations must not decline.

There also remains a lack of understanding of the cumulative impacts on bird populations from multiple wind farms or indeed the impacts of wind farms in combination with other human activities such as fishing. Longer term studies and post-construction monitoring would enable questions such as these to be addressed because over a series of years a population of seabirds for example may be exposed to a fishery, a wind farm or both. However, if it is not possible to disentangle the effects of multiple wind farms, we should at least acknowledge the uncertainty. One approach could be to construct a model including all actions that may impact on birds, accept that we are not omniscient, but explicitly incorporate all uncertainty. Such a model could provide estimates for the number of wind turbines that could be developed in an area and although initially conservative, the uncertainty could be reduced as more data becomes available on the effects. Such risk-based models are frequently used in fisheries management and were implemented by the International Whaling Commission (IWC) in their revised management procedure as a method to set quotas in the absence of complete data.

Within this thesis and in other studies much work has been conducted on the impacts of wind farms on birds. However, the analysis techniques used are not restricted to birds and could be extended to other species affected by wind turbines such as bats. Similarly, the issues are not restricted to wind turbines. For example, it is plausible that marine mammals may avoid tidal turbines as birds avoid wind turbines, causing them to alter their movement paths, and should there be an energetic cost to such changes there may be individual or population level impacts. The rate of development in the marine renewable sector (wave and tidal power) is increasing with ten sites recently announced for development in Scotland. The lessons learnt so far in the development of wind energy should not be wasted but rather used as stepping stones for this up and coming marine renewable industry. Likewise, good examples of the use of techniques such as the IWC’s revised management procedure should not be ignored simply because they are not obviously linked to renewable energy.

7.4 Closing remarks

Throughout this thesis I have used a variety of statistical analyses to assess the impacts of wind farms on birds. Of the techniques used, I believe the models of chapter 5 (movement model) and chapter 6 (individual-based model) have the potential to provide insight into the impacts of wind farms. For example, in chapter 5 I was able to predict the impacts, in terms on additional distance travelled, of four different designs of wind turbine arrays. An assessment of this kind would not be possible without the use of models. Similarly, in chapter 6, I used an individual-based model to compare the impacts of different numbers and placement strategies of turbines on a hen harrier population, highlighting those strategies which had the least impacts. These types of models provide a valuable tool which can be used in the planning process to predict likely environmental impacts, therefore allowing integration and consideration of avian impacts in the ultimate wind farm design.

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8 Appendix 1

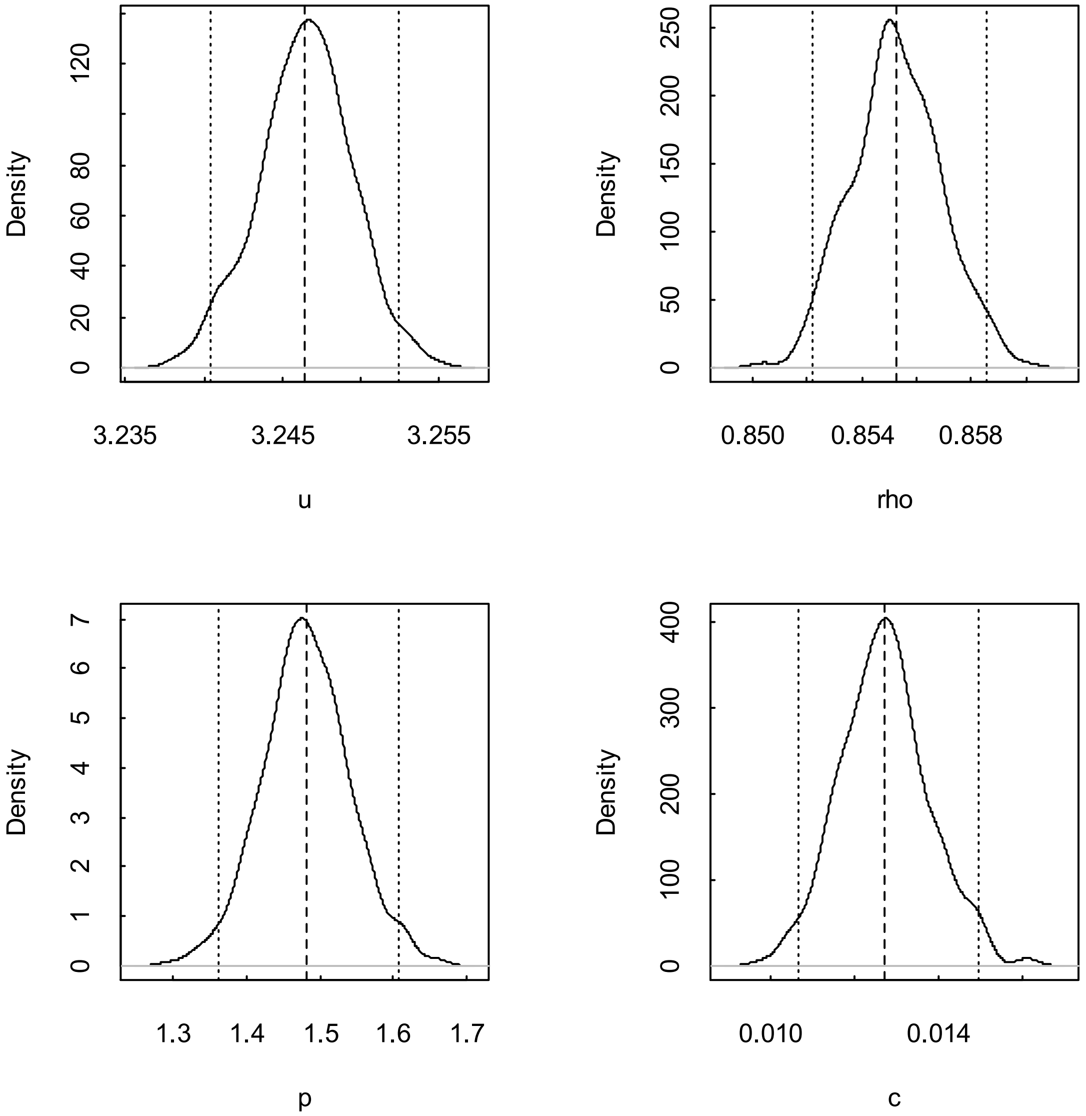

Figure 8.1 Posterior densities for estimated parameters in model 1. The vertical dashed lines are the means of the distributions whilst dotted lines denote 95% credible intervals.

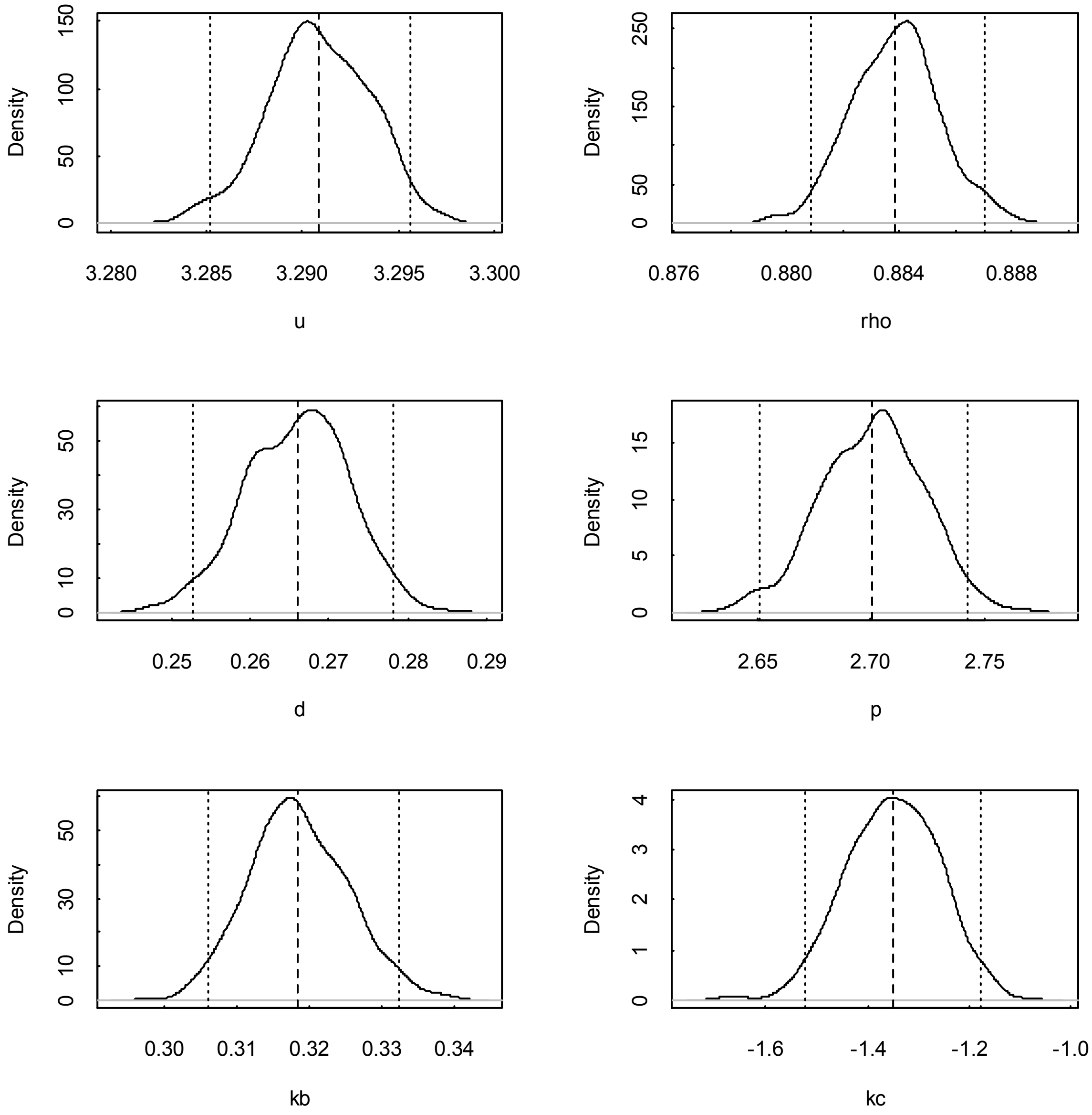

Figure 8.2 Posterior densities for estimated parameters in model 2. The vertical dashed lines are the means of the distributions whilst dotted lines denote 95% credible intervals.

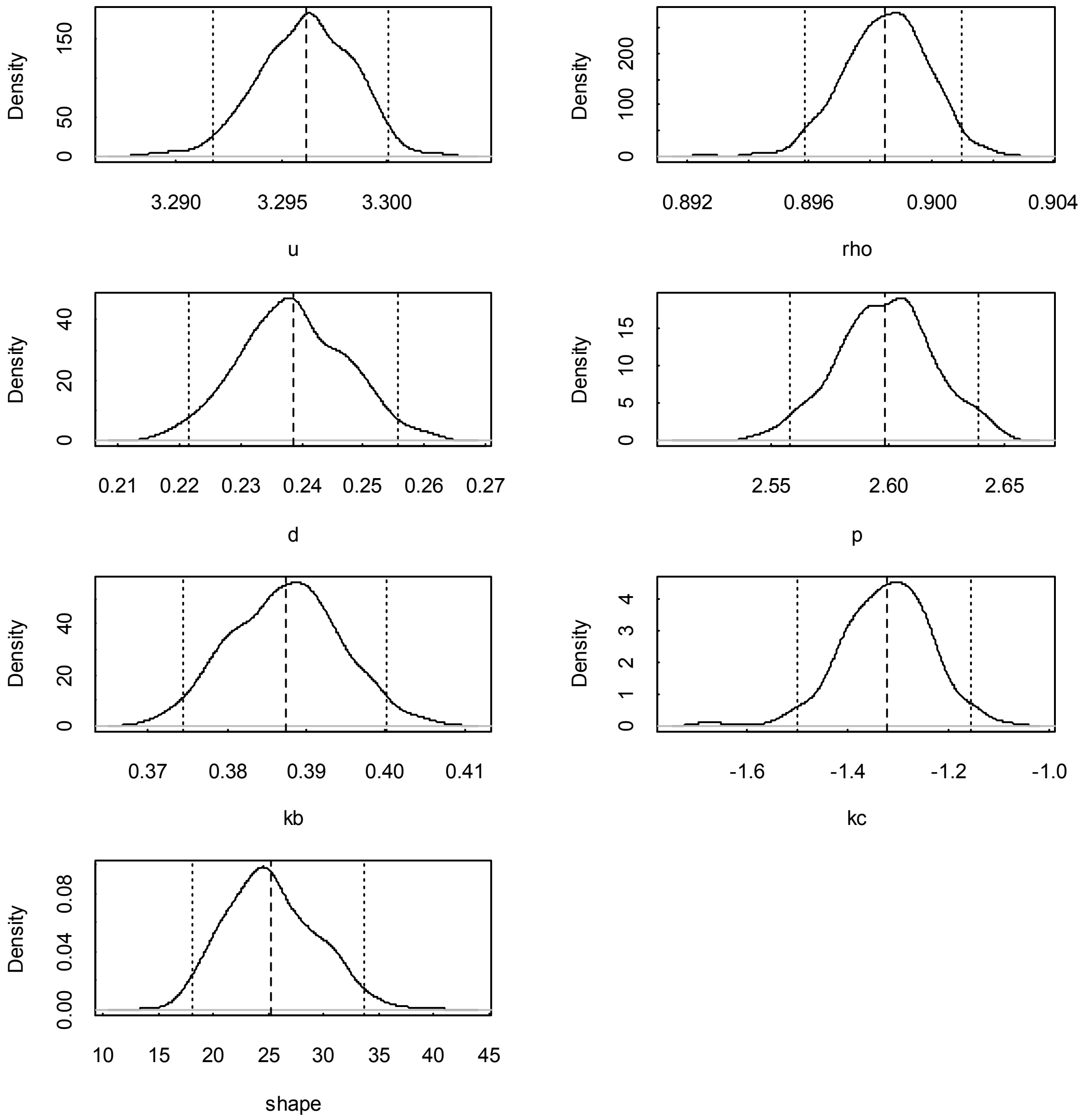

Figure 8.3 Posterior densities for estimated parameters in model 3. The vertical dashed lines are the means of the distributions whilst dotted lines denote 95% credible intervals.

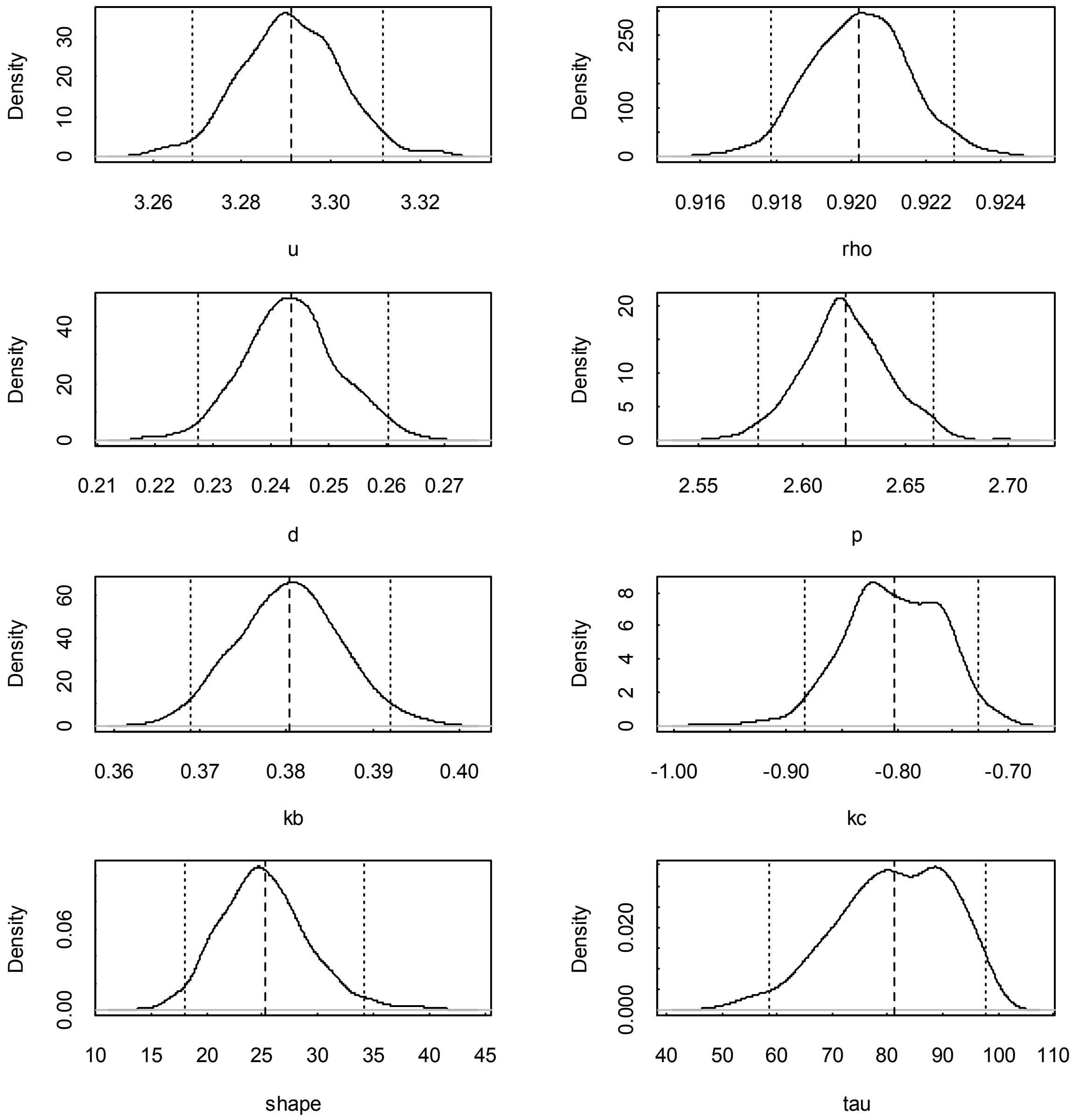

Figure 8.4 Posterior densities for estimated parameters in model 4. The vertical dashed lines are the means of the distributions whilst dotted lines denote 95% credible intervals.

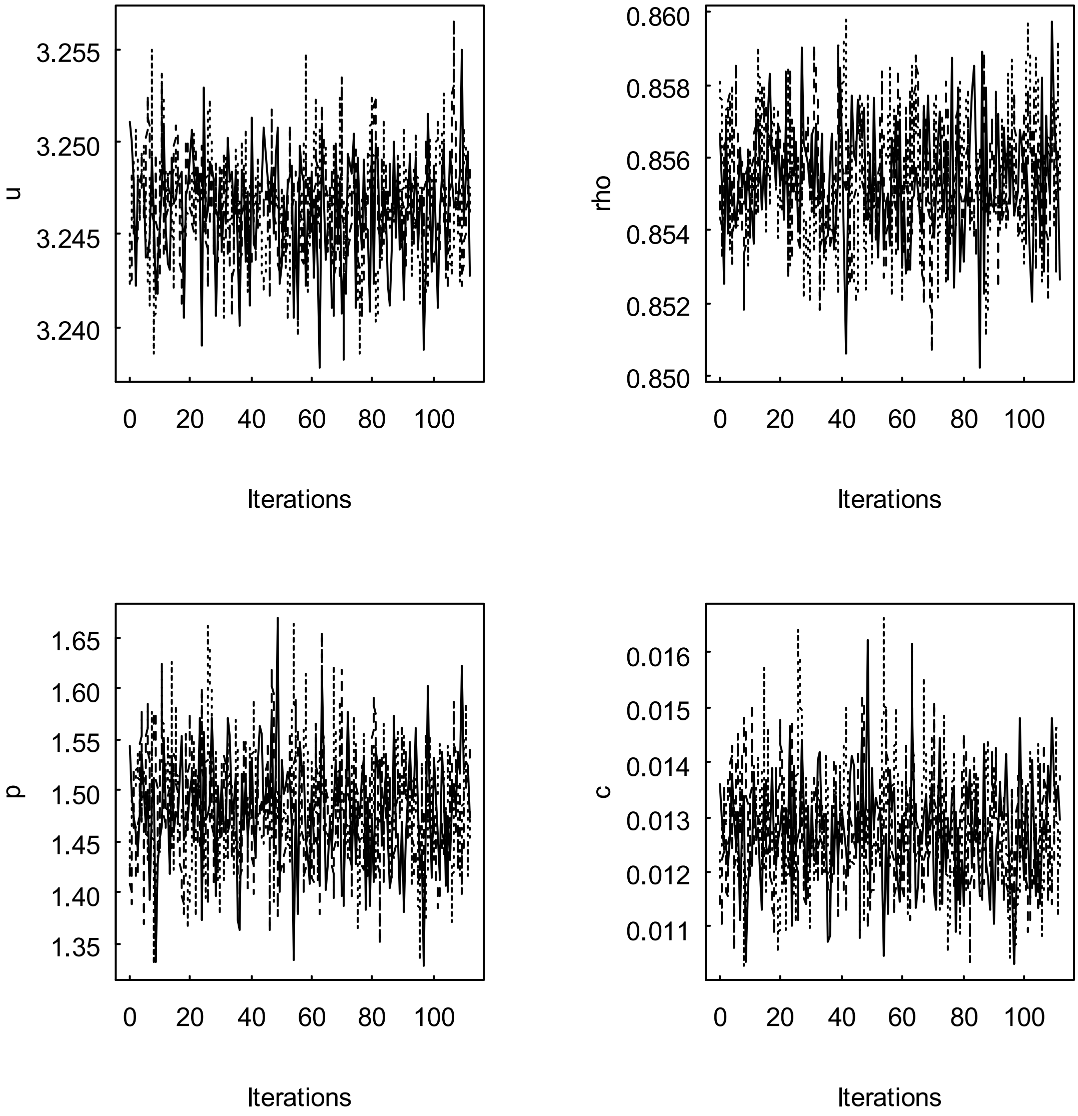

Figure 8.5 Samples of the three MCMC chains for model 1 demonstrating mixing of chains. Line type varies with chain.

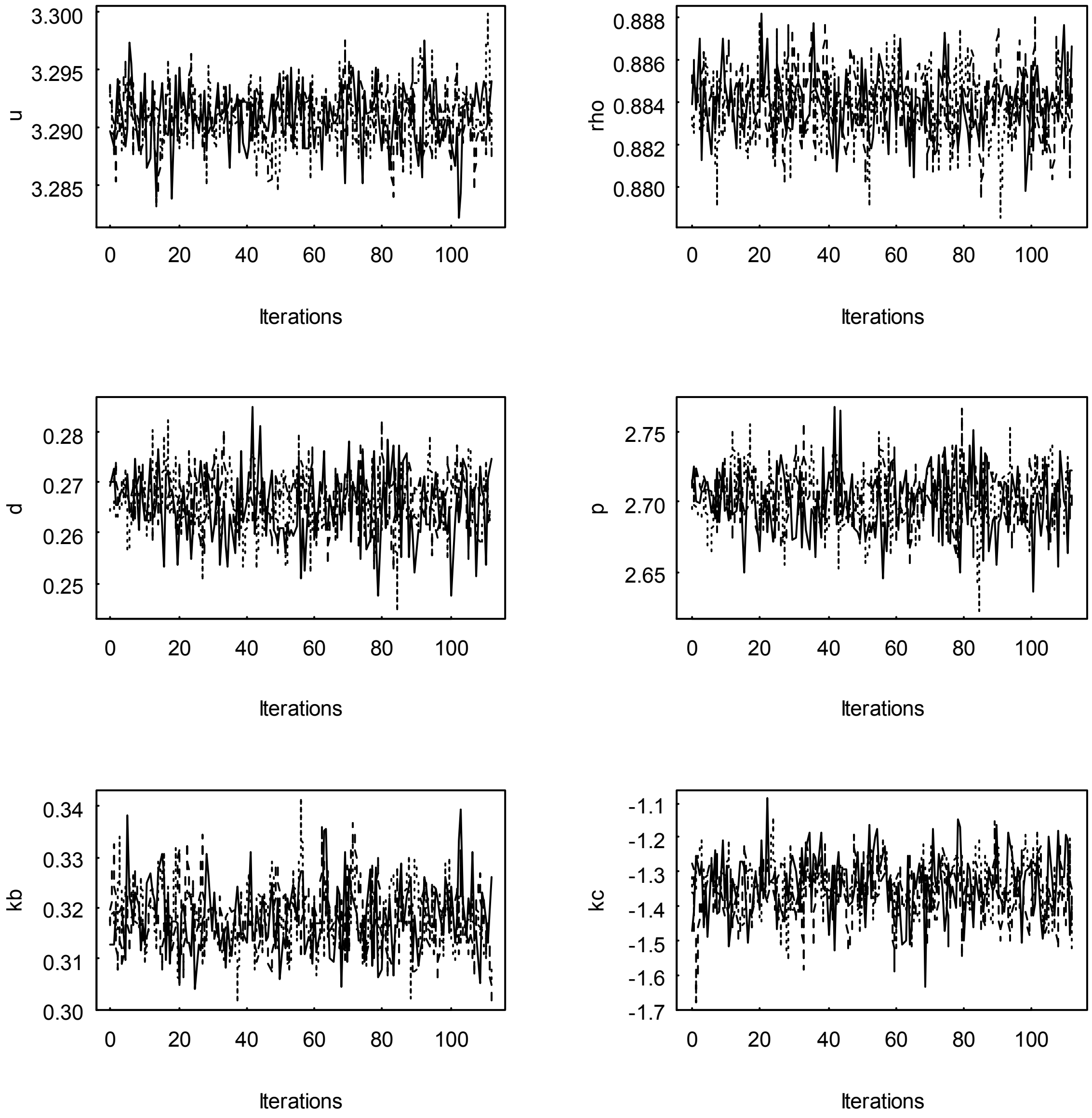

Figure 8.6 Samples of the three MCMC chains for model 2 demonstrating mixing of chains. Line type varies with chain.

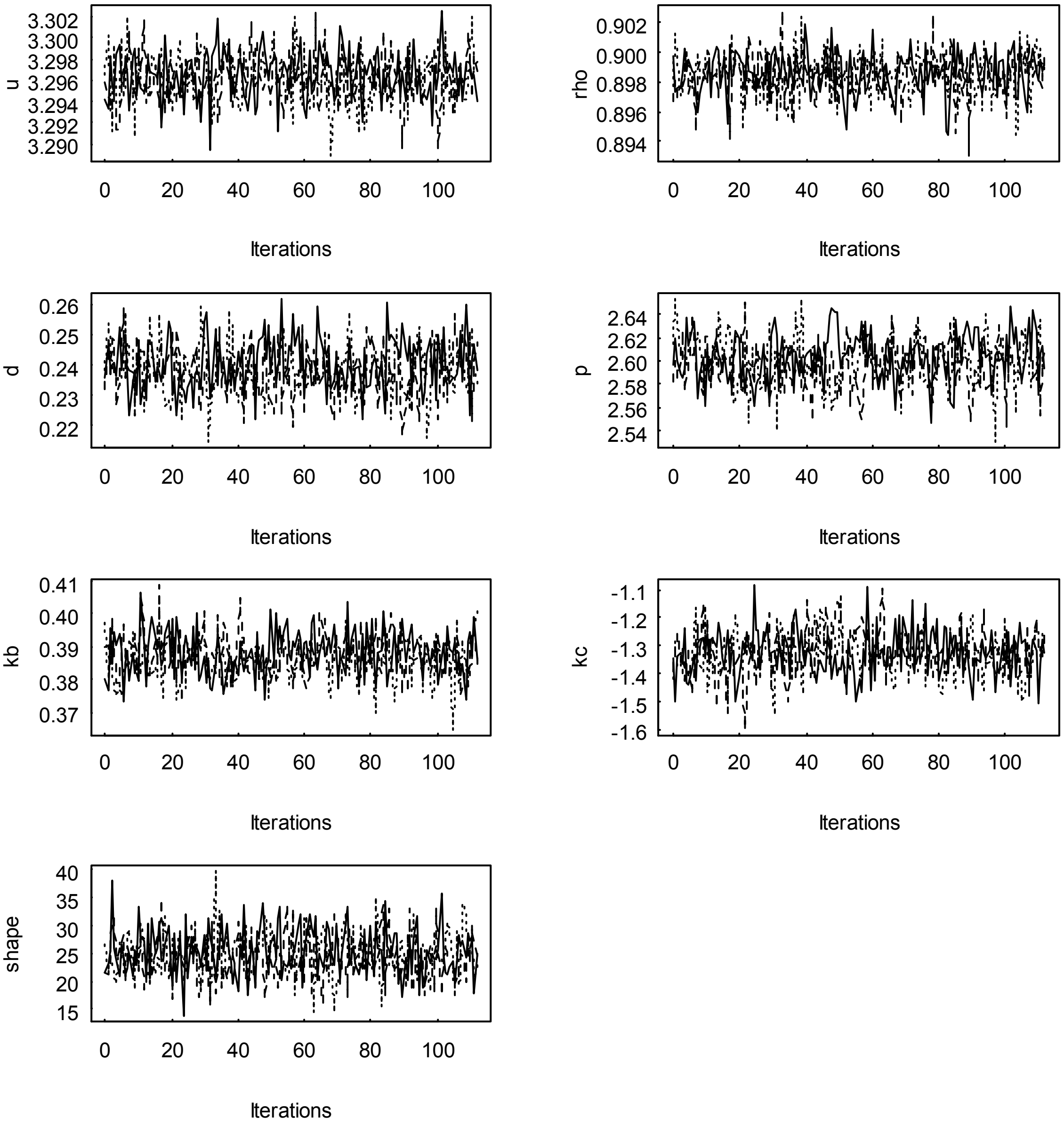

Figure 8.7 Samples of the three MCMC chains for model 3 demonstrating mixing of chains. Line type varies with chain.

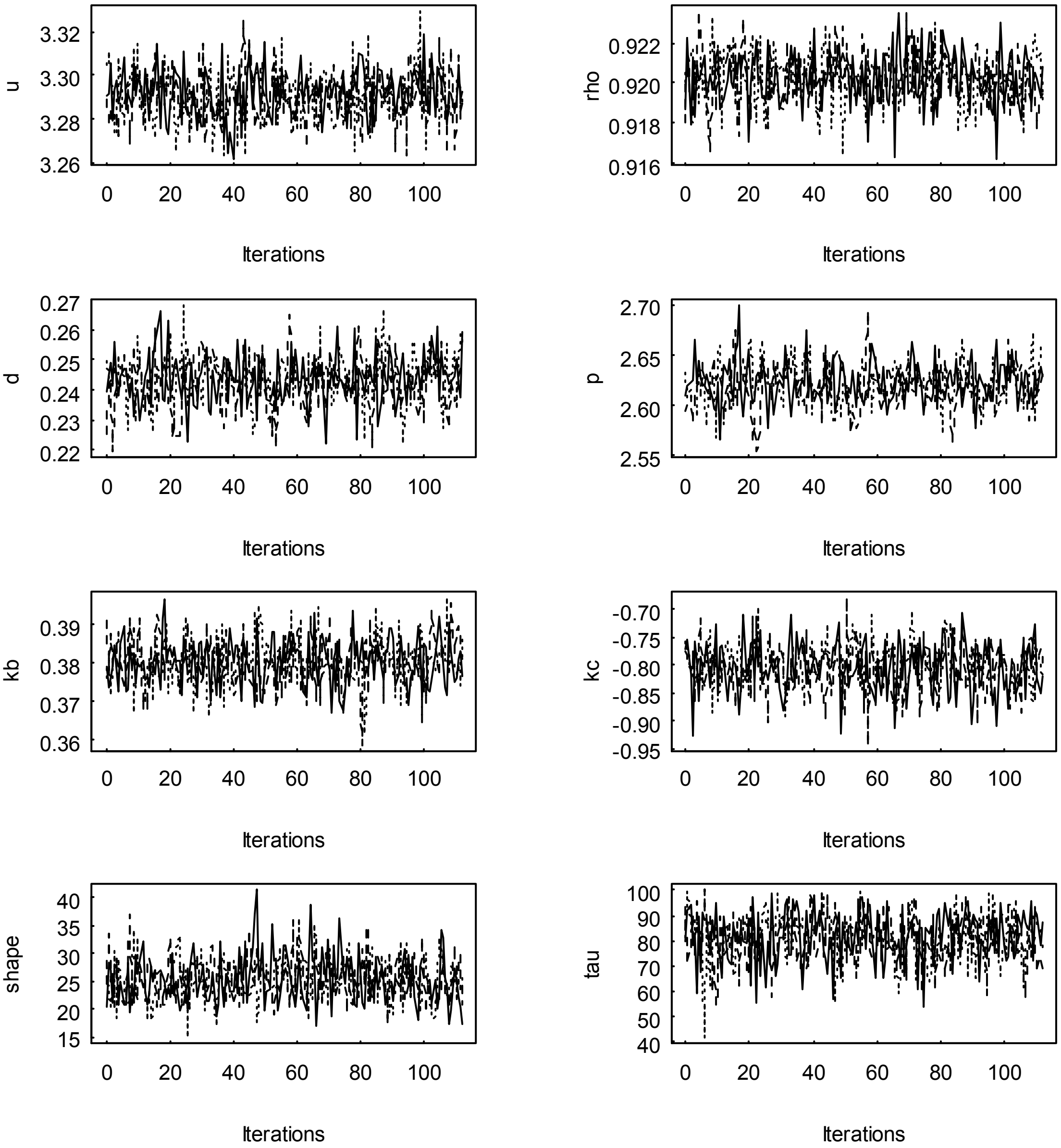

Figure 8.8 Samples of the three MCMC chains for model 4 demonstrating mixing of chains. Line type varies with chain.