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The Avoidance Rates of Collision Between Birds and Offshore Turbines

A S C P Cook, E M Humphreys, E A Masden and N H K Burton

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The avoidance rates of collision between birds and offshore turbines

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Report of work carried out by the British Trust for Ornithology[1]
in collaboration with the Environmental Research Institute[2]
on behalf of the Marine Scotland Science

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EXECUTIVE SUMMARY

1. The selection of appropriate avoidance rates for use in collision risk models at offshore windfarms is often a key part of the Environmental Impact Assessment process. Ideally, these avoidance rates should reflect the behavioural responses of birds to turbines. However, they are often used as a 'fudge-factor' to incorporate aspects of model error. The situation is further complicated by a lack of data for marine birds and offshore windfarms. As a consequence, present guidance is based on values that have been derived for terrestrial species at onshore windfarms. This study reviewed data that have been collected from offshore windfarms and consider how they can be used to derive appropriate avoidance rates for use in the offshore environment. Aims of the study were five-fold:

    - To produce definitions for the types and scales of avoidance;
    - To review current use of avoidance rates;
    - To review and critique existing avoidance behaviour studies and any derived rates;
    - To provide summary avoidance rates and a total avoidance rate for each priority species/species group based on the evidence available at present;
    - To undertake an assessment of the sensitivity of the conclusions reached to inputs and conditions under which they were collected.

    The study focussed on five priority species – northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull and great black-backed gull – whose behaviour and distribution make them particularly prone to collision with offshore turbines.

    Definitions (section 3)

2. A key hurdle to defining appropriate avoidance rates for use in the offshore environment has been a lack of clear, agreed definitions of avoidance behaviour. Therefore, the first step of this review was to define the different scales at which avoidance behaviour may occur. Three categories of behaviour were initially defined – macro-, meso- and micro. Micro-avoidance refers to 'last-second action taken to avoid collision, which is considered to occur within 10 m of the turbine rotor blades. Meso-responses reflect all responses to individual turbines occurring between the base of each turbine and the windfarm perimeter (defined as 500 m from the base of the outermost turbines). Macro-responses reflect all behavioural responses to the presence of the windfarm that occur at distances greater than 500 m from the base of the outermost turbines. Avoidance rates are typically derived by comparing observed collision rates to the number of collisions that would be expected in the absence of avoidance behaviour, considering all bird movements within the perimeter of the windfarm. Consequently, calculations do not usually consider whether any avoidance action takes place at the meso- or micro-scale. It was thus also necessary to consider a fourth category, within-windfarm avoidance, which combines micro-avoidance and meso-responses.

Current use of avoidance rates (section 4)

3. The avoidance rates used with collision risk models have shown substantial variation over time. Initially, very high values, often based on incorrect interpretations of data, were used. Since the earliest environmental impact assessments, there has been a broad tendency to follow standard guidance with avoidance rates of 0.95 and more recently, 0.98 used. However, in light of recent evidence from both on- and offshore windfarms these values are coming under increasing scrutiny from developers and their consultants.

Macro-responses (section 5.1)

4. As with micro-avoidance and meso-responses, the evidence for macro-responses to the presence of a windfarm was typically inconsistent for gulls. Studies designed to look at potential displacement effects reported both evidence for attraction and for displacement and others no significant response at the limited number of sites which were available for consideration. Thus, for gulls, the balance of evidence suggests a macro-response of 0 (i.e. no attraction to or avoidance of the windfarm). However, the response of northern gannet to the presence of windfarms appeared to be more consistent, with strong avoidance evident at several sites, although again it was not always clear whether the macro-response was a result of barrier effects or displacement. Based on the evidence currently available, it is suggested that a macro-response rate of 0.64 is a suitable precautionary value for northern gannet.

Micro-avoidance (section 5.2) and meso-responses (section 5.3)

5. Data for micro-avoidance and meso-responses were extremely limited. No clear and consistent patterns were evident for any of our five priority species. For this reason, it was not possible to derive micro-avoidance or meso-response rates for these species.

Within-windfarm avoidance (section 5.4)

6. A total of 20 sites were identified as having sufficient data to derive within-windfarm avoidance rates by comparing observed collision rates to those expected in the absence of avoidance behaviour. Of these, nine were considered to have data of sufficient quality to estimate robust within-windfarm avoidance rates to be calculated using the Band (2012) collision risk model. Within-windfarm avoidance rates were derived for use with both the basic Band model (Options 1 and 2), that assumes that birds are distributed evenly within the rotor-swept area of a turbine, and with the extended Band Model (option 3) that uses a continuous flight height distribution to estimate collision risk at different points within the turbines rotor-swept area. Based on these data within-windfarm avoidance rates of 0.9959 (± 0.0006 SD) and 0.9908 (± 0.0012 SD) were derived for herring gull for use with the basic Band model and extended Band model respectively. Similarly, within-windfarm avoidance rates of 0.9956 (± 0.0004 SD) and 0.9898 (± 0.0009 SD) were derived for large gulls for use with the basic Band model and extended Band

model respectively, and rates of 0.9921 (± 0.0015 SD) and 0.9027 (± 0.0068 SD) derived for small gulls also for use with the basic Band model and extended Band model respectively. Within-windfarm avoidance rates of 0.9893 (± 0.0007 SD) for the basic Band model and 0.9672 (± 0.0040 SD) for the extended Band model were derived for all gulls. Insufficient data were available to calculate a within-windfarm avoidance rate for northern gannet. (Note, where we report the standard deviation around the derived within windfarm avoidance rates, this relates variability between sites and not to uncertainty in the model input parameters. Estimating the contribution of the model input parameters to the uncertainty associated with the derived avoidance rates requires a more detailed understanding of the real range of values associated with each parameter than is available currently.)

Sensitivity of derived within-windfarm avoidance rates (section 6)

7. The sensitivity of within-windfarm avoidance rate values to model input parameters was also assessed and it was found that the final derived values were most sensitive to assumptions about the proportion of birds at collision risk height. However, it was also found that sensitivity to input parameters declined as the number of flights through a windfarm increased.

Recommended total avoidance rates (section 7)

8. Whilst we have estimated within-windfarm avoidance rates to four decimal places, current guidance from SNH is that expressing avoidance rates to more than three decimal places is unwarranted (SNH 2013). Given the inherent uncertainty in the data we feel that this is a sensible approach to apply to total avoidance rates. For this reason, we round within-windfarm avoidance rates down to three decimal places when deriving recommended total avoidance rates. For gulls the balance of evidence suggests a macro-response of 0 (i.e. no consistent attraction to or avoidance of the windfarm). Consequently, the recommended total avoidance rates for these species are equal to the within-windfarm avoidance rates. Therefore, avoidance rates of 0.995 for herring gull, lesser black-backed gull and great black-backed gull and 0.992 for black-legged kittiwake are recommended for use with the basic Band model. Based on the evidence available, it is suggested that the total avoidance rate for northern gannet is unlikely to be lower than that for all gulls. Assuming a macro-avoidance rate of 0.64, this would reflect a within windfarm avoidance rate of 0.9703. We acknowledge that this is precautionary, but in the absence of more species-specific data, we feel it is appropriate. Hence, an avoidance rate of 0.989 for northern gannet is recommended when using the basic Band model. For the extended Band model, avoidance rates of 0.990 for herring gull and 0.989 for lesser black-backed gull and great black-backed gull were recommended. Based on the evidence available at present, it was not possible to recommend an avoidance rate for use with the extended model for either black-legged kittiwake or northern gannet.

Transferability of avoidance rates between models (section 8)

9. Whilst the basic and extended Band models are the most widely used collision risk models at present, there are a number of alternatives. Based on our assessment of the alternative models which we were able to obtain descriptions of, the definitions and values we present in this report are likely to be broadly applicable to other models.

RECOMMENDATIONS AND LIMITATIONS

Definitions (Section 3)

- **Micro avoidance** should be defined as 'last-second' action taken to avoid collision, occurring within 10 m of the rotor blades.

- **Meso-response** should be defined as all behavioural responses, including attraction, in flight deflection and functional habitat loss, to the presence of a turbine occurring more than 10 m from the rotor blades and within the perimeter of the windfarm (500 m from the base of the outermost turbines).

- **Macro-response** should be defined as all behavioural responses, including attraction, displacement, and barrier effects, to the presence of a windfarm occurring beyond its perimeter (> 500 m from the base of the outermost turbines).

- Where an avoidance rate has been derived by comparing observed collisions to those expected in the absence of avoidance, this should be referred to as **within-windfarm avoidance**, it is a combination of meso-responses and micro-avoidance.

Recommended avoidance rates

- A **macro-avoidance rate of 0.64** is recommended for **northern gannet** (section 5.4). However, no data were available to derive a within-windfarm avoidance rate for this species (section 5.3). Based on the evidence available, there is no reason to suppose that the total avoidance rates for **northern gannet** should be less than those for all gulls. A **total avoidance rate** of **0.989** is thus recommended for use with the basic Band (2012) collision risk model. This would reflect a within windfarm avoidance rate of 0.970. We acknowledge that this is precautionary, but in the absence of more species-specific data, we feel it is appropriate. It was not possible to recommend an avoidance rate for use with the extended Band (2012) collision risk model based on the evidence available at present.

- No consistent evidence of macro-avoidance was found for **black-legged kittiwake** (section 5.4). As it was not possible to derive species-specific within-windfarm avoidance rates for **black-legged kittiwake**, the within-windfarm rates derived for the small gulls group were considered appropriate for use for this species (section 5.3). A **total avoidance rate** of **0.992** is thus recommended for the basic Band model. It was not possible to recommend an avoidance rate for use with the extended Band (2012) collision risk model based on the evidence available at present.

- No consistent evidence of macro-avoidance was found for **lesser black-backed gull** (section 5.4). Whilst it was possible to derive species-specific within-windfarm avoidance rates for lesser black-backed gull, these were based on limited data and thus the within-windfarm avoidance rates for large gulls were

considered more appropriate for use for this species (section 5.3). A **total avoidance rate** of **0.995** is thus recommended for use with the basic Band model and a **total avoidance rate** of **0.989** for use with the extended Band model (section 7).

- No consistent evidence of macro-avoidance was found for **herring gull** (section 5.4) and thus total avoidance rates reflect species-specific within-windfarm avoidance rates. A species-specific **total avoidance rate** of **0.995** is thus recommended for use with the basic Band model and a **total avoidance rate** of **0.990** for use with the extended Band model (section 7).

- No consistent evidence of macro-avoidance was found for **great black-backed gull** (section 5.4). As it was not possible to derive species-specific within-windfarm avoidance rates for **great black-backed gull**, the within-windfarm rates derived for the large gulls group were considered appropriate for use for this species (section 5.3). A **total avoidance rate** of **0.995** is thus recommended for the basic Band model and a **total avoidance rate** of **0.989** for use with the extended Band model (section 7).

- Given the multiple ways in which data can be interpreted, it is vital that future studies in which avoidance rates are derived are completely transparent and present their workings as a step-by-step process. Appendix 7 enables the reader to go back to the original source material and fully understand how the values presented in this report have been derived. This also offers an indication of the uncertainty present in the derived values.

- Based on the available data, it was not possible to derive species-specific avoidance rates for three of the five priority species. Of particular concern is the lack of within-windfarm avoidance data for northern gannet given that it is taxonomically distinct from the other four species, all of which are gulls. Future projects should focus on collecting data for northern gannet as a priority. Given the limitations in the data we identified for macro-responses, especially for gulls, there is also a need to collect further data on barrier effects and displacement/attraction rates.

1. INTRODUCTION

The Scottish Government has a target for 100% of Scottish demand for electricity to be met from renewables by 2020 by creating a portfolio of both onshore and offshore technologies (Marine Scotland 2011). However, concern over the environmental impacts of these developments in the UK, and in particular the risk of birds colliding with wind turbines, has contributed to the delay and cancellation of some projects. In order to quantify the risk of birds colliding with wind turbines, a number of collision risk models have been developed (Band 2012, Smales *et al*. 2013). These include an update to the Scottish Natural Heritage (SNH) collision risk model, originally developed for onshore windfarms (Band 2000, Band *et al.* 2007), redeveloped to better reflect data collected in relation to impact assessments for offshore windfarms (Band 2012). This work was undertaken as part of one of the projects undertaken through the Strategic Ornithological Support Service (SOSS) programme, a joint initiative involving industry, statutory nature conservation bodies (SNCBs) and the RSPB. These models combine a series of parameters describing the turbine design and operation with estimates of a bird's size and behaviour in order to predict the number of birds that would be expected to collide with a turbine over a given time period. Of these parameters, detailed analysis has suggested that these models are highly sensitive to variation in the avoidance rate, the proportion of birds which take action to avoid colliding with a turbine (Chamberlain *et al.* 2005, 2006). Despite this, there has been relatively little empirical evidence put forward to support avoidance rates for offshore windfarms, which are likely to vary according to species and weather conditions, in particular visibility.

Whilst avoidance rates can be determined from observed mortality rates or actual observations of birds' behaviour, defining robust values for use in collision risk modelling can be extremely challenging. However, there are concerns that avoidance rates derived from observed mortality rates may act as a 'fudge-factor', incorporating observer biases and model error, as opposed to the actual behaviour of the birds (May *et al.* 2010, Douglas *et al.* 2012). Current guidance from SNH (2010) is that, in the absence of species-specific empirical data, a default avoidance rate of 0.98 should be used for most species in onshore windfarm assessments and this value has been widely used in the offshore environment as well. However, in light of recent evidence (e.g. Everaert & Stienen 2007, Krijgsveld *et al.* 2011) the validity of this approach has been questioned and concerns have been raised by developers that it will lead to an over-estimate of the likely number of collisions (Moray Offshore Renewables Limited 2012, Trinder 2012, Smartwind/Forewind 2013) and, as a consequence, potentially contribute to the delay and cancellation of key projects. In a policy environment where there is limited evidence on which to base decisions it is important to reflect uncertainty, but not to apply unrealistic levels of precaution which will make it difficult to reach informed decisions about where and where not to build windfarms.

There is a strong need for a consensus on the appropriateness of recommended avoidance rate values given the influence they have on collision estimates and, therefore, consenting decisions. However, at present, there is a lack of clarity over the interpretation of studies of avoidance behaviour and the applicability of the resultant avoidance rates to different collision risk models, study sites and species. As a result, details presented in reviews of avoidance behaviour of birds in the

marine environment (e.g. Maclean *et al.* 2009, Cook *et al.* 2012) have been subject to confusion. A key reason for this is the lack of consistency in the terminology applied to different spatial scales of avoidance, and the widely varying interpretation of the types of avoidance behaviour occurring. There is therefore, an urgent need for a review of avoidance behaviour in offshore windfarms in order to provide a clear appraisal of the existing evidence base, provide a robust critique of the data available with which to refine recommendations on avoidance rates and offer clear guidance as to how they should be used in future collision risk modelling scenarios. Whilst the focus of this review will be on collision risk modelling and species relevant to the UK context, it will draw on evidence from Europe and beyond.

This work aims to reduce the current level of uncertainty around appropriate avoidance rates for seabird species within collision risk modelling by providing a thorough review of the existing evidence base. The scope of this review is broader than those previously undertaken (e.g. Cook *et al.* 2012) and includes quantitative and qualitative analyses of the data identified with a view to identifying representative avoidance rates for five priority species – northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull and great black-backed gull. The review identifies current knowledge gaps and aims to ensure that future strategic work is targeted at addressing the most appropriate issues. Due to the sensitivity of the work and the importance of its conclusions, the work has been overseen by a steering group of key stakeholders and experts, with a view to gaining widespread acceptance of its conclusions.

2. OBJECTIVES

2.1 Produce definitions for the types and scales of avoidance rates that will be used throughout the review document

It is important to make a distinction between avoidance rates, as used in collision risk models, and avoidance behaviour. Avoidance behaviour refers specifically to the behavioural response of birds to wind turbines. However, at present, in addition to accounting for avoidance behaviour, avoidance rates are often used as a 'fudge-factor' to account for error in the model itself and in its input parameters (see May *et al.* 2010, Douglas *et al.* 2012). Whilst SOSS guidance (Band 2012) sets out how these uncertainties should be accounted for in the collision risk modelling process, in practice, this is rarely done. The purpose of this review is to identify suitable avoidance rates for use in collision risk models; these rates will be informed, where appropriate, by recorded estimates of avoidance behaviour.

A lack of clear, working definitions for different avoidance rates has hampered attempts to come up with standardised measures. Present definitions of avoidance rates rely on an ability to collect empirical data with which to compare predicted and observed collision rates (SNH 2010). As this is impractical for the offshore environment, Band (2012) proposes combining estimates of micro- (or near-field) avoidance, where a bird takes action to avoid collision at a point close to the turbine, and macro- (or far-field) avoidance, where a bird takes action to avoid collision at a point distant from the turbine, to generate an estimate of total avoidance. However, the empirical data underpinning such definitions is currently inconsistent and difficult to interpret.

A key problem is often the lack of detail over what spatial scale data have been collected at. For example, radar monitoring has shown that birds may take action to avoid entering a windfarm at distances of up to 6 km (Christensen *et al.* 2004), far further than could be observed by eye. As a result, by relying on visual observations, avoidance rates may be under-estimated as a significant proportion of birds will have taken action to avoid the windfarm before they are visible. Similarly, at present, it is not possible to identify birds to species level on the basis of radar echoes; consequently, by relying on radar, it will not be possible to derive species-specific avoidance rates. This is further complicated by evidence that avoidance can occur in a three-dimensional space, with horizontal avoidance, where a bird alters its heading to avoid collision, and vertical avoidance, where a bird alters its altitude to avoid collision (Krijgsveld *et al.* 2011, Plonczkier & Simms 2012). Such alterations may be relatively subtle and difficult to detect by eye. Where radar is utilised to monitor movements in response to turbines, it requires the use of both horizontal and vertical radar. Evidence describing three-dimensional avoidance behaviour, if it exists, is likely to be extremely limited. In defining different avoidance behaviours, the review therefore gives careful consideration to the methodologies used to collect the necessary data.

Wind turbines are most typically in the order of seven rotor diameters apart (Meyers & Meneveau 2012), based on existing turbine designs, this may vary from 480 m to 1.5 km, depending on the capacity used. Given the variable distances between turbines and the difficulties in obtaining consistent estimates of avoidance behaviour

over the relevant spatial scales, the review considers whether it is possible to define micro-and macro-avoidance with reference to distance to turbines, or whether a more pragmatic approach, basing definitions on whether a bird is inside or outside a windfarm would be more appropriate. The review considers whether these definitions are appropriate to all species and groups, or whether a more flexible approach is necessary. This may depend on what evidence is available for different species. For example, avoidance rates for terns have often been derived from observed collision rates (Everaert 2008), whilst for other species, such as northern gannets, avoidance rates may be more reliant on radar data (Krijgsveld *et al.* 2011). The review then considers evidence for avoidance behaviour occurring over horizontal and vertical planes.

The review provides clear and concise definitions for micro-horizontal avoidance, micro-vertical avoidance, macro-horizontal avoidance and macro-vertical avoidance. Definitions are produced based on the behaviour of the birds as opposed to the requirements of a model and offer guidance about how final values can be adapted for use in different models.

Defining the different forms of avoidance behaviour represents a major step forward in collision risk modelling. These definitions are central to the rest of the project, and, as such, have been agreed through discussion with the project steering group of key stakeholders and experts.

2.2 Review the current use of avoidance rates

In order to provide context to this work, it is important to consider how avoidance rates are currently used. With this in mind, the review considers published EIAs and identifies what avoidance rates have been used within the collision risk modelling process and what justifications have been put forward for their selection. This will help us determine how consistently existing guidance has been interpreted and applied, and help refine future guidance in order to minimise discrepancies in its application.

2.3 Review and critique existing avoidance behaviour studies and any derived rates

Avoidance rates have been derived from both observed mortality rates and actual observations of birds' behaviour (Cook *et al.* 2012, Trinder 2012, Moray Offshore Renewables Limited 2012, Smartwind/Forewind 2013, Everaert 2014). In Belgium, at Zeebrugge port breakwater, onshore collision rates in terns and gulls have been used to derive avoidance rates based on recorded movement patterns and assumptions about turbine design (Everaert & Stienen 2007 Moray Offshore Renewables Ltd. 2012, Everaert 2014). However, the difficulties in directly recording collisions in the marine environment mean that studies of avoidance at offshore windfarms have relied on observing behaviour (Desholm *et al.* 2006, Blew *et al.* 2008, Krijgsveld *et al.* 2011). These studies have varied both in the species they have investigated, and also in the potential form of avoidance behaviour reported.

Recognising that appropriate data may be extremely limited, we initially take a broad approach to our review, reviewing evidence for avoidance behaviour in marine birds

generally. We demonstrate how this evidence relates to the definitions set out in the previous section of the report. Having done this, we assess whether sufficient evidence exists to draw conclusions about avoidance behaviour in five priority species – northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull and great black-backed gull. If this is not possible, we will consider how to combine evidence within groups of species, on the basis of the ecology of the species concerned. Where this is necessary, we clearly state which species are in each group.

In order to make an assessment of the level of confidence in the reported avoidance rates for each species or species group, we make a detailed qualitative critique of each study. Key questions include:

i. How have avoidance rates been derived?

We consider first whether the avoidance rates reported have been determined from observed mortality rates or actual observations of birds' behaviour. The data collection methods used are summarised, and the limitations of each method discussed. Where avoidance rates have been back-calculated from observed collisions at reference windfarms, they may incorporate error associated with model input parameters including population estimates, flight heights and turbine operational characteristics in addition to the actual avoidance behaviour of the birds. In contrast, direct observations of birds' behaviour in relation to turbines will not incorporate model error. However, these observations may still need careful interpretation given methodological constraints over how data may be collected, for example, the distances over which birds can be observed in comparison to the distances over which they may take avoidance action.

ii. How comparable are the different datasets?

Avoidance rates based on behaviour have typically been derived from a series of visual or radar observations (Desholm & Kahlert 2005, Blew *et al.* 2008), or through a combination of both (Krijgsveld *et al.* 2011, Plonczkier & Simms 2012). The range of distances over which data can be collected varies markedly between these platforms (Cook *et al.* 2012) and it is important to consider whether estimates – particularly of macro-avoidance – are comparable between different studies.

It is also important to consider how and when data have been collected. For example, visual observations from land, or an offshore platform, may differ from those obtained during a boat-based survey, where the movement of the boat may mean that surveyors have a less stable platform or because birds may exhibit a behavioural response to the presence of a boat (although following standard guidance should help to minimise the influence of these factors: Camphuysen *et al.* 2004). Visibility may also strongly influence results from visual observations. Seasonality may influence the results from both radar and visual observations as foraging birds may respond very differently to migrating birds (Blew *et al.* 2008, Krijgsveld *et al.* 2011). This may be particularly important for radar studies, where it is not possible to identify radar echoes to species level and, as a result, it is more difficult to separate observations of migrants from those of local, foraging birds during periods of passage.

iii. Are reported avoidance rates affected by any special factors?

The location of the windfarm may have a strong impact on reported collision rates. If these collision rates are then used to calculate avoidance rates, it may lead to an erroneous assessment of avoidance behaviour. For example, a Belgian study has reported collision rates at a windfarm in Zeebrugge for terns (Everaert & Stienen 2007). The results from this study have been widely used to calculate micro-avoidance rates for terns (e.g. Whitfield 2008). However, as this windfarm was located on a seawall, next to a breeding tern colony, it is unclear whether behaviour around the turbines would be consistent with that of foraging terns, further out to sea. In addition, the size of turbines planned for offshore windfarms is significantly greater than those installed at many of the sites for which collision data are available. For this reason, we will consider whether there is any evidence for a relationship between turbine size and the avoidance rates derived from mortality data.

2.4 Provide summary avoidance rates and a total avoidance rate for each priority species/species group based on the evidence available at present

Based on the information compiled from the above review, we derive avoidance rates based on published evidence for each of the five priority species – northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull and great black-backed gull, and other species as relevant. Where necessary, this involved going back to the source material of the studies concerned and back-calculating avoidance rates following the methodology set out by Band (2000). Where insufficient data were available to make recommendations for individual species, we combine estimates within species groups, based on the ecologies of the species concerned. Based on our critique of the studies from our review we then indicate where our confidence in each reported value is affected by the quality of the data it is based on.

Where possible, we combine avoidance rates collected at different scales, in order to calculate a total avoidance rate for each species. Estimates of micro-avoidance and macro-response can be combined to give an overall avoidance rate following equation 1, if sufficient data are available, we will extend this equation to include horizontal and vertical avoidance, as detailed in equations 2 and 3. Given the limited evidence available, it may be necessary to draw in data from closely related species and derive avoidance rates based on a group, rather than species-specific basis. Where this is necessary, we will clearly state what we have done and indicate our confidence in the derived rate accordingly.

$$A_{rate} = 1 - [(1 - A_{micro}) \times (1 - A_{macro})] \quad [Eq. 1]$$
$$A_{micro-} = 1 - [(1 - Mi_{horiz}) \times (1 - Mi_{vert})] \quad [Eq. 2]$$
$$A_{macro-} = 1 - [(1 - Ma_{horiz}) \times (1 - Ma_{vert})] \quad [Eq. 3]$$

Where A_{rate} is the total avoidance rate, A_{micro-} is the micro-avoidance rate, A_{macro-} is the macro-avoidance rate, Mi_{horiz} is the micro-horizontal avoidance rate, Mi_{vert} is the micro-vertical avoidance rate, Ma_{horiz} is the macro-horizontal avoidance rate and Ma_{vert} is the macro-vertical avoidance rate. Note that the ability to combine horizontal

and vertical movements in this way will depend on how data have been collected. It is likely that some birds will make horizontal and vertical movements concurrently, and therefore, it would not be appropriate to combine data in this way.

This summary is used as the basis for a gap analysis based on our earlier definitions of avoidance behaviour. In combination with the above critique of avoidance rate studies, this gap analysis will help provide a target and possible methodologies for future research on avoidance behaviour of birds in relation to offshore windfarms, for example the Offshore Renewables Joint Industry Project (ORJIP), due to get underway in summer 2014 (Davies *et al.* 2013).

2.5 Undertake an assessment of the sensitivity of the conclusions reached to inputs and conditions under which they were collected

The final avoidance rates are likely to be sensitive to both factors which are directly parameterised within the collision risk model, such as species' flight heights, turbines' operational time and rotation speed, those parameterised in collecting collision data such as corpse collection, and also those which are not directly parameterised, such as seasonality, weather conditions and whether data have been collected during the day or night. Whether estimates of avoidance behaviour have been derived from behavioural observations or recorded collision rates, they are likely to be influenced by the factors which are not directly parameterised. For this reason, we assess how such variables are likely to have influenced the final avoidance rate in each study. For example, avoidance rates based on data only collected during conditions with better than average visibility may be expected to differ from those based on data collected during periods of poor visibility, a potential source of model error. Where avoidance rates have been derived from collision data, there is the also potential for the model input parameters to influence the final values.

These methodologies have typically been restricted to turbines at onshore locations (Everaert & Stienen 2007), where corpse collection is practical. There are concerns that this may lead to an over-estimate of the avoidance rate as some corpses go undetected and correction factors to account for this (Winkelmann 1992, Bernardino *et al.* 2013) may not be correctly applied. With this in mind, we focus on the best quality studies, but also consider how undetected corpses may influence the avoidance rate we derive.

Where a collision rate is available for a site, the avoidance rate (A_{rate}) can be calculated as follows:

C_{pred} = (Flux rate * P_{coll}) + error [eq. 4]
$A_{rate} = 1 - (C_{obs}/C_{pred})$ [eq. 5]

Where C_{pred} is the predicted number of collisions in the absence of avoidance action, C_{obs} is the observed number of collisions, flux rate is the total number of birds passing through the rotor swept area and P_{coll} is the probability of a bird colliding with a turbine. The probability of collision, P_{coll} can be calculated following the formula set out in Band (2012). However, this highlights a second area where the conclusions about avoidance rates may be sensitive to the inputs as values of P_{coll} will be specific

to the design of turbines (Cook *et al.* 2011). Consequently, knowledge of rotor speed, radius, chord width and pitch, for the turbine concerned, are required before estimating an avoidance rate from a collision rate. These characteristics can vary considerably, even between turbines of a similar generating capacity (http://www.4coffshore.com). As a result, error is likely to be introduced into the calculation through inaccuracies in estimates of the flux rate and also through inaccuracies in the estimation of P_{coll}.

As detailed in Cook *et al.* (2012), failing to account for turbine design correctly when deriving avoidance rates as described above can lead to erroneous estimates of P_{coll} and, therefore, the avoidance rate. For this reason, where a study reports a collision rate, rather than an avoidance rate, we have attempted to obtain data on these parameters. Where we are unable to obtain this information, we calculate a value of P_{coll} based on the parameters from a range of turbines of a similar size. We then consider whether avoidance rates derived from collision estimates are more sensitive to variation in turbine design or to correction factors that account for failure to detect corpses.

2.6 Applicability of avoidance rates to different collision risk models

We finally consider how the total avoidance rate, and its constituent elements, reflect the values necessary for collision risk modelling. At present, the collision risk model formulated by Band (2012) for use in the offshore environment has three different options which can be used to estimate the total number of birds at risk of collision. These options reflect different ways in which estimates of the proportion of birds at collision risk height can be incorporated into the collision risk modelling process. Band model option 1 assumes that birds are distributed evenly within the rotor-swept area of a turbine. It bases estimates of the proportion of birds at risk of collision on data collected during pre-construction surveys of the site in question. Band model option 2 is mathematically identical to the first option, also assuming an even distribution of birds within the rotor-swept area of the turbine. However, the proportion of birds at collision risk height is estimated from continuous distributions derived from data collected across multiple sites (Cook *et al.* 2012, Johnston *et al.* 2014a,b). Options 1 and 2 of the Band model are collectively referred to as the basic model. In practice, birds are unlikely to be evenly distributed across the rotor-swept area of a turbine (Johnston *et al.* 2014a). Band model option 3, often referred to as the extended Band model, accounts for this by using a continuous flight height distribution to estimate collision risk at different points within the turbines rotor-swept area.

As birds are typically clustered to the lower edges of the rotor-swept area (Johnston *et al.* 2014a), option 3 often results in lower estimates of collision rates. As a consequence, there is intense interest in its use within EIAs for offshore windfarms. However, avoidance rates currently in use that are derived for the onshore environment by combining collision rates with estimates of P_{coll} from the basic Band model are not suitable for use in the extended model, as accounting for a heterogeneous flight height distribution will result in a lower number of collisions predicted in the absence of avoidance. (Although, note that this difference may be partially offset as avoidance rates derived in this way do not account for changes in flight altitude in response to the presence of a windfarm.) As a result estimates of

avoidance behaviour based on the basic model are likely to be higher than is appropriate for the extended model (equations 4 and 5) – this is considered as part of the review.

Where estimates of avoidance rates have been derived from behavioural observations, for example displacement from offshore windfarms, rather than recorded collision rates, the applicability to different models is less clear. We consider how our final avoidance rates have been derived and what implications this has for how they are incorporated in collision risk models.

We also offer guidance not just on the applicability of avoidance rates to the basic and extended Band models, but also their transferability of avoidance rates to alternatives including the Biosis model (Smales *et al.* 2013).

The data necessary to derive avoidance rates suitable for use with option 3 of the Band model following the formula given by equation 6 are often unavailable. However, a suitable avoidance rate can be derived by estimating the ratio of P_{coll} from option 2 of the Band model to P_{coll} from option 3 of the Band model and applying this to the inverse of the avoidance rate used for option 1. For the rationale and a full description of this approach see the supplement to the guidance on 'Using a collision risk model to assess bird collision risks for offshore windfarms' (Band 2012) provided by Bill Band as Annex 1 to this report.

3. DEFINITIONS OF AVOIDANCE BEHAVIOUR

3.1 Introduction

Chamberlain *et al.* (2005, 2006) demonstrated that, of the parameters used in the Band collision model (Band 2006), the avoidance rate used was among those that the predicted collision rates were most sensitive to. Subsequently, the identification of appropriate avoidance rates has been subject to widespread debate. Guidance produced by Scottish Natural Heritage (SNH 2010) has been largely accepted in the UK for the terrestrial environment, subject to revision as additional data become available (e.g. Pendlebury 2006). Whilst this document references some seabird species, its guidance for offshore windfarms is limited to the suggestion that a range of avoidance rates should be presented. Country agencies have provided advice to developers as necessary, but the lack of guidance produced specifically for the offshore environment, and for the updated Band model for use in the offshore environment (Band 2012), has led to uncertainty amongst developers, regulators and other stakeholders as to what values reflect realistic avoidance rates (e.g. MacArthur Green 2012, MORL 2012) and for which collision risk models they are appropriate. Previous studies have attempted to review avoidance behaviour in offshore species (e.g. Maclean *et al.* 2009, Cook *et al.* 2012) but a failure to gain widespread consensus about the values presented has meant the situation remains largely unresolved.

Deriving avoidance rates for terrestrial windfarm developments has been based largely on the ability to estimate the numbers of birds killed by collisions. Every bird flying through the rotor-swept area of a turbine has a probability of colliding with the turbine blades (P_{coll}), typically in the range of 5-10% for seabirds, depending on species and the design of the turbine concerned (Cook *et al.* 2011). By multiplying the total number of birds expected to pass through the rotor-swept area of a turbine by P_{coll} it is possible to predict the number of collisions that would be expected, should birds take no action to avoid collision. In the case of terrestrial windfarms estimates of the total number of collisions actually occurring, once turbines are operational, can be made by using corpse searches around the windfarm to assess actual mortality rates, or observed collision rates[1]. Band (2000) therefore suggests that the avoidance rate can be thought of as equation 6, where the collision rate expected in the absence of avoidance is the total number of birds (Flux rate) passing through the rotor-swept area of a turbine, multiplied by P_{coll}. However, in practice both P_{coll} and the flux rate are likely to be subject to error – P_{coll} in relation to the model input parameters and flux rate in relation to estimates of the total number of birds passing through the windfarm. Of the two, the error associated with the flux rate is likely to be greatest as a result of the difficulty in recording the number of birds passing through a site over an extended period of time and the need to extrapolate from, often brief, observation periods to estimate a flux rate for the study period as a whole. As a result of the need to incorporate this error, it may be better to think of this in terms of an avoidance correction factor, as opposed to an avoidance rate, which implies it is solely dependent on the behavioural responses of birds:

[1] Subject to some carcass recovery factor (i.e. the potential to miss carcasses, removal by predators, etc.).

$$Avoidance = 1 - \frac{Observed\ collision\ rate}{Pcoll \times Flux\ Rate} \quad \text{(eq. 6)}$$

However, in the case of offshore windfarms, recording actual collisions, or mortality rates, is not currently practical, although the forthcoming Offshore Renewables Joint Industry Project (ORJIP) will aim to provide additional data to inform avoidance rates using behavioural observations (Davies *et al.* 2013). Therefore, at present, guidance on appropriate avoidance rates for use in the offshore environment draws on the experiences gained in the terrestrial environment, as well as being informed by studies of bird movements, where suitable data are available (e.g. Desholm & Kahlert 2005, Petersen *et al.* 2006, Masden *et al.* 2009, Blew *et al.* 2008, Krijgsveld *et al.* 2011). Where studies have sought to use movement data to inform values for avoidance rates, this has often led to confusion due to uncertainty over the spatial scales involved. Birds have been shown to alter their flight paths in order to avoid entering an offshore windfarm at distances of up to 6 km (Christensen *et al.* 2004). As a result, where avoidance rates have been derived from human observations they may represent a substantial under-estimate of total avoidance, as many birds will have taken action to avoid the windfarm before they become visible to observers. The difficulties caused in attempting to draw firm conclusions from such disparate data sources has led to a variety of terms being used to sub-divide avoidance behaviour at different spatial scales.

At a simple level, Cook *et al.* (2012) and Band (2012) suggest that the total avoidance rate for an offshore windfarm could be considered as (eq. 7):

$$Total\ Avoidance = 1 - (\ 1 - Macro\ \times\ 1 - Micro\ ) \text{ (eq. 7)}$$

We use this definition as the basis for discussion relating to the different types of avoidance that need to be quantified in order to derive an estimate of total avoidance, and extend it to incorporate meso-avoidance (eq. 8), as defined below.

$$Total\ Avoidance = 1 - (\ 1 - Macro\ \times\ 1 - Meso\ \times\ 1 - Micro\ ) \text{ (eq. 8)}$$

3.2 Defining appropriate spatial scales of avoidance

This section aims to define appropriate spatial scales of avoidance; for detailed review of the evidence for avoidance at these defined scales, see section 5.

A bird may respond to a fixed object, such as a turbine, at any point between the time at which it first observes the object and the time at which it passes or collides with the object, or based on previous experiences of the site. As such, attempts to subdivide avoidance behaviour with reference to spatial scale are largely arbitrary and the different behaviours should be seen as part of a continuum. Nevertheless, such divisions are necessary given the spatial scales over which these behaviours can be recorded. Band (2012) focusses on macro- and micro- avoidance, with a third category, meso-avoidance, fitting in the gap between the two also suggested (Pendlebury, *pers. comm.*). We consider these scales in turn, with each reflecting an increasing distance between the bird and the turbine blades (Figure 3.1). However,

the distances over which these categories of behaviour occur are more difficult to define.

Figure 3.1 Spatial scales over which avian responses to turbines have been recorded

It is also necessary to consider how avoidance rates are applied within the collision risk modelling framework. Expected collision rates (as per eq. 7) are typically derived using estimates of the numbers of birds flying through the windfarm area prior to construction. Therefore, overall avoidance rates need to account for birds no longer entering the windfarm area post-construction (i.e. birds exhibiting displacement and barrier effects) in addition to avoidance of the turbines themselves. As a result, it is necessary to consider how other effects, such as displacement and barrier effects, may contribute to the overall avoidance rates, as part of macro-avoidance.

We consider how each of these scales may be used to inform collision risk modelling below:

Macro- Band (2012) gives the example of displacement as one impact which may contribute to macro-avoidance. Displacement is typically assessed by comparing numbers of birds in the area of the windfarm to those recorded in a baseline period. However, difficulties in quantifying displacement rates – numbers may vary for many reasons in addition to the development of the windfarm, and it is important that this is considered in an appropriate survey design, for example using a BACI-approach (Masden *et al.* 2010) – mean that interpreting these data must be undertaken with caution and careful consideration of the survey design (Maclean *et al.* 2013). Furthermore, published displacement rates can refer to the numbers of birds displaced from the windfarm plus a significant (species-dependent) buffer distance around the windfarm. Consideration must also be given as to whether displacement rates reflect all birds within the windfarm area and buffer, or just those on the water. As collision risk modelling relates only to birds in flight, if displacement rates refer only to birds on the water, they may not reflect macro-avoidance. Relying solely on displacement, as often reported in Environmental Impact Assessments, may therefore underestimate the true scale of macro-avoidance because 1) estimates may not account for birds in flight; and 2) estimates do not account for birds that are displaced from the windfarm area, but remain within the buffer surrounding the windfarm.

In addition to measuring displacement rates, a number of offshore windfarm post construction monitoring studies have used radar to assess the proportion of birds which enter a windfarm area (e.g. Petterson 2005, Petersen 2006, Krijgsveld *et al.*

2011). The potential for windfarms to act as a barrier to birds in this way has been widely discussed, mostly in the context of migrants (e.g. Desholm & Kahlert 2005, Masden *et al.* 2009), although it may also be of relevance to seabirds commuting between breeding colonies and feeding areas – an area of study that needs addressing with some urgency. Such studies would illustrate changes in flight trajectory amongst birds approaching windfarms and would help to determine the spatial scale over which such responses may occur.

In addition to displacement and the windfarm acting as a barrier, several studies have suggested that some species, notably gulls and cormorants, may be attracted to the area of offshore windfarms (e.g. Lindeboom *et al.* 2011, Leopold *et al.* 2011). The macro-avoidance rate needs to capture the change in bird numbers within the windfarm area resulting from the development of the windfarm site. Consequently, the term 'macro-avoidance', may lead to confusion as, conceptually, the idea of a negative macro-avoidance rate (i.e. birds being attracted to a windfarm) may be difficult to communicate to stakeholders. For this reason, use of the more neutral term, **macro-response**, may be preferable as it implicitly covers both attraction and avoidance (Figure 3.2).

Figure 3.2 Range of proportional responses to the presence of an offshore windfarm as they would be incorporated in eq. 2 (above), i.e. a response of -0.1 would reflect an increase in the number of birds present within the windfarm of 10% in comparison to baseline numbers, whilst a response of 0.1 would reflect a decrease of 10% in comparison to baseline numbers, which are sensitive to survey design due to the extent of year on year variation in seabird abundance.

The macro-response of birds to the presence of a windfarm should be defined as the behavioural response taking place outside the windfarm perimeter. It is important that the perimeter of the windfarm is clearly defined. Definitions could be based on characteristics such as turbine rotor diameter, or the inter-array turbine spacing. However, such definitions would vary between sites in relation to the layout and size of turbines used, meaning values for the macro-response rate would be less directly comparable between sites. For this reason, defining the perimeter as extending a fixed distance from the base of the outermost turbines is preferable. The review will define of the perimeter as the boundary of a minimum convex polygon encompassing an area extending from a distance of **500 m from the base of the outermost turbines** (see Figure 3.3).

The term macro-response will be used to refer to changes in bird numbers within the windfarm area resulting from the development of the windfarm site, through processes including, but not limited to, attraction, displacement and barrier effects. Where displacement is used to infer a macro-response rate, it is important to be clear whether this reflects displacement from the windfarm only, or displacement from the windfarm plus a buffer. Buffers considered in the assessment of displacement effects typically extend beyond the 500 m around the windfarm

perimeter considered here as some birds may respond to the presence of the windfarm at distances greater than this. Measures of displacement that use such buffers may thus underestimate the macro-response rate considered here. As collision risk models refer to birds in flight only, when using displacement rates to estimate a part of macro-avoidance behaviour, it is also important to lend more weight to studies that distinguish the displacement rates of birds in flight and on the water, or those for which it is possible to estimate the number, or proportion, of birds in flight.

Micro- Blew *et al.* (2008) suggests that micro-avoidance reflects a 'last-second' alteration to a flight path in order to avoid collision with a turbine. Petterson (2005) and Blew *et al.* (2008) both suggest that birds adjust their flight paths to avoid entering the rotor-swept zone of a turbine and that, therefore, birds may only rarely need to take last second action to avoid collision, possibly as a result of adverse conditions, such as poor visibility. This is borne out by empirical evidence presented in Desholm (2005) and Krijgsveld *et al.* (2011) (see section 5.3).

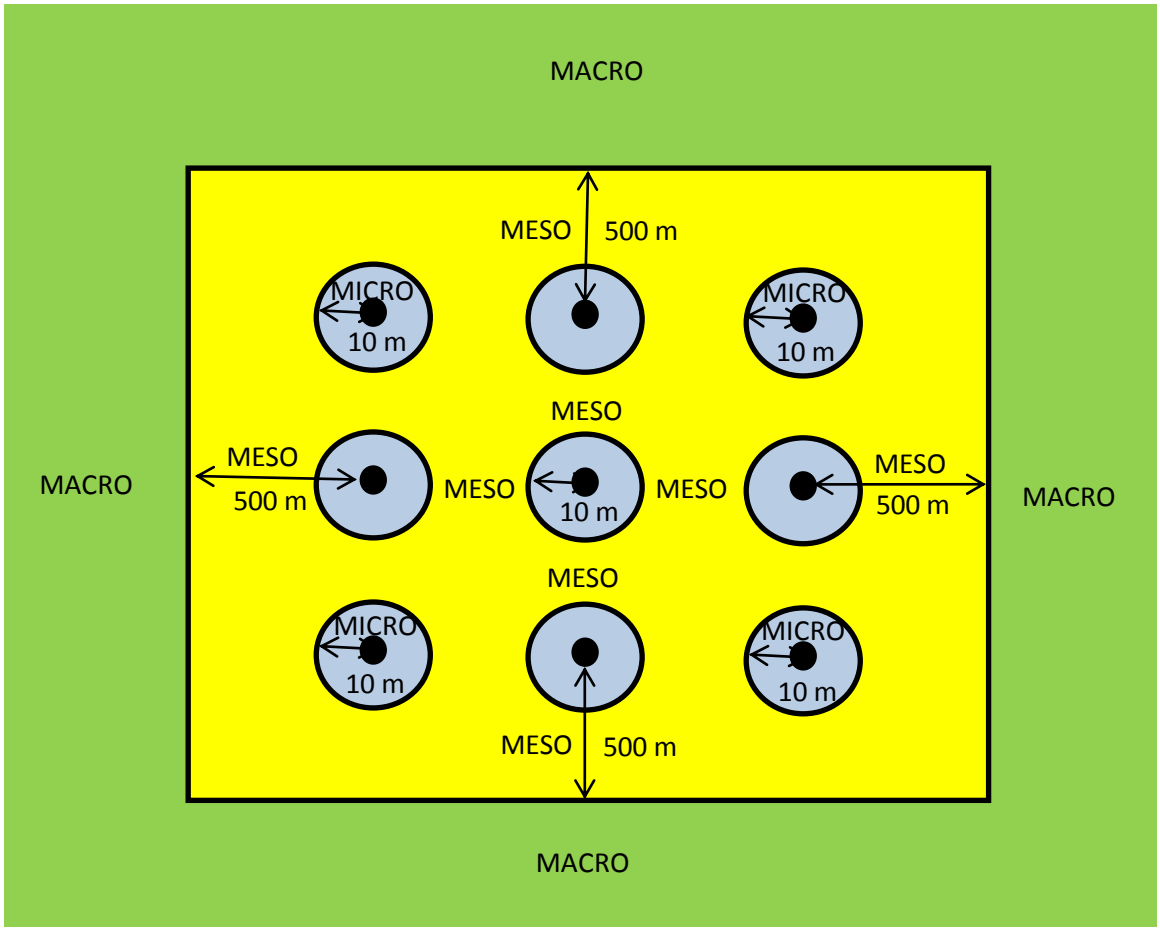


Figure 3.3 Schematic illustrating the spatial scales over which micro-avoidance, meso- and macro- responses operate. Dots refer to turbine tower locations (not to scale).

Therefore, it would seem reasonable to define **micro-avoidance** as a last-second alteration to a bird's flight path in order to avoid collision. For the purposes of

observational studies, such last-second avoidance would be expected to occur in a 3-dimensional space **within 10 m of the turbine blades** (i.e. at distances of 10 m horizontally or vertically from edges of the turbine blades) – though note that this distance (and consequently the appropriate definition of micro-avoidance) may be refined based on future advances in the techniques used to collect the necessary data (see Figure 3.3). Such behaviour is likely to be recorded relatively rarely.

Meso- Whilst macro-responses reflect behaviour outside the windfarm and micro-avoidance reflects last-second action taken to avoid collision, there is a need to consider a third category, reflecting species responses to turbines within a windfarm (Figure 3.4). Both Desholm & Kahlert (2005) and Krijgsveld *et al.* (2011) demonstrated that the majority of birds do not pass within 50 m of a turbine. However, some, such as cormorants, may be attracted to structures, which offer potential roosting sites (e.g. Leopold *et al.* 2011). For this reason, as in the case of macro-response, it may be more straightforward to talk about a **meso-response** to turbines than meso-avoidance. The term meso-response should be used to refer to all behavioural responses to the **turbines beyond the 10 m buffer around the rotor blades, covered by micro-avoidance, and within the perimeter of the windfarm** (see Figure 3.3). This may include, for example the attraction of cormorants to turbine bases as a roosting site, as the base of the turbine would be beyond the 10 m buffer around the rotor blades.

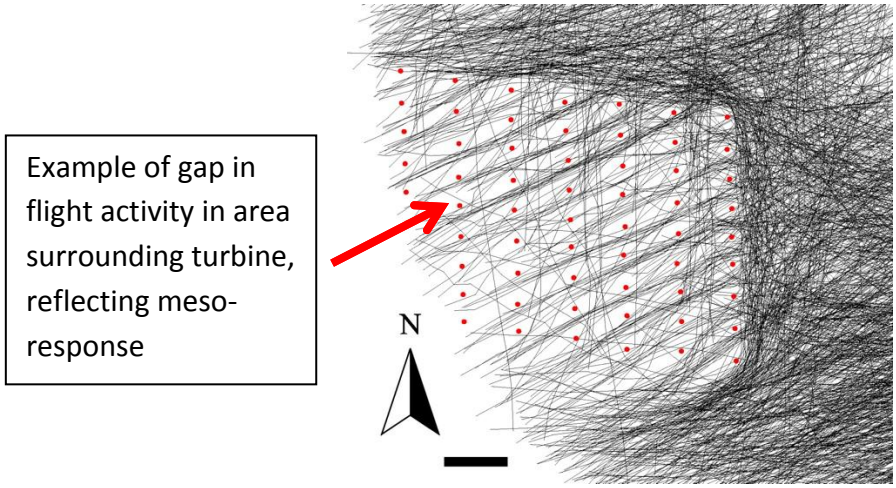


Figure 3.4 Flight trajectories of migrating waterbirds within an offshore windfarm, red dots indicate locations of turbines. Reproduced with permission from Desholm & Kahlert (2005) Avian collision risk at an offshore windfarm. *Biology Letters* 1: 296-298.

At present, the scale at which data are collected may make it difficult to differentiate between a meso-response and micro-avoidance. Therefore, it is recommended that the term **macro-response** is used to refer to a response outside the windfarm and within-windfarm response, covering both the meso- and micro-scale, is used to refer to a response occurring inside a windfarm. In response to technological advances, a fuller separation of meso-responses from micro-avoidance is likely to be possible in the near future. For example, it may be possible in future to combine radar monitoring of flight paths through offshore windfarms to capture meso-responses (as

in Desholm & Kahlert 2005) with images captured from turbine mounted cameras to capture micro-avoidance (as in Desholm *et al.* 2006).

3.3 Defining the appropriate 3-D level of avoidance

This section aims to define appropriate 3-D scales of avoidance; for detailed review of the evidence for horizontal and vertical meso-avoidance, see section 5.2.

In addition to occurring over a range of different spatial scales, avoidance behaviour may occur in both the horizontal and vertical planes. Below, we describe how observations of horizontal and vertical avoidance may be collected and the spatial scales which may be relevant to each. This distinction is important given that some methodologies for recording avoidance behaviour, such as radar, may not detect both horizontal and vertical movements, meaning that where only one is recorded, the derived avoidance rate is likely to be an underestimate, which may be offset by an inability to record horizontal and vertical movements occurring concurrently. There is also a need to consider the relationships between avoidance and other effects of offshore windfarms on birds, for example barrier effects and displacement.

Horizontal Avoidance Much of the research into the avoidance behaviour of seabirds in relation to offshore windfarms has focussed on horizontal avoidance, whereby birds alter their flight paths so that they fly around turbines or do not enter the perimeter of the windfarm (i.e. Desholm & Kahlert 2005, Masden *et al.* 2009). These data have been collected using a variety of methodologies, notably visual observations (i.e. Krijgsveld *et al.* 2011) and radar observations (i.e. Petersen *et al.* 2006). We consider that all 3 spatial scales defined here are relevant in the context of horizontal avoidance.

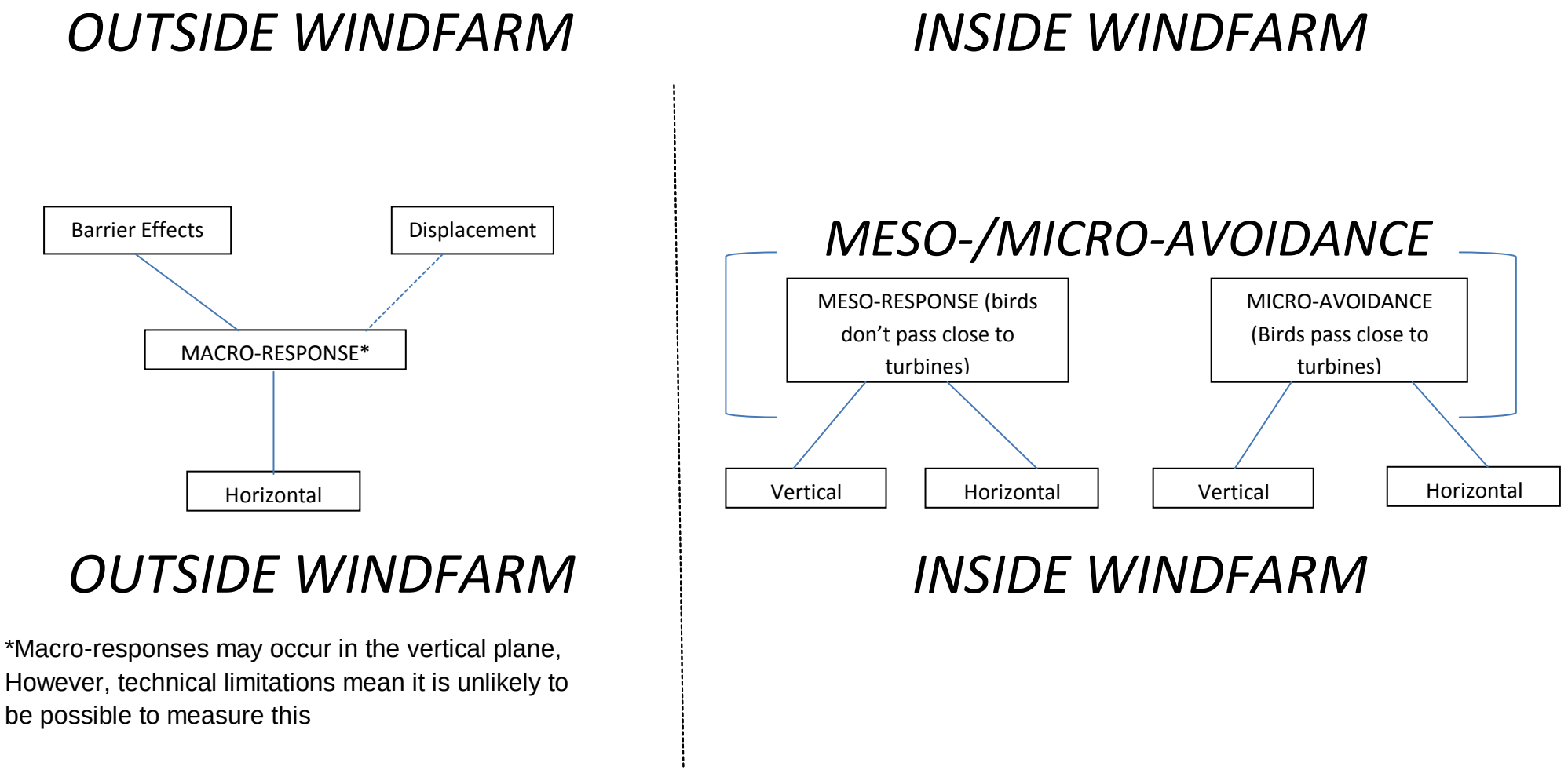
Vertical Avoidance As technologies and survey protocols for monitoring collisions become more developed (e.g. Desholm *et al.* 2006, Collier *et al.* 2011a, 2011b) monitoring of both horizontal and vertical movements around turbines should become more feasible. For radar, however, at greater distance this may be more challenging as detecting both horizontal and vertical avoidance requires the use of both x- and y-band radar. At present, radar monitoring of bird movements in and around offshore windfarms typically focuses on horizontal avoidance behaviour, using horizontal radar (e.g. Petersen *et al.* 2006). Where changes in flight height amongst birds entering the windfarm have been estimated (e.g. Blew *et al.* 2008) this has been at too coarse a resolution to inform vertical avoidance. However, recent developments in radar technology (e.g. http://www.robinradar.com/3d-flex/) may make this a more practical solution to investigate vertical avoidance behaviour amongst birds approaching offshore windfarms.

Krijgsveld *et al.* (2011) demonstrate that a number of species may fly at lower altitudes within-windfarms than outside windfarms and incorporate vertical avoidance behaviour in their estimation of micro-avoidance rates using a combination of visual and radar observations. Their results suggest that a substantial proportion of birds may alter their flight altitudes in order to avoid collision. Given the development of technologies capable of monitoring the movement of birds close to turbines, such as the Thermal Animal Detection System (Desholm *et al.* 2006), these results suggest that focussing on vertical avoidance at a micro-meso, as opposed to macro, scale

may be worthwhile. At a micro-scale, it is likely that vertical avoidance would be captured as part of an evasive manoeuvre.

3.4 Total avoidance rates

In this section, we have produced definitions that are considered to work within the constraints of our current understanding of avoidance behaviour and data collection limitations. It is clear, given the multiple potential components of avoidance behaviour that we have identified (Figure 3.5), that equation 7 is an over-simplification of overall avoidance rates. In future studies it is important to consider how each of these components can be quantified. As technological capabilities advance, the definitions outlined above may become obsolete. However, any refinement to these definitions should be based on the behaviour of the species concerned, rather than artificially induced by methodological constraints, for example, the distance over which observations can be made with the use of binoculars or telescopes.



*Macro-responses may occur in the vertical plane, However, technical limitations mean it is unlikely to be possible to measure this

Figure 3.5 Schematic detailing how different behavioural responses to offshore windfarms may combine to give a total avoidance rate. At each different level birds may respond either vertically or horizontally. Outside a windfarm, both displacement and barrier effects are likely to contribute to the macro-response rate. However, the contribution of displacement to macro-avoidance may be hard to quantify as a result of uncertainty associated with estimating its effects. Avoidance behaviour inside a windfarm is often termed micro-avoidance, however, it may be appropriate to split this term further by considering a meso-response, where birds enter a windfarm but do not pass close to turbines, and micro-avoidance, where birds take last minute action to avoid collisions.

3.5 Recommended Definitions

For the purposes of this review, the definitions we will use for bird behaviour in response to offshore windfarms and turbines are (Figure 3.3):

> **MACRO-RESPONSE –** The response of birds to the presence of the windfarm outside its perimeter, defined as a 500 m buffer surrounding the outermost turbines. Responses may include attraction to the windfarm, displacement from preferred foraging habitat or an alteration to flight paths as a result of seeing the windfarm as a barrier. These may occur in either horizontal or vertical planes, although at present technological limitations mean that it is not possible to measure vertical macro-responses. For this reason, for the purposes of this review, we consider only horizontal macro-responses.
>
> **MESO-RESPONSE –** A redistribution of birds, or alteration of flightpaths within a windfarm in response to the presence of the turbines. This may encompass both horizontal and vertical responses. These responses are in contrast to micro-avoidance, see below.
>
> **MICRO-AVOIDANCE** – Last-second action taken by birds flying at rotor height to avoid collision, encompassing both horizontal and vertical movements, within a 10 m buffer surrounding turbine rotor-swept areas.

Due to current methodological difficulties in distinguishing micro-avoidance behaviour from meso-response behaviour, a fourth category is defined for the purposes of this review to act as a proxy for responses to windfarms at these scales:

> **WITHIN-WINDFARM AVOIDANCE –** Encompassing both meso-responses and micro-avoidance to describe how birds within a windfarm respond to the presence of a turbine.

The review focuses on data relating to macro-responses and within-windfarm avoidance. Distinctions between responses at the meso- or micro-scale and horizontal or vertical responses have not been made at this stage as insufficient data are available to support them. Future studies should aim to be able to make such distinctions to improve our understanding of avian avoidance behaviour at offshore windfarms.

4. REVIEW OF AVOIDANCE RATES USED IN COLLISION RISK MODELLING FOR OFFSHORE WINDFARMS

We reviewed the use of avoidance rates in collision risk modelling as part of the impact assessment process for 35 consented or proposed offshore windfarms (Table 4.1). There was considerable variation between assessments in the rates selected, which were as low as 0.87 and as high as 0.9999. In the majority of cases, a single avoidance rate for all species, ranging from 0.95 to 0.99, has been used in the collision risk modelling process to assess the potential impacts for all species considered. However, in some instances, developers and their consultants have felt that sufficient evidence exists to consider higher rates for some species, notably terns, although these values have not always been accepted within the decision process.

The species assessed during the collision risk modelling process vary on a site by site basis. This typically reflects the distribution of these species, for example, with Manx shearwater likely to be assessed at sites on the west coast of the UK. However, some species, such as northern gannet, black-legged kittiwake, lesser black-backed gull and great black-backed gull, are considered in most assessments, reflecting the broad scale distributions of these species. The flight height of birds is also an extremely important factor in determining the likely risk of collision (Johnston *et al.* 2014a). In several early assessments, a screening process was also carried out whereby species for which only a small proportion of individuals (typically <1%) were recorded flying at heights placing them at a risk of collision were excluded from the collision risk modelling process (Table 4.1). As a result of this screening process, the collision risk of some species, such as auks and divers, was assumed to be negligible and therefore not assessed using collision risk models.

In early assessments, the avoidance rates used in collision risk modelling were often very high, typically in excess of 0.99. The use of these rates was largely founded on collision rates reported at onshore windfarms (e.g. Winkelman 1992, Everaert 2003). However, these do not reflect true avoidance rates as they do not account for birds which pass safely through the rotor swept area of the turbines without taking avoidance action, or indeed those which pass through the windfarm without entering the rotor-sweep of the turbines.

In 2005, SNH issued guidance for sensitive bird species commonly identified in (onshore) windfarm environmental statements (SNH 2010) that a default avoidance rate of 0.95 should be used. This figure was based on expert opinion (Andy Douse *pers. comm.*) and acknowledged as being precautionary. It was felt that, as evidence became available, this rate would be revised upwards. Of the 13 assessments for offshore windfarms published between 2005 and the revision of this guidance in 2010 (SNH 2010), seven followed this guidance (see Table 4.1). The remaining assessments which argued that higher avoidance rates were more appropriate, cited as part of their justification empirical data of collision rates collected from sites in Belgium (see Everaert 2003, Everaert and Stienen 2006, Everaert 2008) or assessments of species' manoeuvrability as determined by Garthe and Hüppop (2004) and Maclean *et al.* (2009).

Following evidence obtained from onshore windfarms suggesting avoidance rates were likely to be significantly higher than 0.95 (Fernley *et al.* 2006, Pendlebury 2006, Whitfield and Madders 2006, Whitfield 2009) the default values were revised by SNH (2010). A default rate of 0.98 was recommended for all species considered in this guidance which included gull spp., tern spp, skua spp and diver spp. Exceptions to the default value included geese, hen harrier and golden eagle, for which sufficient evidence was available to support a 0.99 avoidance rate, and kestrel and white-tailed eagle, for which the 0.95 avoidance rate was retained as it was felt they were particularly susceptible to collisions. Again, a significant proportion (12 out of 18) of environmental impact assessments for offshore windfarms published since 2010 follow this guidance. The remaining studies cite evidence from Belgium (Everaert 2003, Everaert and Stienen 2006, Everaert and Kuijken 2007, Everaert 2008) and the Netherlands (Leopold *et al.* 2011, Krijgsveld *et al.* 2011), or again base avoidance rates on assessments of species' manoeuvrability as determined by Garthe and Hüppop (2004) and Maclean *et al.* (2009) in support of higher avoidance rates. As part of our review, we consider the strength of the quantitative evidence put forward in these studies and how qualitative information may be used to support these conclusions.

The evidence base for the revised avoidance rates is largely based on collision mortality observations at onshore / coastal windfarms – although recent behavioural avoidance evidence from Egmond aan Zee (Krijgsveld *et al.* 2011) is also being used – and there are uncertainties around the applicability of these values to offshore windfarms (Trinder 2012). First, whilst some seabird species may be attracted to offshore windfarms, others such as northern gannet show evidence of macro-avoidance (e.g. Krijgsveld *et al.* 2011, Vanermen *et al.* 2013) (see section 5.1). In contrast, while some terrestrial species, such as geese, may also show strong macro-avoidance of offshore windfarms (Plonczkier & Simms 2012), macro-avoidance is often less likely at terrestrial windfarms (e.g. Devereux *et al.* 2008, Garvin *et al.* 2011, Pearce-Higgins *et al.* 2012). As a result, avoidance rates in relation to offshore windfarms need to capture not just avoidance of the individual turbines, as is the case for species at terrestrial sites, but also of the windfarm itself.

Secondly, estimates of avoidance derived from collision mortality rates (rather than direct observations of avoidance – 'behavioural avoidance') follow the formula given in SNH (2010), whereby observed mortality is divided by the mortality expected in the absence of avoidance based on the flux of birds through the rotor-swept area (equation 6).

Surveys for terrestrial windfarms are usually carried from vantage points within 2 km of the area to be observed, ensuring that all observations are within 2 km. However, these methodologies rarely employ distance correction which means that the flux rates of birds (or population estimates) are likely to be underestimated. If the numbers of birds passing through the rotor-swept area of a turbine, and therefore the expected numbers of collisions, are underestimated, the derived avoidance rate will also be an underestimate. In contrast, population sizes within offshore windfarms of each of the five priority species considered as part of this review may potentially be over-estimated, given the attraction of each to boats (e.g. Garthe & Hüppop 1994, Skov & Durinck 2001). Even where population data have been collected from other platforms, for example, by digital aerial survey (e.g. Buckland *et al.* 2012), the

potential for underestimating population size is considerably less than for surveys of onshore windfarms. As populations within offshore windfarms are unlikely to be underestimated, it has been argued (Trinder 2012) that an avoidance rate suitable for estimating collisions at an onshore windfarm will lead to underestimation of avoidance behaviour if used for estimating collisions at an offshore windfarm.

This review highlights the reliance of offshore windfarm developers, and their consultants, on guidance from Statutory Nature Conservation Bodies (SNCBs) about the use of appropriate avoidance rates. Of the 35 studies we identified, 19 cited the SNH guidance from either 2005 or 2010 in support of the avoidance rates selected for some, or all of their study species. Of these studies, several have suggested that these avoidance rates are potentially overly-precautionary, citing evidence from Belgium (Everaert 2003, Everaert and Stienen 2006, Everaert 2008), and the Netherlands (Winkelman 1992, Krijgsveld *et al.* 2011). The use of avoidance rates in excess of 0.98 in a number of recent environmental statements may reflect an increasing concern amongst developers that the SNH guidance is overly precautionary and posing an unnecessary risk to the consenting process. Many of the early developments were relatively small scale and consequently, collision risk estimates, even with an avoidance rate of 0.95, were extremely low. However, the scale of many of the developments proposed more recently is significantly greater, with commensurate increases in estimated collision rates. Consequently, it is important the subsequent review of avoidance rates can clarify the situation for developers and SNCBs alike.

Table 4.1 Avoidance rates considered during the collision risk modelling undertaken in assessments for proposed offshore windfarms and the justification for their use. All avoidance rates were used in conjunction with the basic (option 1) Band model and were taken from the final submitted environmental statements.

Offshore windfarm	Year	Avoidance rate(s) used	Species considered	Justification
Kentish Flats	2002	0.9998	Red-throated diver	Collision rate of 0.02% presented in Winkelman (1992)
Burbo Bank	2002	No Collision Risk Modelling	Red-throated diver, common scoter, common tern, wader sp., great cormorant, red-breasted merganser, little gull, common guillemot/razorbill	Sensitive species flew below rotor height and, therefore, were not at risk of collision
North Hoyle	2002	No Collision Risk Modelling	Red-throated diver, great cormorant, common scoter, tern sp., European shag, common guillemot, razorbill	Sensitive species flew below rotor height and, therefore, were not at risk of collision
Teesside	2004	0.9962 for all species	Red-throated diver, northern gannet, great cormorant , waders, Arctic skua, herring gull, great black-backed gull, black-legged kittiwake, Sandwich tern, common tern, common guillemot, geese sp.	Based on calculations from Blyth Harbour (citing Still *et al.* 1996, Painter *et al.* 1999)
Beatrice Demonstration Site	2005	0.95 for all species	Black-legged kittiwake, great black-backed gull, northern fulmar, northern gannet, auk spp, herring gull, tern spp	Follows SNH guidance from 2005 (SNH 2010) and is acknowledged as a conservative value.

Offshore windfarm	Year	Avoidance rate(s) used	Species considered	Justification
Thanet	2005	0.99 for all species	Red-throated diver, northern fulmar, northern gannet, common tern, Sandwich tern, black-legged kittiwake, common gull, herring gull, lesser black-backed gull, gull spp, auk spp	In line with previously published estimates of avoidance (Percival 1998, Everaert *et al.* 2002, Henderson *et al.* 1996, Winkelman 1992, Winkelman 1990, Percival 2001, Still *et al.* 1996)
London Array	2005	0.995 and 0.999 for gull spp, tern spp and Northern gannet, and 0.99 and 0.995 for diver sp.	Red-throated diver, black-throated diver, herring gull, lesser black-backed gull, great black-backed gull, common tern, northern gannet, Sandwich tern	Based on vulnerability to collision as assessed by Garthe & Hüppop (2004) and observed collision rates for gulls and terns presented by Everaert (2003)
Greater Gabbard	2005	High (0.9999), Medium (0.9982) and Low (0.87) for all species	Red-throated diver, lesser black-backed gull, great skua	High and Medium rates calculated from data presented in Winkelman (1992) based on total collisions numbers for gulls (High) and nocturnal collisions for gulls (Medium), Low avoidance rate derived from lowest reported avoidance rate of 0.87 found in American kestrel and considered highly unrealistic
Gwynt Y Mor	2005	No Collision Risk Modelling	Diver sp., northern fulmar, Manx shearwater, Leach's petrel, northern gannet, common scoter, small skua spp, great skua, black-legged kittiwake, Sandwich tern, 'comic' tern, common guillemot/razorbill	Sensitive species flew below rotor height and, therefore, were not at risk of collision

Offshore windfarm	Year	Avoidance rate(s) used	Species considered	Justification
Sheringham Shoal	2006	0.98 for all species	Sandwich tern, common tern, northern gannet, little gull, lesser black-backed gull	SNH guidance from 2005 (SNH 2010) guidance felt to be over-precautionary
West of Duddon Sands	2006	0.999	Lesser black-backed gull	Based on vulnerability to collision as assessed by Garthe & Hüppop (2004) and observed collision rates for gulls presented by Everaert (2003)
Humber Gateway	2007	0.95 for all species	Red-throated diver, northern gannet, great skua, Arctic skua, little gull, black-headed gull, common gull, black-legged kittiwake, herring gull, great black-backed gull, lesser black-backed gull, Sandwich tern, common tern, Arctic tern	Follows SNH guidance from 2005 (SNH 2010) and is acknowledged as a conservative value
Lincs	2007	0.95 for all species	Pink-footed goose, red-throated diver, northern gannet, little gull, common gull, lesser black-backed gull, common tern, Common guillemot	Follows SNH guidance from 2005 (SNH 2010) and is acknowledged as a conservative value
Westernmost Rough	2009	0.95 for all species	Northern gannet, black-legged kittiwake, common gull, lesser black-backed gull, herring gull, great black-backed gull, common tern	Follows SNH guidance from 2005 (SNH 2010) and is acknowledged as a conservative value

Offshore windfarm	Year	Avoidance rate(s) used	Species considered	Justification
Race Bank	2009	0.996 for Sandwich tern, 0.95 for all other species	Sandwich tern, common tern, northern fulmar, little gull, northern gannet, lesser black-backed gull, black-legged kittiwake, common guillemot, razorbill	Sandwich tern avoidance rate based on data from Zeebrugge, SNH guidance from 2005 (SNH 2010) for other species, but also discussion as to whether higher avoidance rates may be appropriate in some cases (northern gannet and lesser black-backed gull)
Dudgeon	2009	0.996 for Sandwich Tern, 0.99 for lesser black-backed gull, 0.97 Northern gannet	Sandwich tern, lesser black-backed gull, northern gannet	Evidence presented in Everaert & Stienen (2006) & Everaert (2008) for Sandwich tern and recommendations in Maclean *et al.* (2009) for northern gannet and lesser black-backed gull
LID6	2010	0.95 for all species	Black-throated diver, great northern diver, northern gannet, dark-bellied brent goose, little gull	Follows SNH guidance from 2005 (SNH 2010) and is acknowledged as a conservative value
Triton Knoll	2011	0.98 for all species	Northern fulmar, little gull, black-legged kittiwake, Sandwich tern, northern gannet, common guillemot, Arctic skua, lesser black-backed gull, great black-backed gull, common tern	Follows SNH guidance from 2005 (SNH 2010) guidance
Galloper Offshore Windfarm	2011	0.99 for gulls, 0.98 for other species	Red-throated diver, northern gannet, Arctic skua, great skua, common gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake	Evidence from 'vantage point surveys' for gulls, follows SNH (2010) guidance for all other species

Offshore windfarm	Year	Avoidance rate(s) used	Species considered	Justification
Rampion	2011	0.995 for Northern gannet, Gulls sp. , skuas spp and Auks, 0.99 for terns sp. and waterbirds	Brent goose, common scoter, northern gannet, bar-tailed godwit, Eurasian curlew, great skua, Mediterranean gull, common gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake, Sandwich tern, common guillemot, barn swallow, meadow pipit	Follows Maclean *et al.* (2009)
Aberdeen Offshore Windfarm	2012	0.98 for all species	Common guillemot, razorbill, Atlantic puffin, northern fulmar, common tern, Sandwich tern, herring gull, black-legged kittiwake, great black-backed gull, common gull, common scoter, common eider, European shag, great cormorant, northern gannet, red-throated diver, Arctic skua	Follows SNH (2010) guidance
Blyth Offshore Demonstration Project	2012	0.98 for all species	Northern gannet, common gull, herring gull, great black-backed gull, little gull, black-legged kittiwake, common tern	Follows SNH (2010) guidance

Offshore windfarm	Year	Avoidance rate(s) used	Species considered	Justification
Hornsea Project One	2012	0.98 for all species	Northern fulmar, northern gannet, black-legged kittiwake, little gull, common gull, great black-backed gull, lesser black-backed gull, herring gull, common tern, Arctic tern, common guillemot, razorbill, Arctic skua, great skua	Follows SNH (2010) guidance
Irish Sea	2012	0.98 for all species	Manx shearwater, great black-backed gull, lesser black-backed gull, herring gull, black-legged kittiwake, northern gannet, Greenland white-fronted goose	Follows SNH (2010) guidance
East Anglia One	2012	0.98 for all species	Northern fulmar, northern gannet, black-legged kittiwake, common gull, lesser black-backed gull, herring gull, great black-backed gull	Follows SNH (2010) guidance
Firth of Forth Alpha and Bravo	2012	0.98 for all species	Northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull, great black-backed gull	Follows SNH (2010) guidance
Beatrice Offshore Windfarm	2012	0.99 for all species	Arctic skua, Arctic tern, northern fulmar, great black-backed gull, northern gannet, herring gull, black-legged kittiwake, great skua, common guillemot, razorbill	Review of micro-and macro-avoidance rates and criticism of the transferability of avoidance rates between onshore and offshore windfarms in MacArthur Green (2012)

Offshore windfarm	Year	Avoidance rate(s) used	Species considered	Justification
Dogger Bank Creyke Beck A and B	2012	0.99 for northern gannet, 0.98 for all other species	Northern fulmar, northern gannet, great skua, Arctic skua, black-legged kittiwake, lesser black-backed gull, great black-backed gull, common guillemot, razorbill, little auk, Atlantic puffin	Evidence from Egmond aan Zee(Krijgsveld *et al.* 2011) and elsewhere supporting 0.99 for northern gannet and following SNH (2010) guidance for all other species
Moray Firth Offshore Windfarm	2012	0.995 for northern gannet, 0.985 for lesser black-backed gull, 0.99 for black-legged kittiwake	Northern gannet, black-legged kittiwake, herring gull, great black-backed gull	Consideration of micro-and macro-avoidance rates presented for Dutch and Belgian windfarms (Everaert 2008, Krijgsveld *et al.* 2011)
Nearth na Gaoithe	2012	0.998 for northern gannet, 0.995 for gulls spp., 0.98 for Arctic tern	Northern gannet, little gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake, Arctic tern	High macro-avoidance rates for northern gannet presented in Leopold *et al.* (2011) suggest that avoidance rates presented in both SNH (2010) guidance and MacLean *et al.* (2009) are likely to be over precautionary for northern gannet. Tern and gull avoidance rates follow Maclean *et al.* (2009)
Bligh Bank Windfarm (Belgium)	2013	0.976 micro-avoidance rate for all species	Common gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake	Based on rates estimated at Egmond aan Zee by Krijgsveld *et al.* (2011)
Walney Extension Offshore Windfarm	2013	0.98 for all species	Whooper swan, pink-footed goose, lesser black-backed gull	Follows SNH (2010) guidance

Offshore windfarm	Year	Avoidance rate(s) used	Species considered	Justification
Burbo Bank Extension	2013	0.98 for all species	Red-throated diver, Manx shearwater, common scoter, little gull, black-headed gull, herring gull, lesser black-backed gull, common tern, Arctic tern, Sandwich tern, great cormorant, northern gannet, Arctic skua, great skua, black-legged kittiwake	Follows SNH (2010) guidance
Atlantic Array	2013	0.98 for all species	Manx shearwater, northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull, great black-backed gull, common guillemot	Follows SNH (2010) guidance
Inch Cape	2013	0.99 for northern gannet, 0.98 for all other species	Northern gannet, Arctic skua, pomarine skua, great skua, black-legged kittiwake, great black-backed gull, herring gull	Evidence presented from Egmond aan Zee to justify 0.99 for northern gannet, follows SNH (2010) guidance for all other species

5. REVIEW OF PUBLISHED EVIDENCE FOR AVOIDANCE RATES OF MARINE BIRDS

This section provides a review of published evidence for macro-response (section 5.1), meso-response alone (section 5.2), micro-avoidance alone (section 5.3) and overall within-windfarm avoidance (i.e. combined micro-/meso-avoidance; section 5.4). For macro response rates (section 5.1) we consider data collected from the offshore environment only. The more limited evidence base for meso-response and micro-avoidance rates (sections 5.2 and 5.3) meant that it was necessary to include some evidence from the onshore environment. The difficulties in obtaining estimates of collision rates in the offshore environment mean that the majority of the evidence that relates to within-windfarm avoidance rates (section 5.4) originates from the terrestrial environment.

5.1 Review of Published Evidence for Macro-response Rates of Marine Birds

Here we consider macro-responses as including: (i) barrier effects for migrating birds or those commuting between breeding colonies and foraging areas; (ii) displacement effects from the windfarm area leading to an effective loss of habitat; and (iii) attraction. Each of these responses may result in a change in the numbers of birds in flight present within the perimeter of the windfarm between the pre- and post-construction periods. As collision risk modelling is usually based on the number of birds present during the pre-construction period, these changes must be accounted for as part of the collision risk modelling process. All of the studies we consider in this section originate from the offshore environment.

5.1.1 Causes of barrier, displacement and attraction effects

The term barrier effects describes the behavioural response of flying birds to the presence of the windfarm. The windfarm acts as a physical barrier, impeding the most direct route to a bird's destination, necessitating a change in flight direction in order to avoid entering the windfarm. This will ultimately reduce the numbers of birds recorded in flight within the windfarm area.

The effects of displacement are harder to classify since the habitat within the area of the windfarm may have been used by birds for a variety of purposes, notably foraging, but potentially other essential maintenance behaviours, such as moulting, preening or forming rafts. The availability of alternative foraging habitat may be more restricted, however, and hence for the purpose of this review we consider displacement as the inability of a bird to forage in a particular area due to the presence of the turbines. This may be manifested as a reduction in the number of birds flying into the area of the windfarm to look for food but this does not necessarily mean that birds will no longer enter the windfarm. It is possible, for example, that some species may land outside the windfarm and swim into the windfarm area. Studies of displacement, however, have tended to report the changes in all observed birds within the windfarm's perimeter relative to the areas outside and have not differentiated between the numbers flying and those recorded on the water. To better inform both studies of displacement and macro-avoidance, it would be prudent in future studies to separate flying birds from birds on the water when reporting displacement rates. Another important consideration relates to the flight height

information that may be collected during surveys. This is primarily used to inform collision risk, but could potentially be used to inform on the vertical avoidance of birds over or under the rotor swept area.

Attraction is defined as an increase in numbers of birds within the windfarm area post-construction and can arise though several means. The monopiles of the turbine can act as a useful platform for birds to dry their feathers, rest, and socialise (e.g. great cormorant, Lindeboom *et al.* 2011). There is also evidence that structure of the turbines may also provide feeding opportunities through changes in local hydrography, seabed morphology or by acting as an artificial reef (Inger *et al.* 2009, Wilson & Elliot 2009, Maar *et al.* 2009, Lindeboom *et al.* 2011). Whilst there is the potential for collision risk to increase, as a result of attraction into the windfarm area, this will only occur if birds utilise the space covered by the rotor swept area.

5.1.2 Overall approach to assessing evidence for barrier, displacement and attraction effects

In reality, the ability to differentiate between birds exhibiting barrier and displacement effects may not always be possible since both are manifested as a decrease in the numbers of birds within the windfarm area (as defined both horizontally and vertically). For the purpose of this review, however, we will critique studies carried out at windfarms according to the type of effect they were designed to look at. For each example we present the relevant methods, key results and an overall assessment of the appropriateness of their use in looking at the effect they were designed to measure. Although our brief was to examine the evidence for five key species being considered in this review, we have also included several examples which have been cited as providing supporting evidence of macro-avoidance for seabirds in general (e.g. Desholm & Kahlert 2006 study on common eider and geese spp). We have not included studies carried out solely on migrating terrestrial species, e.g. such as pink-footed geese at Lynn and Inner Downing windfarm (Plonczkier & Simms 2012).

5.1.3 Studies of barrier effects

5.1.3.1 Methodologies used to look at barrier effects

Barrier effects have been measured mostly using (horizontal) radar and/or visual observations from fixed observation points (see Table 5.1 for summary). Radar technology has been used to measure barrier effects directly by quantifying the percentage of bird tracks that are deflected away from the windfarm, and also to look at the distance at which the deflection occurs (e.g. Peterson *et al.* 2006). However, due to technological constraints of horizontal radar (see below), this has been limited to quantifying horizontal macro-responses only. Radar has also been used to look at barrier effects indirectly by comparing the number of flight paths (tracks) inside and outside the windfarms (e.g. Krijgsveld *et al.* 2011), to look at the densities of tracks in relation to distance from the windfarm (e.g. Skov *et al*. 2012) or to look at percentages of flight paths flying towards, away from and parallel to the windfarm (Blew *et al.* 2008). Such indirect measures may not necessarily be able to differentiate between barrier and displacement effects however. Visual observations, whilst also critical for the validation of the results of the radar, in terms of providing

species identification and relative abundance, have also been used independently to compare numbers of birds in flight inside and outside the windfarm (e.g. Krijgsveld *et al.* 2011) although again, these methods may not necessarily preclude the possibility of inadvertently measuring displacement effects. Emerging technology in the form of laser range finders has also recently been used (e.g. Skov *et al.* 2012), and there may be scope to apply this approach in the context of barrier effects. There have been examples, notably in the UK, where data collected from boat based surveys have been used to look at barrier effects but this methodology is not considered to adequately provide the quantification needed here (MMO 2014).

There are a number of limitations associated with the use of radar (for further discussion see Krijgsveld *et al.* 2011 and Peterson *et al.* 2006) in terms of deriving avoidance rates: (i) Identification to the species level is not possible without visual validation and even then this information is generally only available as the species composition of birds passing through in a comparable time period – hence the values cited may be considered relevant only to the most commonly recorded birds species; (ii) There can be problems with distinguishing between flocks or individual birds – tracks recorded by radar may therefore not necessarily correspond to individual birds and corresponding avoidance rates could be more representative for flocks (which are likely to vary in size); (iii) Detection issues exist with picking up individual birds or flocks of small birds; (iv) Detection of birds can be affected by environmental conditions such as wave height and rain; (v) Seabirds such as the northern gannet, tubenose spp, sea duck spp and alcid spp tend to fly in the troughs between waves (as a means of flying in the most energetically efficient manner). In conditions where the wave height is sufficiently high, the total number of these birds is likely to be underestimated; (vi) The relative orientation of the radar beam to the flight direction of the birds can also affect detection (flying head on into the beam is the best) – this can present challenges when considering the optimum position for the radar; (vii) Whilst the range of detection for radar exceeds that of visual observations, there is a risk that birds could start to change their flight orientation beyond the range of the radar which would result in birds not being detected at all and hence the relative contribution of barrier effects to macro-responses is underestimated; (viii) Detection rates have been shown to be lower inside the windfarm due to interference caused by the presence of the windfarms (this is covered more extensively under the site accounts). Another considerable limitation of radar is that horizontal radar can only be used to record horizontal displacement (sometimes referred to as lateral displacement) as no information on altitude is collected. It is possible, therefore, that birds may fly over the windfarm at altitudes higher than the rotor swept area but this would not be picked up as avoidance behaviour (Blew *et al.* 2008). In contrast, vertical radar can only be used to determine flight height (altitude) and densities of birds in passage (flux) directly above the radar itself and provides insufficient information either on horizontal change or vertical avoidance that takes place outwith the windfarm perimeter. Radar has been useful, however, in demonstrating the importance of time of day (day versus night time), wind direction (head versus tail wind), season (spring versus autumn) for avoidance rates (e.g. Peterson *et al.* 2006 and Krijgsveld *et al.* 2011).

In terms of data collection issues for visual observations, there are also limitations when compared to radar: (i) Sampling is limited to daylight with reasonably calm conditions and good visibility. Although, under some circumstances, observations at

night (e.g. moon watching) or auditory observations (based on bird calls) have been used, these have limited use; (ii) The range of detection is smaller; (iii) Individual observers may differ in assessing the distance and altitudes of birds, although there may be scope to reduce such differences through calibration with other techniques (Mateos *et al.* 2010; Norman *et al.* 2005).

5.1.3.2 Results of studies on barriers effects

Overall there is very little species-specific evidence for the five priority species for macro-avoidance as a consequence of barrier effects (see Appendix 1 for detailed site accounts) as radar was the most commonly used method. Of the studies reviewed all but one study looked at barrier effects during the post-construction period only – the exception being Nysted (Desholm & Kahlert 2005; Peterson *et al.* 2006) which also looked at the pre-construction period. Arguably comparison of the pre- and post-construction periods provides the most robust evidence for barrier effects rather than focussing solely on the post-construction period. Avoidance rates were only derived for three windfarms (see Table 5.1): (i) Egmond aan Zee (Krijgsveld *et al*. 2011); (ii) Nysted (Desholm & Kahlert 2005; Peterson *et al.* 2006); and Horns Rev (Peterson *et al*. 2006). The latter two are not considered further here since the derived values are likely only to be relevant to common eider (and geese) and common scoter respectively. Whilst there has been some additional work carried out at the Alpha Ventus test site to look at barrier effects (BSH 2011 and Mendel *et al.* 2014), the data have not been presented in such a way that would allow the derivation of a macro-avoidance rate and are hence not considered further here.

The only study which has specifically looked at barrier effects for northern gannet was that of Krijgsveld *et al.* (2011) at Egmond aan Zee which derived a macro-avoidance rate of 0.64. This was derived from indirect measure of barrier effects using visual observations made during panoramic scans to calculate the number of birds in flight within, at the edge and outside the windfarm (and by using a factor to correct for relative surface area – see Appendix 1, section A1.1). It is therefore not possible to discount the possibility that the apparent decreases within the windfarm could have included displacement due to the methodology used. These data were based on a total of 405 panoramic scans from spring 2007 to the end of 2009 (see Table 4.2 Krijgsveld *et al.* 2011) with particular emphasis on the spring and autumn periods as a total of 140 and 121 scans were carried out respectively compared to 71 and 73 scans in the summer and winter respectively. Overall, the sample sizes of the numbers of flying birds observed for northern gannet and common scoter were 282 and 123, although these figures were not broken down on a seasonal basis. However, it is also worth highlighting that northern gannets' use of the area – based on the density of flying birds – was highest during the spring, autumn and winter with an order of magnitude less use during the summer (mean density or numbers of birds per km^2 per scan for the periods of spring, summer, autumn and winter were 0.03, <0.005, 0.05 and 0.02 respectively – see Table 8.3 Krijgsveld *et al.* 2011). The extent to which the derived macro-avoidance rate is representative of breeding birds is thus questionable due to the relatively low use of the Egmond aan Zee site at this time and the lower sampling frequency. Therefore until such time that data are collected on northern gannet flights around OWFs specifically during breeding, this value should be applied with caution when considering the breeding season. It is also worth noting that Krijgsveld *et al.* (2011) reported a deflection rate of 0.89 for

northern gannet based on the assessment of visual observations of flight paths. However, this result was based on a sample size of 38 birds and these observations were not based on systematic recording methods (c.f. the panoramic scans, which were based on strict protocols and recorded all birds seen). Consequently the authors do not recommend that these values be used as macro-avoidance rates (Karen Krijgsveld *pers. comm.*). Note, however, that these deflection rates have been cited as evidence for macro-avoidance rates by industry (e.g. Natural Power 2013).

There are no species-specific macro-avoidance rates, relating to barrier effects, for any of the four priority gull species of this review. Arguably, the most relevant study is that of Krijgsveld *et al*. (2011) which derived a macro-avoidance rate of 0.18 for the generic group of gull spp. These data were based on the indirect measure of barrier effects of the relative percentage of tracks that were outside the windfarm in winter. This was justified on the grounds that the species composition of bird tracks was heavily dominated by gulls spp (and great cormorants) at that time of year. A deflection rate of 0.4 was reported (based on the flight paths for 78 birds recorded as gull spp) but, as before, this value is not derived from systematic recording methods and the authors do not recommend this as evidence as macro-avoidance.

Table 5.1 Summary of key studies of barrier effects, the stage of data collection, methods used, parameters measured and species or species groups reported. *Italics* indicates species for which values were based on averages of other species (see Appendix 1 for site accounts)

Windfarm site	Study	Stage of data collection	Method used/parameter measured	Species/spp groups (values of macro-avoidance are given in parentheses where available)	Time of year data collected
Egmond aan Zee	Krijgsveld *et al.* (2011)	Post-construction	Radar /Numbers of tracks inside and outside the windfarm	*Gull spp (0.18)* *Grebe spp (0.28)* *Tubenoses spp (0.28)* *Skua spp (0.28)* *Tern spp (0.28)*	Winter All year All year All year All year
			Visual observations /Numbers of birds inside and outside the windfarm	*Northern gannet (0.64)* *Seaducks/scoter (0.71)* *Diver spp (0.68)* *Alcid spp (0.68)*	All year All year All year All year
Horns Rev I	Peterson *et al.* (2006)	Post-construction	Radar / The percentage of tracks that were considered to have a theoretical chance of entering the windfarm Radar / The distance at which deflection occurs	Common scoter (range 0.71-0.86 based on inter-annual variation and the direction at which birds approach the windfarm)	Spring/autumn combined
Horns Rev I	Blew *et al*. (2008)	Post-construction	Radar - Orientation of tracks in relation to the windfarm (% flying towards, away or parallel to the windfarm)	All birds	Spring/autumn combined
Horns Rev I and II	Skov *et al*. (2012)	Post-construction	Radar / Densities of tracks in relation to the radar station and windfarm	Common scoter and all birds	Spring/autumn combined

			Laser range finders / The distance at which deflection occurs (based on peak densities of radar tracks)		
Nysted	Peterson *et al.* (2006) Desholm & Kahlert (2005)	Pre- and post-construction	Radar / The percentage of tracks that were considered to have a theoretical chance of entering the windfarm Radar / The distance at which deflection occurs	Common eider and geese spp (0.78 – inter-annual variation 0.63-0.83)	Autumn
Nysted	Blew *et al*. (2008)	Post-construction	Radar / Orientation of tracks in relation to the windfarm (% flying towards, away or parallel to the windfarm)	All birds	Spring/autumn combined

5.1.4 Studies of displacement/attraction

5.1.4.1 Methodologies (and survey design) used to look at displacement/attraction

Data used to look at displacement effects have tended to be derived from boat and/or visual aerial surveys. Whilst industry guidance (Camphuysen *et al.* 2004) tends to be cited as the basis of the methodologies used, the extent to which guidelines are followed may be unclear (Maclean *et al.* 2009). Although digital aerial surveys are becoming more commonly used by the offshore windfarm industry (MMO 2014, Mackenzie *et al.* 2013), there appears to be a lack of sites where this technology has been used during all phases of the development. Further consideration of the advantages and disadvantages of these methodologies are given in Mackenzie *et al.* (2013). Additional to these, and of relevance to the assessment of displacement, there are concerns that boat surveys may overlook birds flying at higher altitudes and that might, therefore, fly over windfarms (Hartman *et al.* 2012). As is true for most of the studies designed to target barrier effects which may not necessarily exclude displacements effects, the same is true for the reverse situation.

Studies of displacement effects carried out at offshore windfarm sites within the UK have largely been based on the Before and After Control Impact (BACI) design which was viewed as being best practice at the time these sites were being set up (based on Stewart-Oaten *et al.* 1986). The extent to which this approach has been successfully implemented has been hampered by inadequate survey design including: (i) Location of the reference site often situated immediately adjacent to the impact site of the windfarm area – thus any changes as result of the windfarm may be over- estimated e.g. displaced birds could move into the adjacent area resulting in higher numbers recorded than during the pre-construction period; (ii) Insufficient spatial coverage e.g. boat surveys often only covered the windfarm area and a buffer, hence any possible changes that may have occurred in the wider environment cannot be taken account of; (iii) Gaps in temporal coverage e.g. survey periods between the different phases of the development did not always correspond or visual aerial surveys having to be abandoned following construction of the windfarm, due to Civil Aviation Authority flight height restrictions; (iv) The ability to select control sites which are truly comparable to the area impacted by the windfarm area (e.g. in terms of hydrography, seabird populations) has been questioned. For further consideration of these issues see MMO (2014).

A further limitation of displacement studies in their survey design is that little consideration is usually given to the power to detect change, which is related to a number of factors including the frequency of surveys and their relative spatial and temporal coverage (Maclean *et al.* 2013; Vanermen *et al.* 2012; Pérez Lapeña *et al.* 2010). The distribution and relative abundance of seabirds show high levels of both spatial and temporal variability within and between years. Therefore the use of power analyses, particularly at the start of any offshore windfarm development, can be extremely helpful in determining the most appropriate survey design in order to be able to adequately test for whether a windfarm impacts birds through either displacement or attraction effects.

There is also the problem that the post-construction reports, notably those leading up to the final report, have tended not to provide formal statistical analyses and any assessments of changes in species abundance are often based on simple comparisons of changes in absolute numbers or are qualitative (e.g. visual inspection of maps: MMO 2014). Even in instances where the significance of change has been looked at, the focus has been on measuring differences in numbers or densities between the pre-construction and post construction periods and any changes in distribution within the study area may go undetected (MacKenzie *et al.* 2013). Recently, there have been developments in model-based approaches such as density surface modelling (Rexstad 2011) which allow the inclusion of covariates (e.g. environmental factors such as water depth, sea surface temperature) which can help better explain inherent spatial and temporal variability in the abundance and distribution of animal populations. The resulting distribution maps of relative abundance provide a more robust means of assessing whether changes have occurred as a result of the presence of an offshore windfarm. There also appears to be a shift away from using BACI survey design for looking at displacement, with a Before-After-Gradient approach being recommended (MMO 2014, Jackson & Whitfield 2011), and this is highly compatible with density surface modelling approaches.

It is also important to highlight that displacement studies to date have tended to focus on comparing numbers or densities of birds pre-construction and post-construction which, in general, do not distinguish between birds in flight and birds on the water (the former group being more likely to show displacement). Despite ship-based data collection methods being distinct for birds on the water and birds in flight, counts are generally combined and for most studies presented below are not considered separately. Similarly, whilst visual aerial surveys do differentiate between birds on the water and those in flight, estimates are usually collated.

5.1.4.2 Results of studies on displacement/attraction

Of the studies considered, comparisons of pre- and post-construction surveys were carried out in all cases with the exception of Egmond aan Zee (Leopold *et al.* 2011), where it was argued that this was not possible due to considerable annual variation in seabird presence (Appendix 2, section A2.1). Instead analysis of the effect of the windfarm was carried out based on individual surveys (e.g. species monthly counts which were converted into presence/absence data) for which there were sufficient data and the results should therefore be considered with caution (see Table 5.2).

It was only possible to calculate actual values of macro-avoidance for a single study carried out at the Blighbank and Thorntonbank windfarms for which the model co-efficients generated from the Generalised Linear Models were provided (Vanermen *et al.* 2013). Results for Thorntonbank are not considered here, however, as they relate either to the first post-construction phase when only six turbines were operational or during the second phase of construction which was still ongoing at the time of reporting. Other studies have reported evidence for displacement or attraction based on the results of Jacob's selectivity indices (Nysted and Horns Rev - Peterson *et al.* 2006) or density surface maps of the predicted distribution over the different phases of the development (Robin Rigg – Natural Power 2014).

For northern gannet there was strong evidence for displacement effects at Blighbank based on comparisons of pre- and post-construction data. From this study, therefore, it was possible to derive a macro-avoidance rate of 0.84 for northern gannet. Currently the vast majority of monitoring tends not to present a seasonal breakdown of displacement (macro-avoidance) values and this report does not differ in that respect. However, there is notable variation in the seasonal use of the windfarm and the surrounding area (termed the BPNS) by northern gannet – mean numbers across the period of 2001-2007 in winter and autumn were 1,799 and 4,990 respectively compared to spring and summer at 737 and 556 respectively (see Table 2 in Vanermen *et al*. 2013). Therefore, as for barrier effects, the extent to which these data are representative of northern gannet during the breeding season is debateable. It is also worth reflecting that further monitoring work has been carried out since the publication of Vanermen *et al*. (2013) and that these results should be considered as being provisional (Nicholas Vanermen *pers. comm*.). Potential corroboration that northern gannets are displaced by windfarms is also provided by results from Egmond aan Zee (Leopold *et al.* 2011) where it was shown that the presence of northern gannets was significantly negatively related to the presence of the windfarm in two of nine monthly post-construction surveys (no other significant effects were reported for the other seven surveys). However the strength of this evidence is relatively weak as the analyses were based on within survey (monthly) comparisons – a comparison of pre- and post-construction data was not considered feasible (see Appendix 2 – A2.1 for further details). The study at Robin Rigg found no response from northern gannet to the windfarm which tend to use the site mainly during the breeding season, though the pre- and post-construction densities were generally rather low e.g. across the entire study within which the windfarm is located, a total of 352 birds were recorded in flight for the entire pre-construction period compared to 397 in the post-construction period (up to and including year 3 - see Table 3.22 in Natural Power 2014). Similarly at Horns Rev, there were never any birds recorded within the windfarm itself either pre- or post-construction (although an increased avoidance was reported for both the 2 km and 4 km buffers post-construction based on Jacob's selectivity indices). More recent work carried out at Alpha Ventus was inconclusive as the overall abundance of northern gannet was very low e.g. a total of nine individuals were seen in the pre-construction period (BSH 2011 and Mendel *et al.* 2014).

Lesser black-backed gull was only considered by three of the studies reported in Table 5.2. There was strong evidence of very high levels of attraction at Blighbank (Vanermen *et al.* 2013) – with relative increases in numbers at the windfarm provisionally estimated in the order of 3.81 (see Appendix 2, section A2.3 for further details). Far weaker evidence to support lesser black-backed gulls being attracted to windfarms was provided from Egmond aan Zee (Leopold *et al.* 2011) where the presence of lesser black-backed gulls was significantly negatively related to the presence of the windfarm in at least one out of 12 possible monthly post-construction surveys. However, despite the results being suggestive of displacement, it was concluded by the authors that, given the strong association shown by lesser black-backed gulls to fishing vessels (based on anecdotal observations during the surveys) attraction to the windfarm was apparently being masked by their strong association with boats which were excluded from the windfarm in the post-construction period. Completely contradictory results were derived for Alpha Ventus where comparison of the distribution of birds pre- and post-construction showed a marked decrease in

densities (based on maps of 1 km^2 cells) and statistically significantly lower abundances were reported for the 0-2 km, 2-6 km and 6-10 km distance classes from the windfarm (BSH 2011 and Mendel *et al.* 2014). There is also no consistent pattern in the studies summarised in Table 5.2 for either displacement or attraction being shown by herring gulls, great black-backed gulls and black-legged kittiwake (Table 5.2).

Furness *et al.* (2013) developed a scoring system to quantify the vulnerability of marine bird population to offshore windfarms with respect to collision and disturbance/displacement. Northern gannet, lesser black-backed gull, herring gull, great black-backed gull, and black-legged kittiwake all scored very highly with respect to collision risk (within the top seven of all the species considered) and this was largely a result of time spent flying at rotor height (other parameters considered included flight agility, % of time flying, night flight and an overall conservation score). In contrast, with respect to displacement, all five species scored very low (species concern index values were no more than 6 compared to the highest value of 32). This was a result of the species being little affected by the disturbance effects associated with ships/helicopters and not being particularly constrained by foraging habitat (the same overall conservation score used for collision risk was also used with respect to displacement). Given this, it is therefore unsurprising that the majority of priority gull species appeared to show no consistent pattern for displacement.

5.1.5 Evidence for an overall macro-response rate

In terms of assessing whether changes in numbers (e.g. from the pre-construction to post-construction periods) are statistically significant, this has only been possible for displacement /attraction studies and not for barrier studies. The notable exception to this is the work carried out at Nysted windfarm (Desholm & Kahlert 2005; Peterson *et al.* 2006) where it was possible to record the number of flight paths that changed their direction by comparison of the pre-and post- construction periods.

There are also considerable issues in how data are collected in terms of differentiating between barrier and displacement effects. Migratory species, which have a distinct passage period during spring and/or autumn and do not occur in the vicinity of the windfarm outwith these periods (e.g. geese spp and passerine spp), are likely to experience solely barrier effects. In contrast, species which are resident in the vicinity of the windfarm, may be subject to a combination of barrier effects or displacement /attraction effects (e.g. the vast majority of seabird spp, at least in the breeding season). This is certainly the case for all of the five priority species being considered here and to date, there has not been a single study which can be considered as exclusive evidence for either barrier or displacement effects.

It is also worth flagging up that the extent to which impacts of the windfarm actually affects bird populations is likely to be site specific. Therefore it would be reasonable to expect that the barrier effects for migrating birds are far more likely to be pronounced when offshore windfarms are located on major flyways. Similarly, an offshore windfarm that is located within the foraging ranges of breeding seabirds is more likely to be an issue in terms of barrier and displacement/attraction effects compared to one that is not (although the latter scenario is unlikely). Another consideration which has been picked up by this review occurs when the windfarm

has relatively low numbers of certain species using the site pre-construction. This may give a misleading impression as to the extent of any changes pre- and post-construction. Whilst an increase or decrease in numbers between these periods may give the impression of a significant effect, the power to detect such a change is extremely low, and, as a consequence, we cannot have much confidence in these results.

Another important caveat related to all studies of barrier and displacement/attraction effects, is that there has been very little attention given to teasing out potential variation over the annual cycle and only a single value of relative change between pre- and post-construction is presented. Yet there may be significant time and energy constraints imposed by the breeding season when birds have to return repeatedly to the nest whereas at other times of year they can move more freely (Stephens *et al.* 1986). In addition to this shift due to the onset of the breeding season, notable changes in foraging behaviour within the breeding season have also been extensively documented in seabirds (e.g. black-legged kittiwake trip duration typically decreases from incubation to the chick rearing period due to the need to feed the young frequently Hamer *et al*. 1993). Hence, the response of foraging and commuting birds to the presence of a windfarm may vary according to the stage of their life cycle e.g. birds which are limited in terms of time or energy may be willing to take more risks by entering the windfarm when otherwise they would simply avoid the area. While, due to the absence of evidence, any such seasonal variation in birds' responses to the impacts of windfarms is hypothetical, when utilising derived macro avoidance rates, the extent to which these values are considered representative for all times of year should be given careful consideration, particularly if they contribute to the collision risk modelling. There may also be further scope in the future for investigating variation in macro-responses between the breeding and non-breeding seasons (although investigating within the breeding season differences may be more problematic).

Among the priority species considered by this review, there is limited evidence, however, to suggest that northern gannet may show a tendency towards a negative macro–response. The study of barrier effects at Egmond aan Zee, Krijgsveld *et al.* (2011) suggests a macro-avoidance rate of 0.64, while the study of displacement at Blighbank, Vanermen *et al.* (2013) suggests a macro-avoidance rate of 0.84. At this stage, the lower and therefore the most conservative of these values is assumed to be a reasonable macro-response rate.

In contrast, there is a lack of species- or even species group-specific evidence for barrier effects relating to gulls. With respect to displacement/attraction, the evidence is equivocal, with some studies suggesting evidence for attraction, others evidence for displacement, and others no significant response. For gulls, the balance of evidence thus suggests a macro-response of 0 (i.e. no attraction to or avoidance of the windfarm).

Table 5.2 Summary of key studies of displacement and attraction studies, the stages of development at which data were collected, main methods used, parameters collected, species reported and responses. Grey indicates species which were not covered by that particular study. For further information see Appendix 2 for site accounts.

Windfarm	Study	Survey/s used	Modelling approach	Species	Response (values are given in parentheses where available)		
					Displacement	Attraction	None
Blighbank	Vanermen *et a*l. (2013)	Boat	Generalised linear models with a negative binomial distribution with count data as the response	Northern gannet	✓(0.84)[1]		
				Lesser black-backed gull		✓(-3.81)[6]	
				Herring gull		✓(-51.98)[6]	
				Great black-backed gull			✓
				Black-legged kittiwake			✓
Egmond aan Zee	Leopold *et al.* (2011)	Boat	Presence/absence modelling of individual monthly surveys (Generalised Additive Modelling)[2]	Northern gannet (10/2) [2]	✓		
				Lesser black-backed gull (12/1) [2]	✓		
				Herring gull (14/3) [2]	✓		
				Great black-backed gull (17/6) [2]	✓[4]	✓[5]	
				Black-legged kittiwake (5/1) [2]		✓	
Horns Rev	Peterson *et al.* (2006)	Aerial	Comparison of Jacob's Selectivity Indices Encounter rates per survey km (student's t-test)	Northern gannet			✓
				Lesser black-backed gull			
				Herring gull			✓
				Great black-backed gull			
				Black-legged kittiwake			

Windfarm	Study	Survey/s used	Modelling approach	Species	Response (values are given in parentheses where available)		
					Displacement	Attraction	None
Nysted	Peterson *et al.* (2006)	Aerial	Comparison of Jacob's Selectivity Indices Encounter rates per survey km (student's t-test)	Northern gannet			
				Lesser black-backed gull			
				Herring gull			✓
				Great black-backed gull			
				Black-legged kittiwake			
Alpha Ventus	BSH (2011) and Mendel *et al.* (2014)	Boat and aerial	Comparison of changes distribution patterns (1 km[2)] Generalised Linear Mixed Models with a Poisson error[7]	Northern gannet			✓
				Lesser black-backed gull	✓		
				Herring gull			
				Great black-backed gull			
				Black-legged kittiwake	✓		
Robin Rigg	Natural Power (2014)	Boat	Generalised Additive mixed effects mixture modelling within a Bayesian framework	Northern gannet -			✓
				Lesser black-backed gull			
				Herring gull	✓		
				Great black-backed gull	✓[3]		
				Black-legged kittiwake			✓

[1] See Appendix 2 for calculations
[2] The total numbers of post-construction monthly surveys for which there were sufficient data for modelling / the number of which the results were significant
[3] Between pre-construction and construction only
[4] Four surveys
[5] Two surveys.
[6] negative values for attraction;
[7] lesser black-backed gull only.

5.2 Review of Published Evidence for Meso-Response Rates of Marine Birds

5.2.1 Studies of meso-avoidance

Within a windfarm, birds may respond to the presence of a turbine either by altering the altitude at which they fly, termed a vertical meso-response, or by altering the flight path they take, termed a horizontal meso-response. This is distinct from micro-avoidance, which occurs as a 'last-second' reaction to avoid collision, as meso-responses may take place at some distance from the turbines but still within the windfarm site.

On entering a windfarm, birds may alter their horizontal flight path so that they fly around, or between, turbines, thereby lowering their risk of collision. Alternatively, they may make no response, or even be attracted to a turbine, as a potential roost or perch. In such circumstances, the risk of collision for each individual bird would remain the same, or increase. Such behaviours can be assessed by examining flight paths within the windfarm and considering whether these show a change in direction as they approach the turbines or considering whether birds approach turbines in the proportions that may be expected if they were randomly distributed within the windfarm.

Birds may also alter their flight heights in response to turbines. They may alter their flight heights so that they fly under, or above, the turbines in order to avoid collision. Alternatively, they may make no alteration to their flight height in response to encountering a turbine, meaning either they make a horizontal alteration to their flight path to avoid collision, or their risk of collision remains the same.

Avian flight heights are commonly assessed during surveys of onshore or offshore windfarms. However, concluding that a certain proportion of birds might fly below the rotor swept area of a turbine does not necessarily imply avoidance behaviour as seabirds commonly fly at low altitudes in the absence of turbines (Cook *et al.* 2012, Johnston *et al.* 2014a). In order to assess the scale of any vertical responses to turbines it is necessary to compare the proportion of birds flying at rotor height within the windfarm to data collected either prior to the windfarm construction, or to the proportion of birds flying at rotor height in control areas adjacent to the windfarm.

5.2.2 Horizontal meso-response conclusions

Evidence for the extent and direction of horizontal meso-responses to turbines is extremely limited (see Appendix 3). We identified two studies with relevant information from the onshore environment (Everaert 2008 and Janoska 2012) and two studies from the offshore environment (Skov *et al.* 2012 and Krijgsveld *et al.* 2011). At De Put in Belgium, no evidence of a response was recorded amongst either black-headed or common gulls (Everaert 2008). Similarly, the data presented for Horns Rev I and II in Denmark only support a meso-response for large gulls, with none of the 402 flight paths recorded passing within less than 50 m of a turbine (Skov *et al.* 2012). The data presented do not make it possible to determine whether meso-responses occur within northern gannet, common scoter or terns although, on average these species passed turbines at a greater distance than large gulls. Data from two terrestrial sites in Hungary also suggest a strong, meso-response for large

gulls, with only 2.5% of birds flying within 75 m of a turbine (Janoska 2012). However, confidence in these data is extremely limited given the lack of detail available about the methodology of this survey. The strongest evidence for a meso-response rate from an offshore windfarm comes from Egmond aan Zee in the Netherlands. Here, the number of birds recorded by radar within 50 m of a turbine was 66% of those recorded elsewhere within the windfarm (Krijgsveld *et al.* 2011), reflecting a meso-response rate of 0.34, considerably lower than the meso-response rate reported in the Hungarian study.

However, it should be noted that measurements of the proportions of birds passing within a set distance of a turbine may not be an accurate reflection of the true meso-response rate. To estimate species' meso-response rates it is necessary to consider whether the proportion or density of birds in areas close to turbines is higher or lower than would be expected within the windfarm as a whole. This could, potentially, be assessed either through visual observations during surveys of the area, or with the use of remote tracking technologies, such as radar. At present, however, such data are too limited to reliably quantify the horizontal meso-response rates of birds within a windfarm.

5.2.3 Vertical meso-response rates conclusions

All evidence for vertical meso-response rates which we identified originated from the offshore environment (Table 5.3). The quality of evidence presented by each of these studies varies considerably (see Appendix 4). For example, at Blyth, there was a reported increase in the proportion of birds flying at altitudes of more than 9.1 m above mean sea-level between pre- and post-construction (Rothery *et al.* 2009). However, as the rotor sweep of turbines at this site is between 26.4 and 92.4 m above mean sea level, it is unclear as to whether, despite this apparent increase in flight height post-construction, there was a significant increase in the proportion of birds flying at rotor height. Similarly, data from Nysted and Horns Rev were collected by radar and cover all birds flying below 200 m above mean sea level and are also, therefore, likely to incorporate a significant number of birds flying outside the rotor sweeps at these sites (Blew *et al.* 2008). Due to the significant proportion of birds in both of these studies that are likely to fly outside the turbine rotor sweeps, it is not possible to obtain useful information about the level of vertical meso-responses from either. In addition, at Robin Rigg (Natural Power 2013) concerns have been raised about the power of the available data to detect changes in species' flight heights, and about the methodology used to collect data on species in flight which may have led to the double-counting of individuals. For these reasons, data from these sites are not considered further in this section.

Of the remaining sites, estimates of vertical meso-avoidance rates can be obtained from Barrow (Barrow Offshore Wind Farm Ltd) and Gunfleet Sands (NIRAS 2011, GoBe Consultants Ltd. 2012) by comparing the proportion of birds flying in different height bands pre- and post-construction, and at Egmond aan Zee (Krijgsveld *et al.* 2011) by comparing the proportion of birds at different heights inside and outside the windfarm. Of the species or groups for which data were available, only divers showed a consistent vertical response to turbines, in the form of a reduced proportion of birds at rotor height. Other species appear to show a full range of responses covering a strong vertical avoidance to a strong vertical attraction. For

example, the proportion of northern gannet assessed to be flying at heights placing them at risk of collision increased by 59% between pre- and post-construction at Barrow, but the proportion at risk height at Egmond aan Zee within the windfarm was 49% lower than the proportion outside the windfarm. A similarly mixed picture is evident for each of the remaining four priority species. The differences in the methodologies used by each study and the inconsistency in the different results mean it is not possible to draw conclusions about the magnitude or direction of any vertical meso-response to turbines.

Table 5.3 Vertical meso-avoidance rates obtained from reviewed studies – see Appendix 4 for the origin/derivation of these figures. Values of 0 indicate no response, values <0 indicate an attraction response, values >0 indicate an avoidance response.

	Barrow	**Blyth Summer**	**Blyth Winter**	**Egmond aan Zee**	**Gunfleet Sands 2010/11**	**Gunfleet Sands 2011/12**	**Nysted / Horns Rev**	**Robin Rigg**
	Barrow Offshore Wind Farm Limited.	**Rothery *et al.* (2009)**	**Rothery *et al.* (2009)**	**Krijgsveld *et al.* (2011)**	**Niras (2011), GoBe Consultants Ltd, (2012)**	**Niras (2011), GoBe Consultants Ltd, (2012)**	**Blew *et al.* (2008)**	**Natural Power Consultants (2012)**
Diver spp					1.00	1.00		
Red-throated diver					0.39	0.86		
Northern gannet	-0.59	-27.00	*exp**	0.49				*exp**
Common scoter	-0.24	0.00	0.00					
Common eider		0.00	-2.00					
Great cormorant	*exp**	-1.38	-0.61	-0.38				*exp**
Arctic skua	-1.00							
Black-headed gull	0.56	0.07	-7.00	0.49	-10.70	1.00		
Common gull	*exp**			-0.20	-0.25	0.33		
Little gull				-0.65				
Black-legged kittiwake	-0.41	-0.14	-28.00	0.20	-0.47	0.05		-1.00
Small gulls				-0.26				
Lesser black-backed gull	0.72			-1.00	-0.44	0.00		
Herring gull	0.35	-1.16	-1.67	0.02	-0.02	0.11		-8.00

Great black-backed gull	0.28	-1.38	-3.50	0.17	-0.75	-0.53		-0.67
Large gulls				-0.01				
Gull spp	-0.85			0.45	-1.98	-1.13		
Sandwich tern	0.56	-1.94		0.35				
Common guillemot	-1.00							
Auks spp	-1.00							
Daytime migrants <200							0.17	
Night time migrants <200 m							0.18	

*Increase in the number of birds recorded from 0 in pre-construction surveys, meaning it is not possible to calculate a proportional increase.

5.2.4 Meso-response rates conclusions

Data quantifying meso-response rates to turbines within offshore windfarms are extremely limited and of variable quality. Overall, evidence describing horizontal meso-responses appears to be stronger than the evidence for vertical meso-responses. Data from one onshore (Janoska 2012) and one offshore site (Krijgsveld *et al.* 2011) appear to suggest a moderate, negative horizontal meso-response to turbines. Whilst there was a stronger meso-response rate at the onshore site, a lack of methodological detail made it difficult to understand the reasons for this difference. Furthermore, an additional two studies did not offer evidence of a horizontal meso-avoidance rate (Everaert 2008, Skov *et al.* 2012). As all four studies we identified had limitations at this stage it is not possible to be confident about the magnitude of any horizontal meso-response, particularly at a species specific level. Whilst a greater quantity of data were available describing vertical meso-responses to turbines, the variable nature of these data and limitations associated with each study, mean it is not possible to draw firm conclusions about either the magnitude or direction of any vertical meso-response. Particular concerns included the low power of some of the datasets, and a lack of overlap between the height bands assessed and the rotor-swept areas of the installed turbines.

However, some studies do indicate how meso-responses may vary within-windfarms. Data from Horns Rev suggest that as birds travel further into a windfarm, they respond more strongly to turbines, with a greater number of directional changes in response to the third or fourth turbine rows than to the first or second rows (Petersen *et al.* 2006). Similarly, the operational status of turbines may influence species responses. Again at Horns Rev, common scoter, Arctic skua, herring gull, great black-backed gull, kittiwake, common/Arctic tern and Sandwich tern were all found to be less likely to pass by operational than non-operational turbines. This response is even stronger when considering birds passing between two adjacent

turbines which are both either operational or non-operational (Petersen *et al.* 2006). Similar results have been found at Alpha Ventus and Egmond aan Zee, where concentrations of birds were higher when turbines were non-operational than when they were operational (Krijgsveld *et al.* 2011, Mendel *et al.* 2014).

5.3 Review of Published Evidence for Micro-Avoidance Rates of Marine Birds

5.3.1 Studies of micro-avoidance

We consider micro-avoidance to be the 'last-second' action taken to avoid collision with a turbine. In practice, this can be difficult to measure given the effort required to generate meaningful data. Several strategies have been employed to collect such data including: direct observations of bird interactions with turbines, using radar to track birds as they approach turbines and fitting cameras to turbines to record interactions. Interpretation of these data may be challenging and necessitate subjective judgements in relation to whether a bird is at risk of collision and what behavioural responses reflect a reaction.

5.3.2 Micro-avoidance conclusions

Data describing the 'last-second' response of birds to turbines have been collected from 16 individual turbines, of which 14 were offshore and two were onshore, across four sites for in excess of 3,000 hours (Desholm 2005, RPS 2011, Krijgsveld *et al.* 2011, Wild Frontier Ecology 2013; see Appendix 5). Despite this effort, very few birds have been recorded passing close enough to turbine rotors to necessitate micro-avoidance action. In total, 45 birds (excluding those recorded at Nysted in Denmark, which were not recorded passing within less than 20 m of turbines, Desholm 2005) have been recorded passing close enough to turbines to necessitate some form of avoidance action, and at least 42 of these have been recorded as taking some form of avoidance action (RPS 2011, Krijgsveld *et al.* 2011, Wild Frontier Ecology 2013). The remaining three birds were tracked at Egmond aan Zee in the Netherlands, using radar and it is unclear whether or not these may also have taken some form of avoidance action, although they were not recorded as colliding with the turbines (Krijgsveld *et al.* 2011).

These data suggest that last-second action to avoid collision is an extremely rare event. This is not because birds do not respond to turbines, but because most avoidance action takes place at distances from the turbines beyond which the methodologies in the studies above could record (i.e. at the meso- and/or macro scales). Whilst only limited data are available describing micro-avoidance rates, the 45 flights considered in the studies described above suggest that a high proportion of birds, >0.93 based on the data described above, may take last second action to avoid collision.

5.4 Review of Published Evidence for Within-Windfarm Avoidance Rates of Marine Birds

5.4.1 Background

In addition to monitoring behavioural avoidance of birds at windfarms, as described in the micro-avoidance and meso-response sections above, a key part of the post-construction monitoring programmes at onshore windfarms is recording the incidence of collisions between birds and turbines. This is typically achieved through organised searches at regular intervals around turbine bases (e.g. Winkelman 1992, Thelander *et al.* 2003, Everaert 2008). Corrections are then applied to account for factors including searcher efficiency and the removal of corpses by scavengers (e.g. Winkelman 1992). These records are often presented as a collision rate per turbine per year (e.g. Winkelman 1992, Musters *et al.* 1996, Brown & Hamilton 2004, 2006, Grunkorn *et al.* 2009). Whilst such values may provide a useful comparison of collision risk between individual turbines within a windfarm, or between windfarms in general, they do not, by themselves provide useful information about the behavioural responses of birds to the presence of turbines.

In order to use collision rates to derive meaningful information about the behavioural responses of birds to the turbines, it is necessary to combine them with an estimate of the rate at which birds pass through the windfarm. Estimates of the rate at which birds pass through the windfarm can be derived by converting the total number of birds observed over a known period of time into an hourly, or daily rate. These flux rates can then be multiplied by the total length of the study period, taking care to correct for factors such as variable day length, to estimate the total number of birds passing through the windfarm during the period in question – for example, the months over which searches were made for collision victims. It may also be necessary to rescale these estimates, for example if only a proportion of the windfarm was covered during surveys. However, as movement data refer to the windfarm as a whole, it is not possible to separate the meso and micro elements of these mortality derived avoidance rates. For this reason, these are collectively referred to as within-windfarm avoidance rates.

5.4.2 Methodology

5.4.2.1 Deriving within-windfarm avoidance rates

We identified 20 sites at which data were available combining an estimate of the collision rate with an estimate of the rate of flux through the windfarm that made it possible to derive within-windfarm avoidance rates (see Appendix 6). Of these, 17 sites were onshore and three were offshore. Using the methodology set out in Band (2007) it is possible to calculate the number of birds expected to collide with turbines at each of these sites if no avoidance action is taken.

The first step of this process is to estimate the total number of birds likely to have passed through the windfarm during the period in which collisions were recorded. As surveys are not, typically, carried out continuously over the study period, the number of birds recorded must be converted to an hourly rate. The total number of birds passing through the windfarm is then estimated by multiplying the hourly rate by the

total number of hours over the study period as a whole, with a correction applied to account for the nocturnal activity level of the species concerned, based on the figures reported in Garthe and Hüppop (2004). However, it should be noted that the difficulty of collecting data on nocturnal activity in seabirds means it is unclear how accurate these figures are.

The next step is to use this value to estimate the total number of birds likely to pass through the turbine rotor sweeps. The total number of birds flying through the windfarm is multiplied by the proportion estimated to fly at rotor height, based on the original survey data. This value is then converted to the number of flying birds per m^2 and multiplied by the total area occupied by the turbine rotors.

A significant proportion of the birds passing through the turbine rotors are likely to do so without colliding (Band 2007). Therefore, a correction, the Probability of Collision (P_{coll}), must be applied to the data to account for this. This is calculated based on the turbine specifications, design of the windfarm array and the flight behaviour and morphometrics of the species of interest and based on the methodology set out in Band (2007). Species morphometric and behavioural data used to estimate P_{coll} are given in Table 5.4, whilst turbine details for each site are given in Table 5.5.

Table 5.4 Bird parameters to estimate P_{coll} for each windfarm. Speed data taken from Pennycuick (1997) and Alerstam *et al.* (2007), morphometric data from Robinson (2005), where species groups are given, data come from a species likely to be representative of that group as a whole, within the offshore wind context.

Species	Length (m)	Wingspan (m)	Speed (m/s)	Flap/glide
Diver (red-throated diver)	0.61	1.11	14.50	flap
Grebe (great crested grebe)	0.48	0.88	18.65	flap
Northern gannet	0.94	1.72	14.90	glide
Arctic skua	0.44	1.18	13.30	flap
Great cormorant	0.90	1.45	14.50	flap
Common eider	0.60	0.94	18.65	flap
Common scoter	0.49	0.84	18.65	flap
Long-tailed duck	0.58	0.88	18.65	flap
Black-headed gull	0.36	1.05	9.50	flap
Common gull	0.41	1.20	9.50	flap
Black-legged kittiwake	0.39	1.08	13.10	flap
Franklin's gull	0.41	1.20	18.65	flap
Ring-billed gull	0.41	1.20	9.50	flap
Little gull	0.26	0.78	11.50	flap
Lesser black-backed gull	0.58	1.42	13.10	flap
Herring gull	0.60	1.44	12.80	flap
Great black-backed gull	0.71	1.58	13.70	flap
Little tern	0.23	0.52	10.00	flap
Common tern	0.33	0.88	10.00	flap
Sandwich tern	0.38	1.00	10.00	flap
Auk (common guillemot)	0.40	0.67	19.10	flap

Table 5.5 Turbine data used for each site. Figures in red indicate that the parameter was not presented for site in question and had to be estimated from a turbine with a similar design. Row colours indicate confidence assigned to data collected at each site – green indicates highest confidence, where there was both spatial and temporal overlap in the collection of corpse and movement data; yellow indicates moderate confidence where there was temporal overlap in the collection of corpse and movement data, but incomplete spatial overlap, meaning that bird activity had to be extrapolated across the site; red indicates lowest confidence, sites where there was incomplete spatial and temporal overlap in the collection of corpse and movement data, meaning bird activity had to be extrapolated both spatially and temporally; grey indicates studies in which flights through the windfarm were recorded so that collisions could be directly recorded, such studies typically had very little power.

Windfarm	N turbines	Turbine capacity (MW)	Width of survey window (m)	Height of survey window (m)[1]	N blades	Blade width (m)	Rotor diameter (m)	Rotor speed (rpm)	Pitch (degrees)	Hub height (m)
Altamont	685	0.12	7713.624[2]	33.5	3	0.66	19	43.0[25]	10	24.0
Blyth	2	2.00	600[3]	92.4	3	4.40	66	18.0[26]	10	59.4
Blyth Harbour	9	0.30	925[4]	37.5	3	66[32]	25	43.0	10[32]	25.0
Boudwijnkanaal	5/7/14[5]	0.6	1040/1536[6]	79	3	1.10	48	43.0[25]	10	55.0
Bouin	8	2.5	4000[7]	100	3	4.40	80	18.0[26]	10	60.0
Buffalo Ridge	143	0.75	9600[8]	74	3	1.10	48	32.3	10	50.0
De Put	2	0.8	300[9]	100[26]	3	1.10	48	43.0[25]	T10	75.0
Gneizdzewo	19	2.00	3700[10]	120	3	4.40	80	18.0[26]	10	80.0
Greater Gabbard	7	3.6	4000[11]	180[27]	3	4.20	107	15.0[27]	10	77.5
Groettocht	5	1.65	1000[12]	140[28]	3	4.40	66	21.3	10	78.0
Haverigg	8	0.6	920[13]	66	3	1.10	42	43.0[25]	10	45.0
Kauwnee County	31		12247[14]	89	3	1.10	47	43.0[25]	10	65.0
Kessingland	2		800[15]	126	3	2.50	92	15.0	10	80.0

Kleine Pathoweg	7	1.8	1820[16]	120	3	4.40	70	18.0[26]	10	85.0
Oosterbierum	18	0.3	1430[17]	60[18]	3	0.66	30	43.0[25]	10	35.0
Waterkaaptocht	5	1.65	1000[12]	140[28]	3	4.40	66	21.3	10	78.0
Yttre Stengrund	5	2.0	20000[19]	250[29]	3	4.40	72	10.0	10	60.0
Zeebrugge	4/6/23/24/25[20]	0.4[21]	400/720[22]	65/80[23]	3	0.66	34	43.0[25]	10	34.0
Hellrigg	4	2.3	4000[24]	121	3	4.40	82	18.0[25]	10	80.0
Avonmouth	3	2.00	1300[30]	160[31]	3	4.40	83	17.5	10	79.0

[1]Maximum turbine height unless otherwise stated; [2]see Table 2 of Thelander *et al.* 2003, total survey area of 59.5 km^2, width of survey window assumed to be $\overline{59.5} * 1000$; [3]Rothery *et al.* (2009) state 600 m scan area; [4] Lawrence *et al.* (2007) state that observations were carried out between turbines 5-9, turbines separated by 200 m with a rotor diameter of 25 m and arranged in a single line; [5]Collisions were recorded under all 14 turbines in 2002-2006. In 2001, bird activity surveys were carried out around five turbines and avoidance rates derived from collisions around these turbines are also presented. In 2005, bird activity surveys were carried out around seven turbines and avoidance rates derived from collisions around these turbines are also presented; [6]In 2001, only five turbines were present with diameters of 48 m and spacing of 200 m, therefore, the total survey window in 2001 was 1,040 m wide (section 3.3.1 in Everaert *et al.* 2002, Table 27 in Everaert 2008). In 2005, 14 turbines were present, but activity was only monitored around seven of these, therefore in 2005 the total survey window was 1,536 m wide (Table 27 in Everaert 2008). Turbines were all arranged in a single line; [7]Observations carried out along four 1 km linear segments on the edge of the windfarm, see section 5.1 of Dulac (2008); [8] Raptor/large bird surveys carried out through point counts at six locations, each with a radius of 0.8 km, (page 7, Johnson *et al.* 2000); [9]Estimated from Figure 101 in Everaert (2008); [10]Estimated from Google Earth map of windfarm (https://www.google.co.uk/maps/place/Gnie%C5%BCd%C5%BCewo/@54.7467485,18.3525275,3643m/data=!3m1!1e3!4m2!3m1!1s0x46fdb3a54ca46bb1:0x5926557d4b8964d0); [11]Data collected within viewing arc with a radius of 2 km, covering seven turbines (Galloper Offshore Windfarm Environmental Statement, Appendix 4); [12]Data presented as number of birds/km/hr; [13]Table A.3.13 in Galloper Offshore Windfarm Environmental Statement; [14]Abstract of Howe *et al.* (2002) states that 150 km^2 were surveyed, width taken as $(\overline{150}) * 1000$; [15]Birds recorded were those passing within a 200 m radius around each turbine, Wild Frontier Ecology (2013); [16]Table 32 of Everaert (2008), activity monitored around turbines 3-7 which are each separated by 280 m, arranged in a single line and have a diameter of 140 m; [17]Section 2.1 of Winkelman (1992), turbines have a diameter of 30 m and are 250 m apart and arranged in three lines of six turbines; [18]Birds up to 60 m recorded (Tables 12a-d Winkelman 1992); [19]Movements monitored over four 5 km observation lines (Figure 3, Petterson 2005); [20]Based on Everaert (2008) – 23 turbines were operational and searched for corpses in 2001-2003, 25 turbines were operational and searched for corpses in 2004, and 24 turbines were

operational and searched for corpses in 2005-2007. In addition, collision data for the four turbines monitored for gull activity in 2000 and 2001 (Everaert *et al.* 2002) and the seven turbines monitored for tern activity in 2004 and 2005 (Everaert & Stienen 2007, Everaert 2008) are also analysed in this report; [21]While different turbine types have been used at Zeebrugge, the analysis in this report is based on the assumption that they share the characteristics of those on the eastern wall, where the greatest number of collisions are typically recorded (Everaert 2008); [22]Gull activity was monitored along a 400 m section of the eastern wall in 2000 and 2001 (Everaert *et al.* 2002) and tern activity was monitored along a 720 m section of the eastern wall in 2004 and 2005 (Everaert & Stienen 2007, Everaert 2008); [23]In 2000 and 2001, flight height was estimated up to a maximum of 65 m and in 2004 and 2005 flight height was estimated up to a maximum of 80 m; [24]States that standard SNH vantage point methodology with radius of 2 km from a single point used (Percival 2012, 2013);[25]Based on rotational speed of Blyth Harbour turbines;[26] Based on rotational speed of Enercon E-70 2.3 MW turbine;[27]Similar size to Kessingland turbines; [26]Birds up to 100 m recorded, see Table 37 of Everaert (2008); [27]Birds up to 180 m recorded, see section 1.11 of Appendix 4; [28]Radar monitoring of flight heights up to 140 m, see Krijgsveld *et al.* (2011); [29]Flights monitored up to altitude of 250 m, see figure 11 of Petterson (2005); [30]Estimated from Google Earth map of windfarm (https://www.google.co.uk/maps/search/Bristol+Port+Wind+Park/@51.5117476,-2.7031114,1372m/data=!3m1!1e3); [31]Paragraph 2.3 of The Landmark Practice (2013).[32]highlighted grey so red numbering shows up against red background.

The number of birds predicted to collide with the turbines in the absence of any avoidance action can be estimated by multiplying the total number of birds predicted to pass through the rotor sweep of the turbines over the course of the time period in which collision searches were carried out by the probability of those birds colliding with the rotor blades. An avoidance rate can now be derived from these data by dividing the observed collision rate by the predicted collision rate, as in equation 6.

Avoidance rates were derived, as described above, for each species-site combination for which sufficient data were available in the studies identified as part of our literature review. The quality of data presented in each of these reports was highly variable, in particular in the level of spatial and temporal overlap between the periods over which corpses were collected and bird movement data were collected. The feasibility of collecting movement data over the course of the study periods as a whole meant that some extrapolation was inevitable when calculating avoidance rates. However, we sought to minimise this extrapolation and sought to categorise the studies we identified accordingly (Table 5.5).

The first category (green) we identified, which we had greatest confidence in, was that in which activity data were collected at intervals throughout the period in which corpse data were collected, and from around all turbines which were searched for corpses. This meant that no spatial extrapolation was necessary to derive an avoidance rate, and the need for temporal extrapolation was minimised. The second category (yellow) we identified was similar to the first, with the exception that activity data were not collected around all of the turbines which were searched for corpses, for example at Kleine Pathoweg, where bird movements were only monitored around five of the seven turbine where corpse searches were carried out. This meant that spatial extrapolation of movement data was necessary, potentially leading to erroneous conclusions if flights were not to occur evenly throughout the site. The third category (red) also required spatial extrapolation of activity data. In addition, movement data were only collected for a portion of the time in which corpse data were collected, meaning that bird activity had to be extrapolated across seasons or years. Such extrapolation is extremely likely to give a misleading picture of the true level of bird activity at a site over the study period which is likely to vary seasonally, e.g. over breeding or migration periods. The final category (grey) relates to studies in which bird movements through windfarms have been monitored in order to directly observe collisions. Given the relative rarity of birds colliding with turbines, these studies typically have low power to detect a collision.

We consider how each of these categories influences the avoidance rates that are derived. We also consider the influence of other factors, such as turbine size, on avoidance rates in order to assess whether it is appropriate to apply avoidance rates from some of the smaller onshore turbines to the much larger turbines used in the offshore environment.

The estimation of predicted collisions requires assumptions to be made regarding the proportion of birds flying at collision risk height and their flight height distributions. Consequently, we derive avoidance rates appropriate for use with each of the three model options presented in the Band offshore collision risk model spreadsheet (Band 2012):

i. Option 1, where site specific flight height data are used to estimate the proportion of birds flying at collision risk height;
ii. Option 2, where modelled data are used to estimate the proportion of birds flying at collision risk height, based on the distributions presented in Johnston *et al.* (2014a) and the exact rotor dimensions presented in each report;
iii. Option 3, where modelled flight height distributions are used to account for collision risk not being distributed evenly within a turbine's rotor swept area.

It should be noted that different values would be expected for Band model options 1 and 2 because option 2 uses generic distributions from compiled data sources, which may not be directly comparable to data collected from some of the sites included in this review. In some cases, option 2 may also use a better defined risk window, as it reflects the actual turbine dimensions rather than a pre-defined window set during pre-construction surveys.

5.4.2.2

The aim of this review was to derive representative within-windfarm avoidance rates that can be used to inform a total avoidance rate for use in collision risk modelling for each of the priority species. Whilst the above methodology can give us a range of different values for marine birds in general, and some of the priority species in particular, combining them to get a single, representative figure is far from straightforward. This is further complicated as several studies report no collisions, suggesting an avoidance rate of 1 over the study period. However, were the study periods of these studies to be extended indefinitely, it is likely that the avoidance rate would drop to below 1 as some individuals will always fail to take action to avoid collision, given sufficient time and bird flux within the site. Whilst one approach would be to discard studies in which no collisions were recorded, this would be inappropriate as it would risk negatively biasing our dataset and, potentially, result in a within-windfarm avoidance rate which is overly precautionary.

We identified five methodologies – ratio estimators, meta-analysis, proportional hazard models and mark-recapture models, events-trials models and Poisson regression – that could potentially be used to combine collision records and flux rates across sites in order to derive representative avoidance rates (Table 5.6). We then considered the limitations and assumptions associated with each technique, before determining which was likely to be the most effective approach.

Meta-analysis is most appropriate when estimates of variance around effect sizes are available, which was not the case in this instance. The data available from the studies we reviewed fail basic assumptions about perfect detectability required for proportional hazard models. Similarly, as individual birds are recorded only upon their deaths, and not on their entry to the population, mark-recovery models were not appropriate. Collisions between birds and turbines are rare events. As event-trials models are most effective when the probability of an event is moderate, this methodology is also likely to be ineffective. Poisson regression models may be an effective approach. However, such an approach would require time to develop and test using simulated data. It may also be ineffective without access to raw survey data from each site. Whilst this approach may provide a useful framework for future studies it was not considered feasible within the framework of the current project.

Having considered each of the different approaches, we concluded that ratio estimators would be the most appropriate approach to combining the avoidance rate data. Given the limitations of the data, we felt that any of the more complex modelling approaches may result in undue confidence being assigned to the derived values. In the absence of raw data, we feel that any more involved modelling approach is likely to be less than robust and that, in this instance, a simpler approach, such as ratio estimators, is most appropriate.

Ratio estimators divide the total number of collisions across all sites by the total number of collisions predicted in the absence of avoidance behaviour across all sites (equation 9). By dividing the total number of collisions by the predicted collision rate, sites with greater levels of bird activity are given greater weighting than sites at which bird activity is relatively low. Arguably, this approach to weighting is more appropriate than weighting flux rate alone, as it accounts for the fact that a higher flux rate may not necessarily reflect a greater number of birds at risk of collision. For example, a site may have a relatively high flux rate, but only a relatively small proportion of these birds may be at a height which places them at risk of collision. Using equation 9, we derive representative avoidance rates for all species and groups for which sufficient data were available.

$$Within\ Windfarm\ Avoidance = 1 - \frac{Observed\ collisions}{Pcoll \times Flux\ Rate} \quad \text{(eq. 9)}$$

As data come from multiple sites, there is likely to be a degree of uncertainty associated with avoidance rates derived in this manner. The importance of incorporating uncertainty in the Environmental Impact Assessment process is receiving increasing recognition (Masden *et al.* 2014). The variance associated with the avoidance rates derived using ratio estimators can be calculated using the delta method (Powell 2007). The square root of this value will give an estimate of the standard deviation around the avoidance rates derived using ratio estimators (Batschelet 1976). It is important to note that this value will reflect variability between sites, as opposed to uncertainty in the input parameters. At present, many of the input parameters for the Band model are only available as single values (e.g. mean rotor speed), until a realistic range of values is available for the key parameters, quantifying uncertainty from these sources will be challenging.

As we are looking for representative values for the within-windfarm avoidance rates, it is important to ensure that the values we are deriving are not unduly influenced by a single data point (each data point reflecting a single site-year-species combination), or set of data points. For this reason we investigate how different factors may influence the final avoidance rates we derive. As a first step, we explore how much influence (leverage) each data point has on the final, representative avoidance rates. We identify sites which have a high leverage and determine whether there are any common factors linking them, for example, an unusually high or low flux rate or the presence of small turbines.

We then consider how bird flux and turbine size may influence the final derived avoidance rates using a stepwise approach. These analyses are not an essential part of deriving our final avoidance rates, instead, they help us to understand how

reliant our values are on the inclusion of all of our data points. Ideally, as we drop data points from our calculations, the avoidance rates derived should remain fairly constant. In the first analysis, we drop sites based on their estimated flux rates. This helps to demonstrate whether our final avoidance rate is dependent on the inclusion of data from a handful of sites with high levels of bird activity. In our second analysis, we drop sites based on maximum turbine height, to identify whether sites with smaller turbines, less typical of the offshore environment are unduly influencing the values we derive. A more detailed analysis of the sensitivity of our derived values is carried out in section 6 (below).

Table 5.6 Methodologies considered for synthesising avoidance rates across multiple data sources.

Method	Description	Used
Ratio estimators	Ratio estimators provide a relatively simple approach that compares the mean of the number of collisions to the mean of the number of birds at risk of collision (Cochran 1977). The approach does this by combining data across sites prior to any calculation and, therefore, accounting for the differing levels of bird activity at each site. As the number of birds at risk of collision is proportional to the bird flux at a site, this approach effectively weights sites by the level of bird activity recorded. Depending on the data available, such calculations can be undertaken on a species, group or global basis. They have the advantage of offering a single, easily interpretable output. This approach has previously been used to derive avoidance rates for geese from multiple data sources (Pendlebury 2006).	✓
Meta-analysis	Meta-analysis provides a way of combining studies, which may have different uncertainties attached to them, to determine the size and statistical significance of a given effect. The units of meta-analysis are the independent results of studies, rather than the responses of individual subjects (Arnqvist & Wooster 1995), with a strong recommendation from statisticians that they should use weighted combination of effect sizes (Stewart 2010). Meta-analyses are most appropriate when studies present estimates of variance around the effect sizes (Gurevitch & Hedges 1999, Stewart 2010), which were not available from the studies we have reviewed.	×
Proportional hazard models / mark-recovery models	We considered the possibility of using time to event style models such as proportional hazard or mark-recovery models. In the case of proportional hazard models, the data fail basic assumptions about perfect detectability necessary for such analyses. As each individual bird is recorded only on its death (and not on entry to the population, i.e. when it enters the turbine space), it was not possible to use mark-recovery type models to produce synthesised ARs from the various studies.	×
Events-	Events-trials models involve combining the number of events	×

trials models	(in this case, collisions) with the number of trials (in this case, birds passing through the turbines) within a binomial generalised linear model (GLM). However, collisions are rare events and binomial GLMs work best when the probability of an event is moderate (typically in the region of 0.2-0.8). We therefore feel such a methodology is inappropriate in this instance.	
Poisson regression	As collisions are rare events the mean across sites is likely to be low and may be expected to follow a Poisson distribution. We could use bird flux as an offset in such a model to account for the different abundance of birds at each site and incorporate a weighting factor to account for survey effort. Zero-inflation is also likely to be an issue (i.e. many sites record no collisions). Whilst this approach may be possible and provide a useful framework for future analyses, it would require some time to develop and test using simulated data and was thus outside of the scope of this project. It should also be noted that we are uncertain about how effective such an approach would be without access to the raw survey data from each site.	×

5.4.3 Results

5.4.3.1 Derived within-windfarm avoidance rates

Data combining collision rates and passage rates through windfarms were obtained from 20 sites – see Appendix 6 for details of sites and species, and Appendix 7 for full results. However, based on the available data, it was only possible to derive within-windfarm avoidance rates for eider, gulls and terns. Whilst other species had been recorded within the windfarms, these were often present in relatively low numbers, and only a single collision event, involving a flock of four eider, was observed during visual observations of turbines. The range of species reflects the onshore or coastal locations of the study sites, and it should be noted that, among the priority species being considered in this review, no estimates could be derived for northern gannet.

The range of responses estimated from the available data runs from an apparent strong attraction whereby the proportion of birds within the rotor-swept area increases by >1000% in some cases, to strong avoidance, where close to 100% of birds avoid the rotor-swept area.

Multiple years' data were collected from several onshore sites including Avonmouth and Hellrigg in the UK, Boudwijnkanaal and Zeebrugge in Belgium, Gneizdzewzo in Poland, and an offshore site at Yttre Stengrund in Sweden. Multiple years' collision data were also available from Kleine Pathoweg in Belgium. However, bird activity data were not collected concurrently with data on collision rates for this site, meaning the results cannot be used for the purposes of this review. Whilst we have been able to derive a within-windfarm avoidance rate in both study years for this site, this approach is flawed as it involves extrapolating from one year's activity data to the

next. The same is true for some of the data collected for both Zeebrugge and Boudwijnkanaal. Whilst multiple years' data were also collected from sites such as Altamont Pass and Buffalo Ridge in the U.S.A. and Blyth Harbour in the UK, these data were summarised across years so it was not possible to examine year to year variation in derived avoidance rates and the resultant avoidance rates should be treated with a high degree of scepticism. Of the sites where it may be possible to compare values between years, only Hellrigg, Gneizdzewo, Boudwijnkanaal, Yttre Stengrund and Zeebrugge provide data that allow this.

We present within-windfarm avoidance rates from all sites for illustrative purposes only (Appendix 7). For the purposes of deriving representative values, we only use what we consider to be the highest quality data (green rows in Appendix 7) where there is both spatial and temporal overlap between the collection of corpses and the collection of bird activity data. Unless otherwise stated the within-windfarm avoidance rates presented in the text from this point refer to those derived using option 1 of the Band model, but these are applicable to option 2.

In the five years for which data were available for Gneizdzewo, only a single collision involving a gull species was recorded (Appendix 7). Similarly, in the years for which data are available from Hellrigg and Yttre Stengrund, collisions were only recorded in a single year at each site. At Boudwijnkanaal, the within-windfarm avoidance rate for herring/lesser black-backed gulls declined from 0.9903 in October 2001 to 0.9556 in October 2005. At Zeebrugge, it was possible to compare within-windfarm avoidance rates for herring and lesser black-backed gulls both between seasons and years. For herring gulls within-windfarm avoidance rates declined from 0.9861 in the 2000 breeding season to 0.9722 in the 2001 breeding season. For lesser black-backed gulls the equivalent figures were 1 in 2000 and 0.9706 in 2001. In 2001, activity data at Zeebrugge were collected in both the breeding season and autumn. The within-windfarm avoidance rates showed an increase for both species during the autumn, to 0.9976 in the case of herring gulls and 0.9990 in the case of lesser black-backed gulls. However, given the limited data available to explore these patterns, more data are required to make firm conclusions about aspects such as seasonal variation in avoidance rates.

Everaert (2014) presents within-windfarm avoidance rates for gulls derived from the same datasets for Zeebrugge, Boudwijnkanaal, Kleine Pathoweg and De Put, using the basic Band collision risk model. The results differ from those we present. The reason for this is likely to be that Everaert (2014) extrapolate bird activity data to cover broader spatial and temporal scales, whilst we focus only on the turbines and months in which bird activity data were specifically collected. The author highlights this extrapolation as a reason why his results should be treated with caution in his discussion of the results. For our purposes, we felt that focussing on the period when activity data were collected when deriving within-windfarm avoidance rates was more consistent with our approach at other sites. A similar issue has been raised in the past in relation to Sandwich tern within-windfarm avoidance rates derived from collision data at Zeebrugge, where rates derived from the same dataset have varied from 0.9664-0.9955 (see NE/JNCC note on subject). This highlights the importance of transparency in the calculations used to derive within-windfarm avoidance rates, enabling people to understand why differences may have arisen and come to an

informed position about which values are likely to be most applicable to the situation at hand.

Our analyses of the tern data from Zeebrugge suggest that within-windfarm avoidance rates are likely to be towards the high point of this range. Using only collisions reported in June and around the seven turbines from which activity data were collected, we estimated a within-windfarm avoidance rate of 0.9944 for common tern in 2004 and 0.9948 in 2005. For Sandwich tern, we estimated within-windfarm avoidance rates of 0.9980 in 2004 and 0.9989 in 2005. No collisions involving little terns were recorded around these turbines in either year. These data suggest that tern within-windfarm avoidance rates are very high, and may be consistent year on year.

In addition to estimating collision rates from fatality searches, at four sites – Blyth Offshore Windfarm, Greater Gabbard Offshore Windfarm, Haverigg Windfarm and the Yttre Stengrund and Utgrunden Offshore Windfarms in Sweden – bird activity has been monitored with a view to directly observing collisions. In total, 646 hours of observations have been collected in this manner across the four sites and five windfarms. These surveys documented 3,167,238 bird movements within-windfarms, including at least 5,319 involving gulls. Despite this, these had relatively low power to detect a collision. In the absence of avoidance action, across these sites only 63 collisions would have been expected based on the basic Band model and only 45 based on the extended Band model (Appendix 7). In relation to the priority species covered by this report, 17 of the collisions predicted using the basic Band model and 13 predicted using the extended Band model would have involved gulls. A single collision involving a gull would reflect an avoidance rate of less than 0.95 for both the basic and extended Band model. Such an avoidance rate would be extremely conservative, and it is therefore, unsurprising that no collisions were recorded during visual observations. Indeed, over the course of these studies, only a single collision event, involving four common eider at a single turbine at Yttre Stengrund Offshore Windfarm was observed, reflecting a within-windfarm avoidance rate of 0.1861 for common eider in autumn 2003 or 0.9024 across all seasons and years. Consequently these studies do not provide strong evidence for the behavioural response of our five priority species to turbines.

Deriving within-windfarm avoidance rates using the different Band model options

Whilst the observed number of collisions remains constant, regardless of the model option used, the predicted number of birds at risk of collision varies. As avoidance rates are derived by dividing observed collisions by predicted collisions (eq. 6), avoidance rates derived using different model options will vary. Collision estimates produced using the different Band model options and option-specific avoidance rates will only be identical if the windfarm in question has the same specifications as used to derive those avoidance rates. However, this will not be the case when these avoidance rates are applied to a novel site as a result of differences in model input parameters (e.g. turbine specifications and site-specific estimates of the proportion of birds at collision risk height).

Within-windfarm avoidance rates derived using option 1 of the Band model are higher than those derived using options 2 and 3. The difference in values derived

using option 1 and option 2 results from the use of site-specific data on the proportion of birds at risk in option 1, and the use of a generic flight height distribution to inform the proportion of birds at risk in option 2 – in other respects these options are mathematically identical. The difference between option 2 and option 3 lies in how the flux rate and probability of collision are applied across the turbines rotor-swept area. Using option 2, an average collision probability is multiplied by an average flux rate. This introduces error when a species' flight height distribution is not uniform. Option 3 accounts for the non-uniform flight height distribution, common to many species (Johnston *et al.* 2014a), by integrating the flux rate and collision probability over the turbines' rotor-swept area.

5.4.3.2 Representative within-windfarm avoidance rates

The within-windfarm avoidance rates data described above, and presented in Appendix 7 are of extremely variable quality. The final, derived within-windfarm avoidance rates are heavily dependent on the accuracy of the estimated flux rates at each site and on the accuracy of collision estimates. As continuous monitoring of bird activity at these sites was not feasible, some degree of extrapolation to estimate the total flux rate will be inevitable. However, it is desirable to keep this extrapolation to a minimum. For this reason, we only combine data from sites at which it was not necessary to make a spatial extrapolation in order to estimate a flux rate, and for which activity data were collected at intervals throughout the period in which collisions were monitored, to minimise the potential for inappropriate temporal extrapolation. The sites meeting these criteria were Avonmouth (Winter 2007/08, 2008/09, 2009/10, 2011/12), Boudwijnkanaal (October 2001 and October 2005), Bouin, De Put, Gneizdzewo (autumn 2007, 2008, 2009, 2010, 2011, 2012), Hellrigg (winter 2011, 2012), Kessingland, Oosterbierum (autumn 1990, spring 1991) and Zeebrugge (June-July 2000, June-July 2001 and September-October 2001). All of these sites were located onshore.

Across these sites, a total of 3,880,794 seabirds, of which the majority (66%) were gulls, were expected to have passed through the windfarms over the periods in which corpse searches were carried out. We determined that sufficient data were available to derive avoidance rates for four species – black-headed gull, common gull, lesser black-backed gull and herring gull – and four species groups – all gulls, large gulls (lesser black-backed gull, herring gull, great black-backed gull, Caspian gull, yellow-legged gull), small gulls (black-headed gull, common gull, little gull) and all terns.

Black-headed gull

A total of 746,668 black-headed gulls were expected to have passed through seven sites – Avonmouth (four studies), Boudwijnkanaal (one study), Bouin (one study), Gneizdzewo (three studies), Hellrigg (two studies), Kessingland (one study) and Zeebrugge (three studies) – over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 1,839 collisions based on option 1 and 582 collisions based on option 2, and 297 based on option 3. However, in total only 38 black-headed gull collisions were recorded across

all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9795 (± 0.0033 SD) using option 1 of the Band model, 0.9351 (± 0.0031 SD) using option 2 of the Band model and 0.8731 (± 0.0056 SD) using option 3 of the Band model.

We investigated the leverage that each study site had on the final within-windfarm avoidance rates derived for black-headed gull. We identified three sites which had high leverage for the within-windfarm avoidance rates derived using options 1 and/or 2 and/or 3 of the Band model (Figure 5.1). Of these, the exclusion of data from Bouin resulted in an increase in the value derived using option 1. This is likely to be due to the presence of the turbines on the edge of a black-headed gull breeding colony. This may have led to a greater number of flights through the rotor-swept area of turbines by adult birds returning to provision chicks and/or newly fledged chicks less experienced at flying. As a result of the relatively high collision rate at this site, including this site in our analysis reduced the overall within-windfarm avoidance rate derived using option 1. The leverage of data from Hellrigg in 2012/13 was of a similar magnitude, but in the opposite direction. Despite having the highest level of black-headed gull activity and a high proportion of birds flying at collision risk height, no collisions were recorded at this site, in this year. As a consequence, excluding these data from our analysis resulted in a reduction in the overall within-windfarm avoidance rate.

In contrast to option 1, the exclusion of data from Bouin led to a substantial decrease in the overall within-windfarm avoidance rates derived using options 2 and 3. The relative importance of data from Bouin is exaggerated using options 2 and 3 of the Band model as modelled flight height distributions suggest that other sites with high levels of bird activity, such as Hellrigg, should have very low proportions of birds flying at collision risk height. As such, the predicted collision rates at these sites are much lower than when using option 1 and they have much less influence when used to derive overall within-windfarm avoidance rates using ratio estimators. In addition, the within-windfarm avoidance rates derived for Bouin using options 2 and 3 are significantly higher than for other sites at which collisions were recorded meaning, given its importance relative to other sites, excluding data from Bouin from the analysis results in a significant decrease in the overall within-windfarm avoidance rate derived. Excluding data from Boudwijnkanaal from the analysis for options 2 and 3 results in an increase in the overall within-windfarm avoidance rate. As with Bouin, a relatively high number of collisions were recorded at this site. However, as observed data suggest a high proportion (69%) of birds flew at collision risk height, this site did not have particularly high leverage for the within-windfarm avoidance rates derived. However, using options 2 and 3, only 4.5% of birds were predicted to fly at collision risk height, meaning the final within-windfarm avoidance rate derived for this site was relatively low.

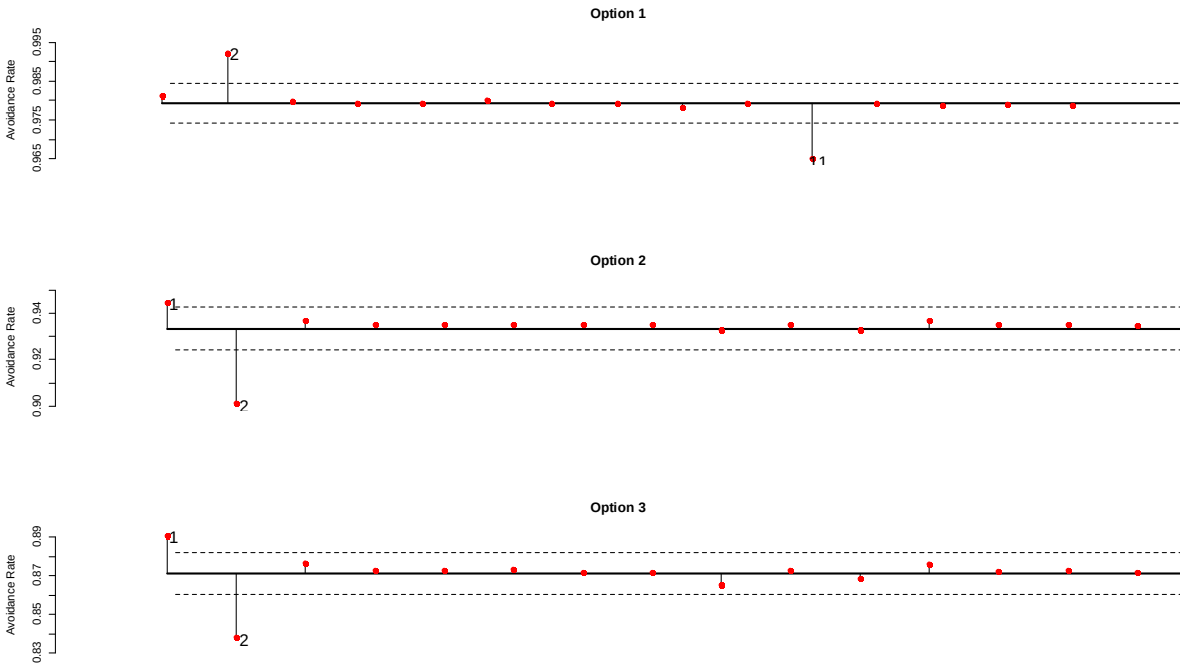


Figure 5.1 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for black-headed gull. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Sites with high leverage are: 1 – Boudwijnkanaal, 2 – Bouin and 11 – Hellrigg in 2012/13.

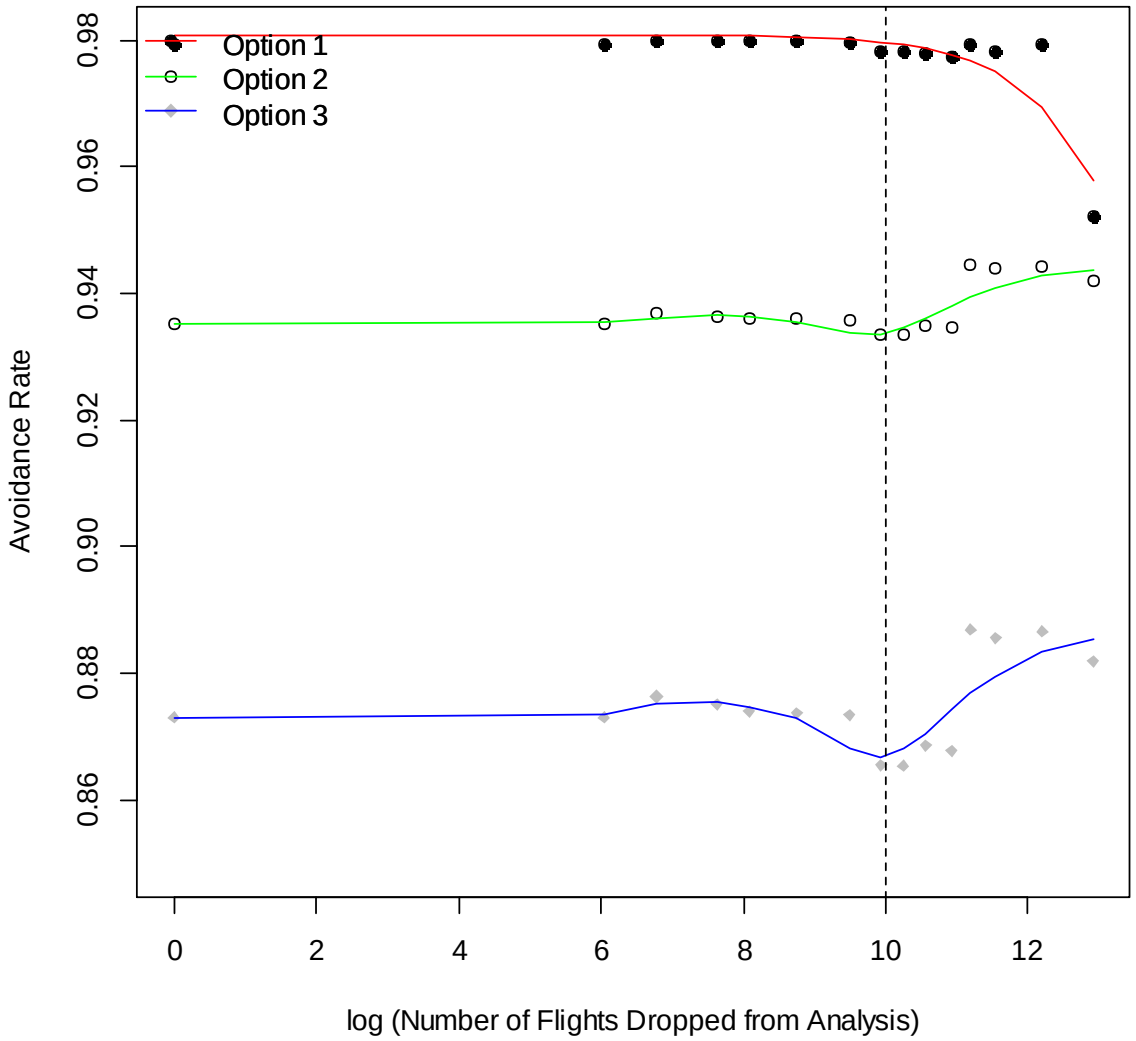


Figure 5.2 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for black-headed gull.

As might be expected, dropping sites from the analysis can influence the final within-windfarm avoidance rates. Only sites at which there is a relatively limited level of flight activity can be dropped from the analysis before the within-windfarm avoidance rates derived become less stable (Figure 5.1). In all three model options, this is noticeable after around 22,000 of the 746,668 flights through the windfarms have been removed (Figure 5.2).

Using option 1 of the Band model, the derived within-windfarm avoidance rate remains relatively stable at around 0.9795 until Bouin is the only site remaining in the analysis at which point it drops to around 0.9370. As discussed previously, this may reflect the fact that Bouin is located on the edge of a black-headed gull breeding colony, resulting in a higher number of collisions than were recorded elsewhere. In contrast, using options 2 and 3,

within-windfarm avoidance rates start to increase after the first 22,000 flights have been dropped. Again, as discussed previously, this is likely to reflect the fact far fewer collisions were predicted at several key sites due to differences in the predicted proportions of birds at collision risk height. As a consequence, as more sites are dropped from the analysis the influence of Bouin, previously identified as having a strong influence on the final derived values for options 2 and 3, becomes stronger.

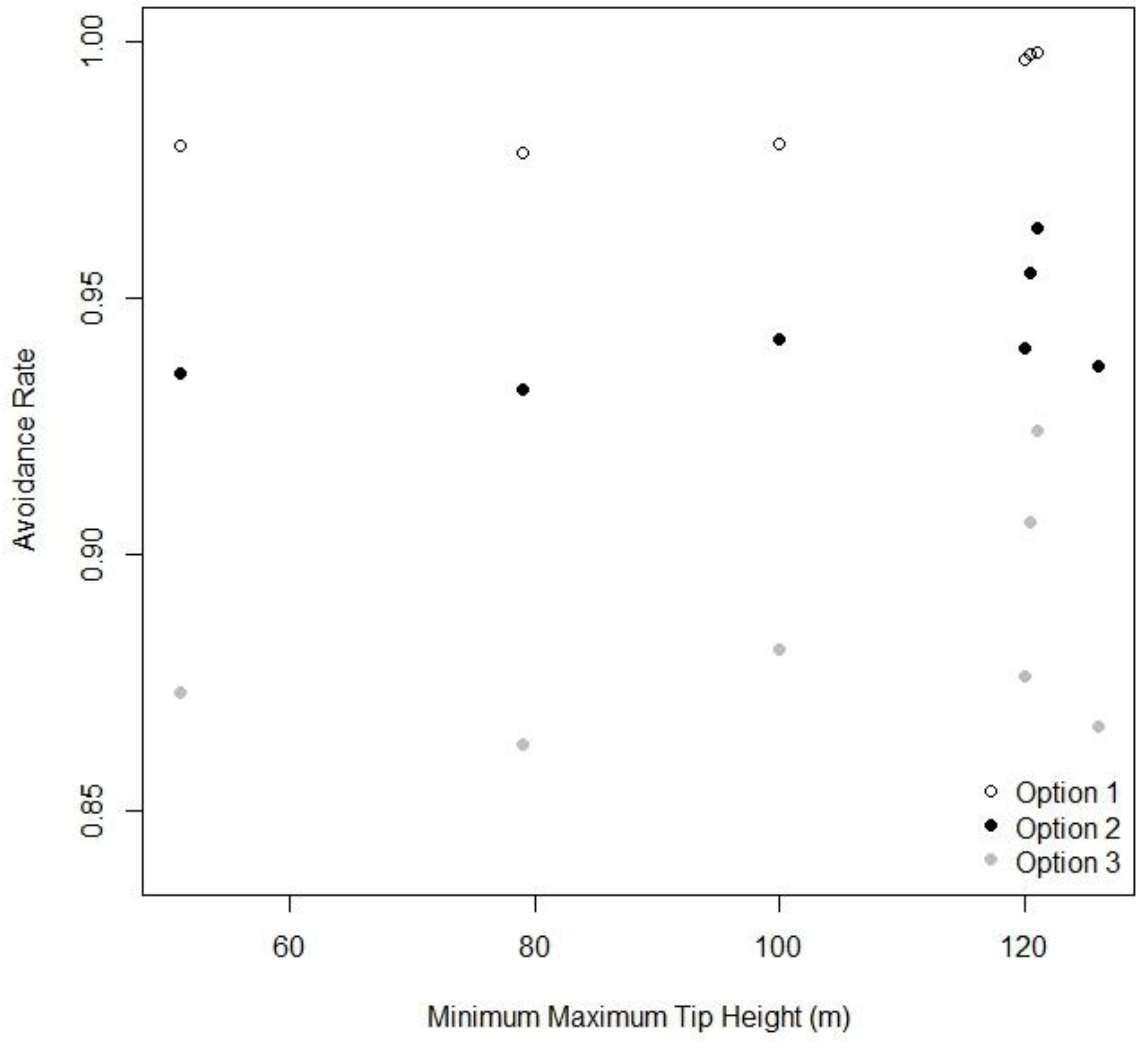


Figure 5.3 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for black-headed gull.

The inclusion of sites with smaller turbines did not appear to strongly influence the final within-windfarm avoidance rates derived for black-headed gull using any of the three model options (Figure 5.3).

We consider within-windfarm avoidance rates of 0.9795 (± 0.0033 SD) for the basic Band model, and 0.8731 (± 0.0056 SD) for the extended Band model to be realistic, precautionary values given the data available. Whilst we identified several sites as having a strong influence over the final values derived, we do not feel there is sufficient reason to exclude these data from our analysis. It should be noted that the influence of these sites occurs in similar magnitudes in both positive and negative directions. The within-windfarm avoidance rates derived, especially for option 1, remain relatively stable regardless of which sites are included in the analysis. We did not identify any strong impact of turbine size on the final within-windfarm avoidance rate derived.

Common gull

A total of 841,008 common gulls were expected to have passed through three sites – Gneizdzewo (three studies), Kessingland (one study) and Hellrigg (two studies) – over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 3,405 collisions based on option 1 and 218 collisions based on option 2, and 129 based on option 3. However, in total only two common gull collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9995 (± 0.0003 SD) using option 1 of the Band model, 0.9918 (± 0.0046 SD) using option 2 of the Band model and 0.9861 (± 0.0078 SD) using option 3 of the Band model.

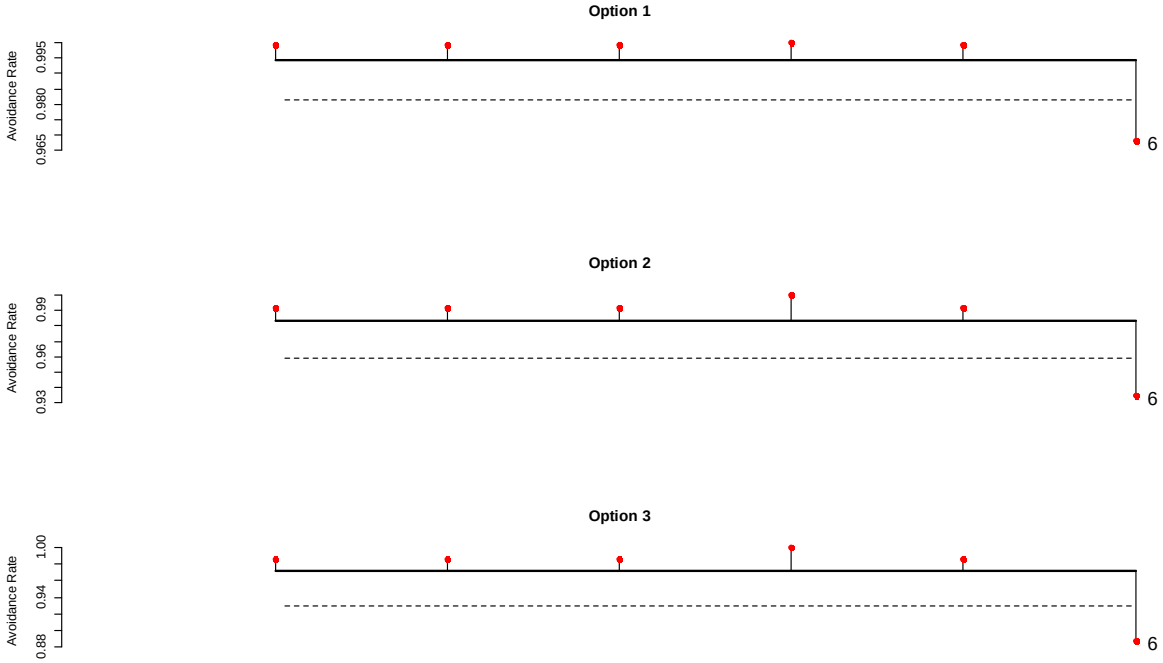


Figure 5.4 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for common gull. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± standard deviation, dots indicate mean within-windfarm

avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Site with high leverage is 6 – Hellrigg in 2012/13.

For all three model options, Hellrigg in 2012/13 appears to have a strong influence over the final within-windfarm avoidance rate derived (Figure 5.4). This is likely to reflect the fact that of the total number of common gulls estimated to have flown through windfarms, over 94% were estimated to have flown through Hellrigg in this year. Despite this, no collisions were recorded involving common gulls at Hellrigg in 2012/13. As a result, excluding these data from our analyses results in an overall within-windfarm avoidance rate of 0.9680 for option 1 of the Band model, 0.9345 for option 2 of the Band model and 0.8865 for option 3 of the Band model. However, we do not feel it would be appropriate to exclude such a substantial portion of our data from the analysis in this way.

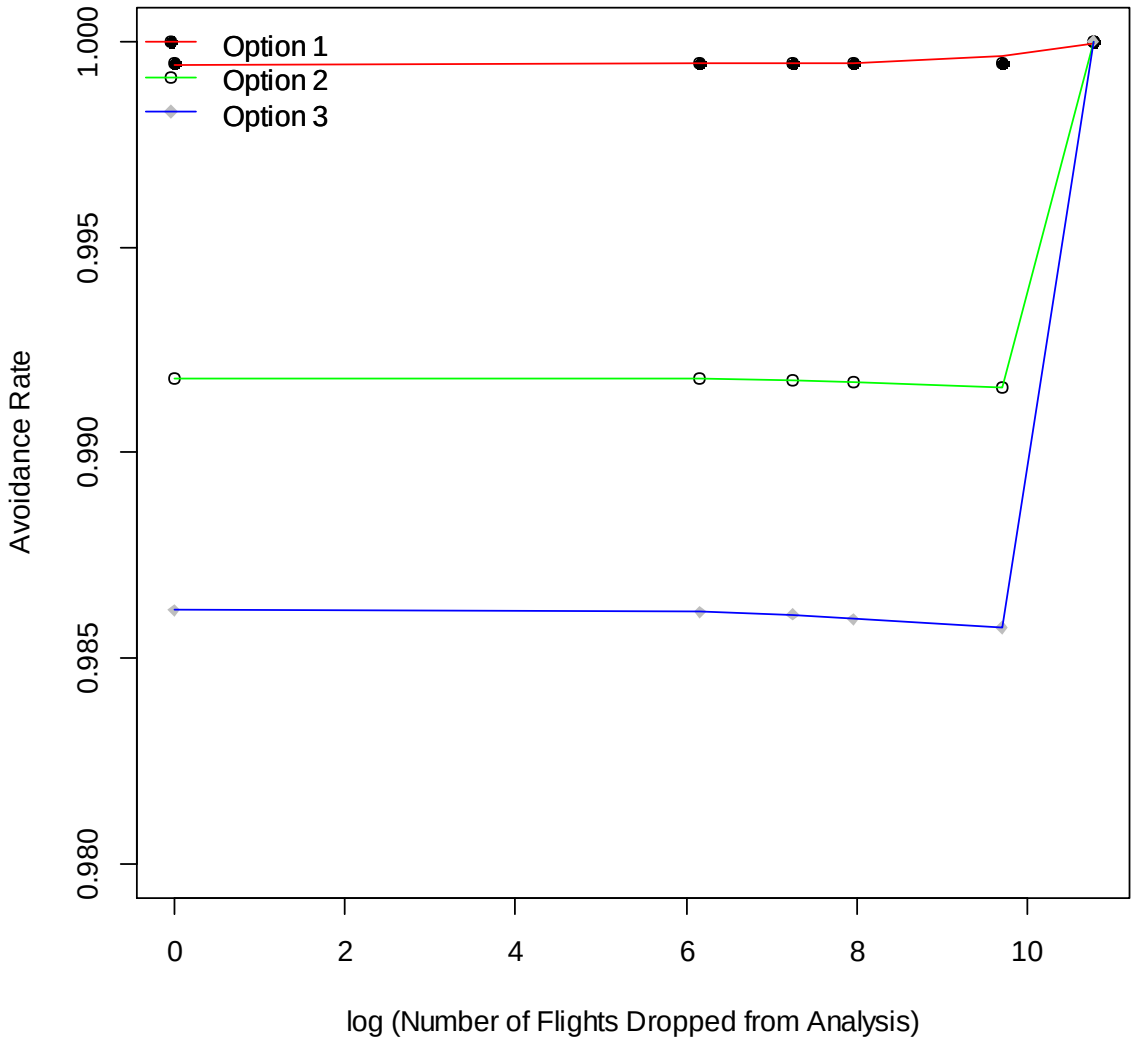


Figure 5.5 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for common gull.

For all three model options, the within-windfarm avoidance rate derived using ratio estimators remains stable until the only site remaining in the analysis is Hellrigg in 2012/13 (Figure 5.5). As stated above, this is likely to reflect the extremely high leverage of this data point.

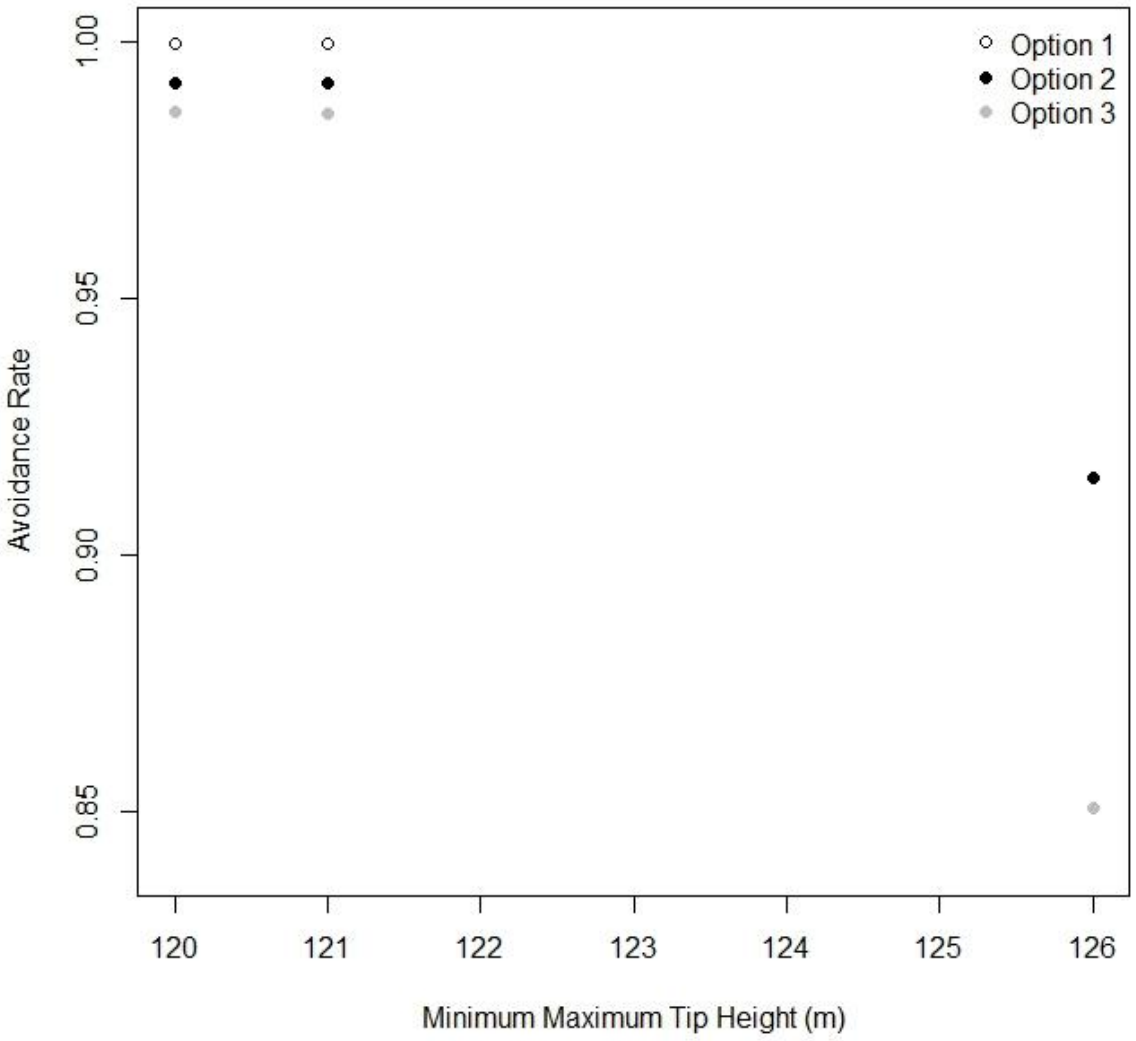


Figure 5.6 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for common gull.

Maximum tip height appeared to influence the within-windfarm avoidance rates reported, with lower within-windfarm avoidance rates associated with the tallest turbines (Figure 5.6). In reality, this is likely to reflect the fact that collisions were only recorded at Kessingland, the site with the largest turbines, and may, therefore, be coincidence.

Whilst data from Hellrigg in 2012/13 have strong leverage, this must be considered in the context of the sheer number of flights that were estimated at the site in that year, and in combination with the fact that collisions involving common gulls were only recorded at one of the three study sites in a single year. We therefore feel that within-windfarm avoidance rates of 0.9995 (± 0.0003 SD) for the basic Band model and 0.9861 (± 0.0078 SD) for the extended Band model are likely to reflect realistic, precautionary within-windfarm avoidance rates for common gulls. Whilst we feel there is no valid reason to exclude the data from Hellrigg in 2012/13 from our analyses, we feel that its high leverage means that the final within-windfarm avoidance rates derived must be treated with caution.

Herring gull

A total of 526,047 herring gulls were expected to have passed through seven sites – Avonmouth (four studies), Boudwijnkanaal (one study), Bouin (one study), Gneizdzewo (one study), Hellrigg (two studies), Kessingland (one study), Zeebrugge (three studies) – over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 2,157 collisions based on option 1, 1,147 collisions based on option 2, and 957 based on option 3. However, in total only nine herring gull collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9959 (±0.0006 SD) using option 1 of the Band model, 0.9924 (±0.0010 SD) using option 2 of the Band model and 0.9908 (±0.0012 SD) using option 3 of the Band model.

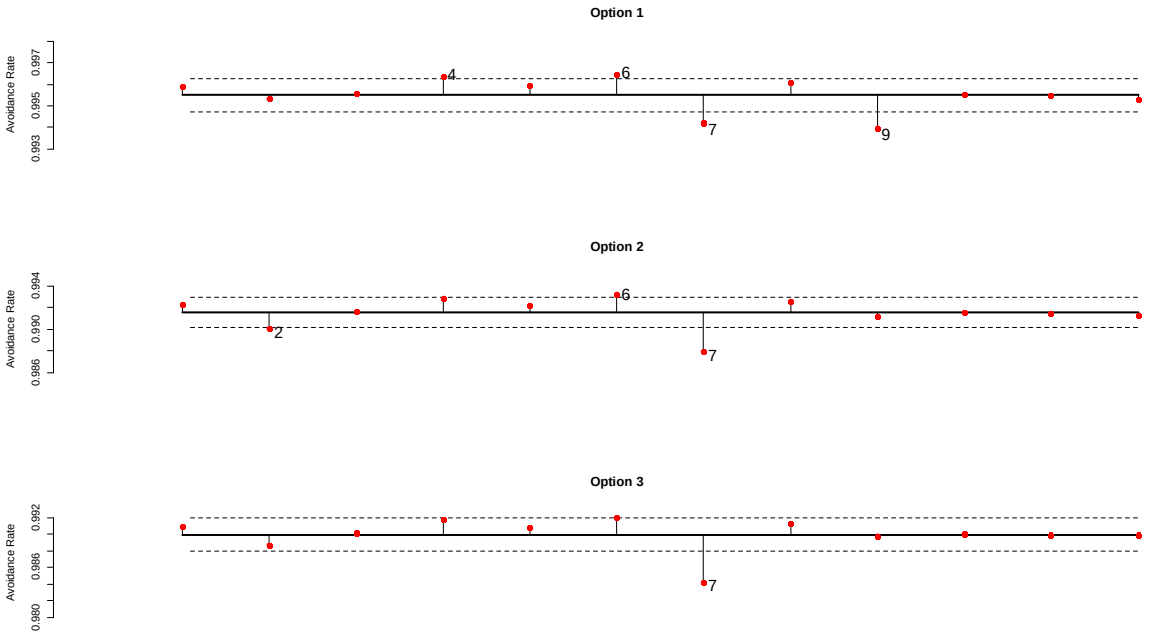


Figure 5.7 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for herring gull. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-

windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Sites with high leverage are 2 – Bouin, 4 – Kessingland, 6 – Zeebrugge (June-July 2001), 7 – Zeebrugge (September-October 2001) and 9 – Hellrigg (2012/13).

No obvious patterns were evident amongst the sites with high leverage (Figure 5.7). The exclusion of data from Kessingland and Zeebrugge (June-July 2001) from the analysis led to an increase in the overall within-windfarm avoidance rates as both these sites recorded two collisions over the course of their respective study periods. Whilst these were amongst the highest collision rates at the sites we considered, there is no evidence that turbine size played a role. Whilst the turbines at Zeebrugge were the smallest among our study sites, those at Kessingland were the largest. The exclusion of Hellrigg (2012/13) and Zeebrugge (September-October 2001) led to a decrease in the overall within-windfarm avoidance rates. This is likely to reflect the fact that whilst these data points represented the greatest numbers of birds passing through the sites, only two collisions were recorded at Zeebrugge (September-October 2001). It is worth noting that the magnitude of the effect of removing data from Zeebrugge was similar whether data from June-July 2001 or September-October 2001 were removed, although the effect was in opposing directions. Based on these analyses, we did not feel it was appropriate to exclude any data points from our analysis when deriving an overall within-windfarm avoidance rate for herring gull.

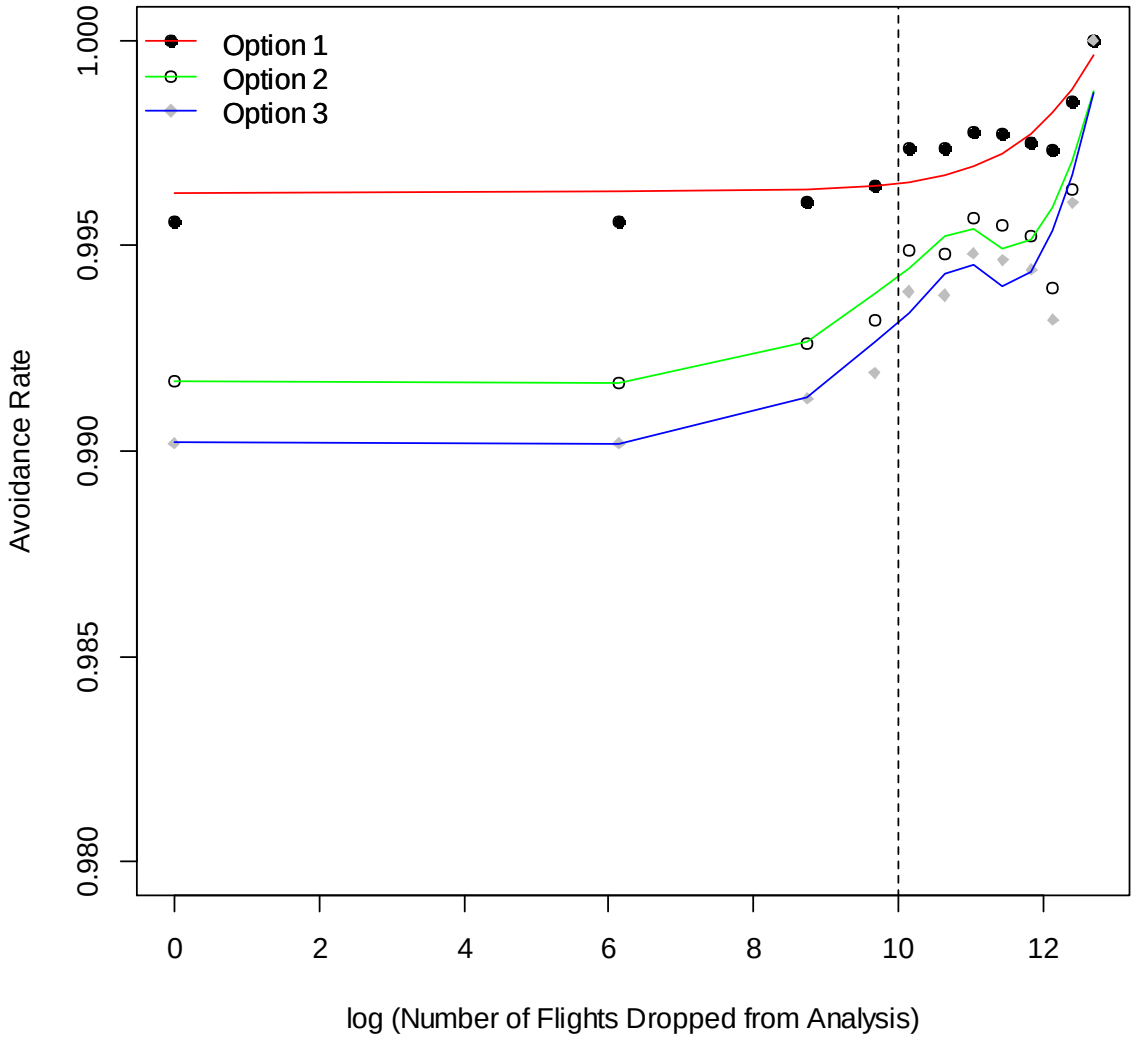


Figure 5.8 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for herring gull.

Dropping sites with lower levels of flight activity leads to an increase in the within-windfarm avoidance rates derived for herring gull using all three model options (Figure 5.8). Whilst ideally, within-windfarm avoidance rates would remain stable, regardless of the number of flights included in the analysis, it does suggest that the rates derived using the full dataset may be realistic, precautionary values.

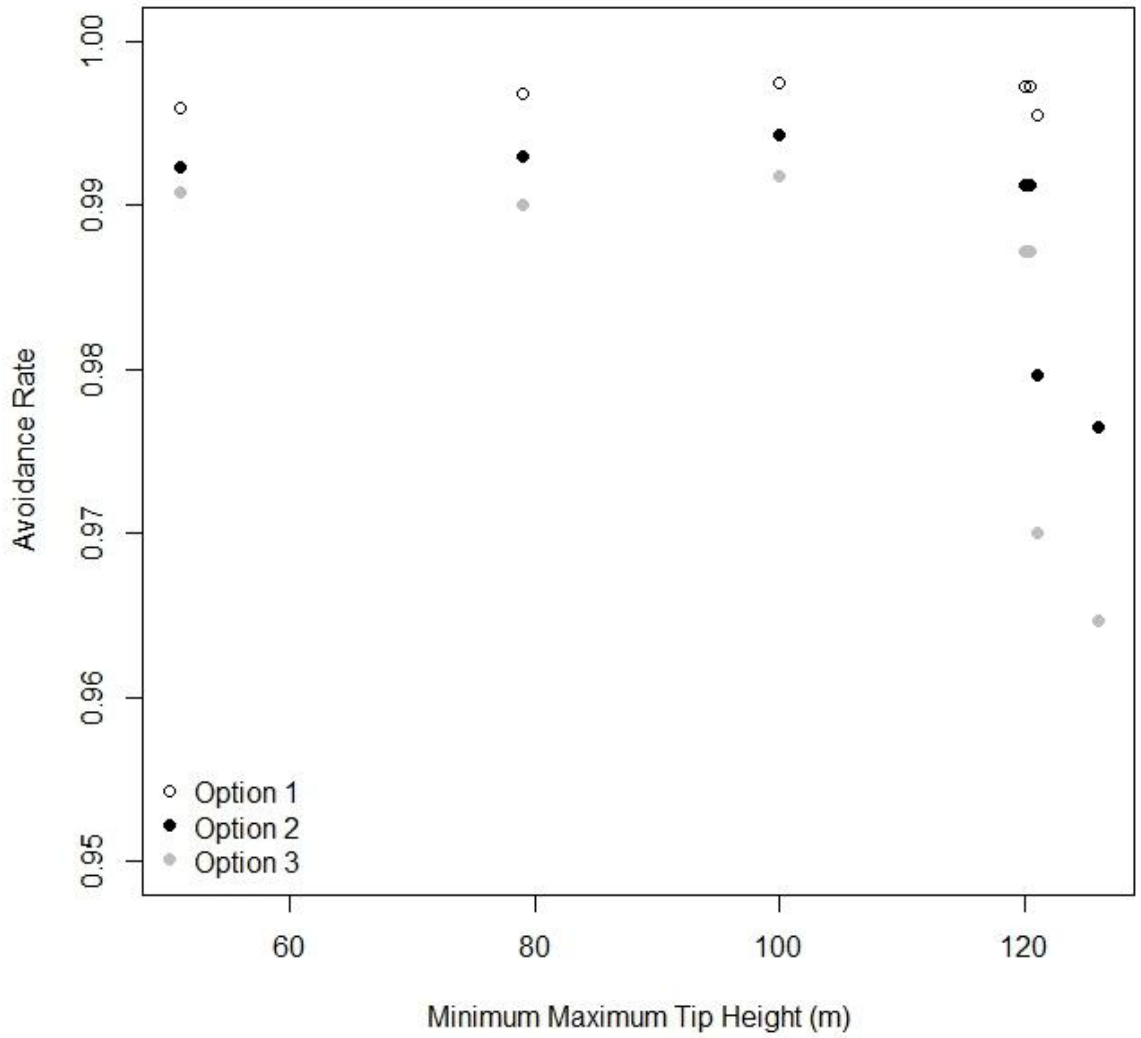


Figure 5.9 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for herring gull.

Using option 1 of the Band model, there does not appear to be a relationship between turbine size and the within-windfarm avoidance rates derived using ratio estimators (Figure 5.9). However, in the case of options 2 and 3, there is a trend for lower within-windfarm avoidance rates with larger turbines. This apparent discrepancy is likely to reflect differences between the proportion of birds observed flying at collision risk height and the proportion of birds estimated to fly at collision risk height from generic distributions. The generic distributions estimated a lower proportion of birds flying at collision risk height for the larger turbines, meaning the predicted collision rate, and therefore overall within-windfarm avoidance rate, was reduced.

We consider within-windfarm avoidance rates of 0.9959 (±0.0006 SD) for the basic Band model, and 0.9908 (±0.0012 SD) for the extended Band model to be realistic,

precautionary values given the data available. Whilst we identified several sites as having a strong influence over the final values derived, we do not feel there is sufficient reason to exclude these data from our analysis. It should be noted that the influence of these sites occurs in similar magnitudes in both positive and negative directions. We did not identify any strong impact of turbine size on the final within-windfarm avoidance rate derived.

Lesser black-backed gull

A total of 101,745 lesser black-backed gulls were expected to have passed through three sites – Hellrigg (two studies), Kessingland (one study) and Zeebrugge (three studies) – over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 1,110 collisions based on option 1,512 collisions based on option 2, and 473 based on option 3. However, in total only two lesser black-backed gull collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9982 (±0.0005 SD) using option 1 of the Band model, 0.9960 (±0.0010 SD) using option 2 of the Band model and 0.9957 (±0.0011 SD) using option 3 of the Band model.

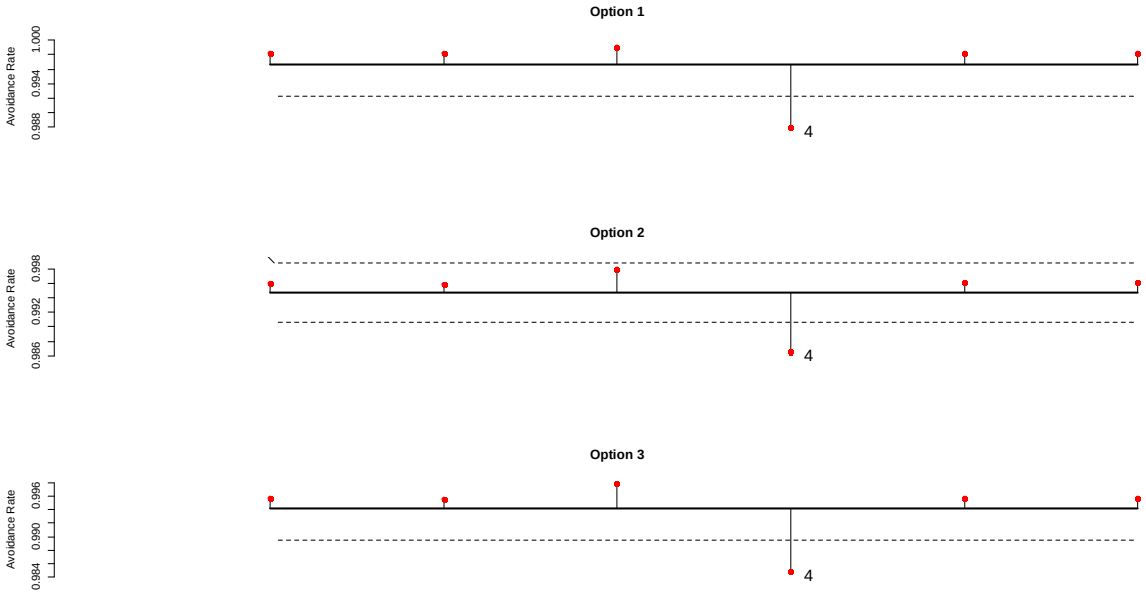


Figure 5.10 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for lesser black-backed gull. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Site with high leverage is 4 – Zeebrugge (September-October 2001).

Data from all three model options indicated that Zeebrugge in September-October 2001 had a relatively high leverage on the final within-windfarm avoidance rates derived using ratio estimators (Figure 5.10). This is likely to reflect the fact that Zeebrugge in September-October 2001 had the highest levels of bird activity by some distance. Despite this, only a single collision was recorded over the study period. Excluding these data from the analysis results in within-windfarm avoidance rates of 0.9878 using option 1, 0.9865 using option 2 and 0.9847 using option 3. However, we do not feel it is appropriate to exclude data in this way.

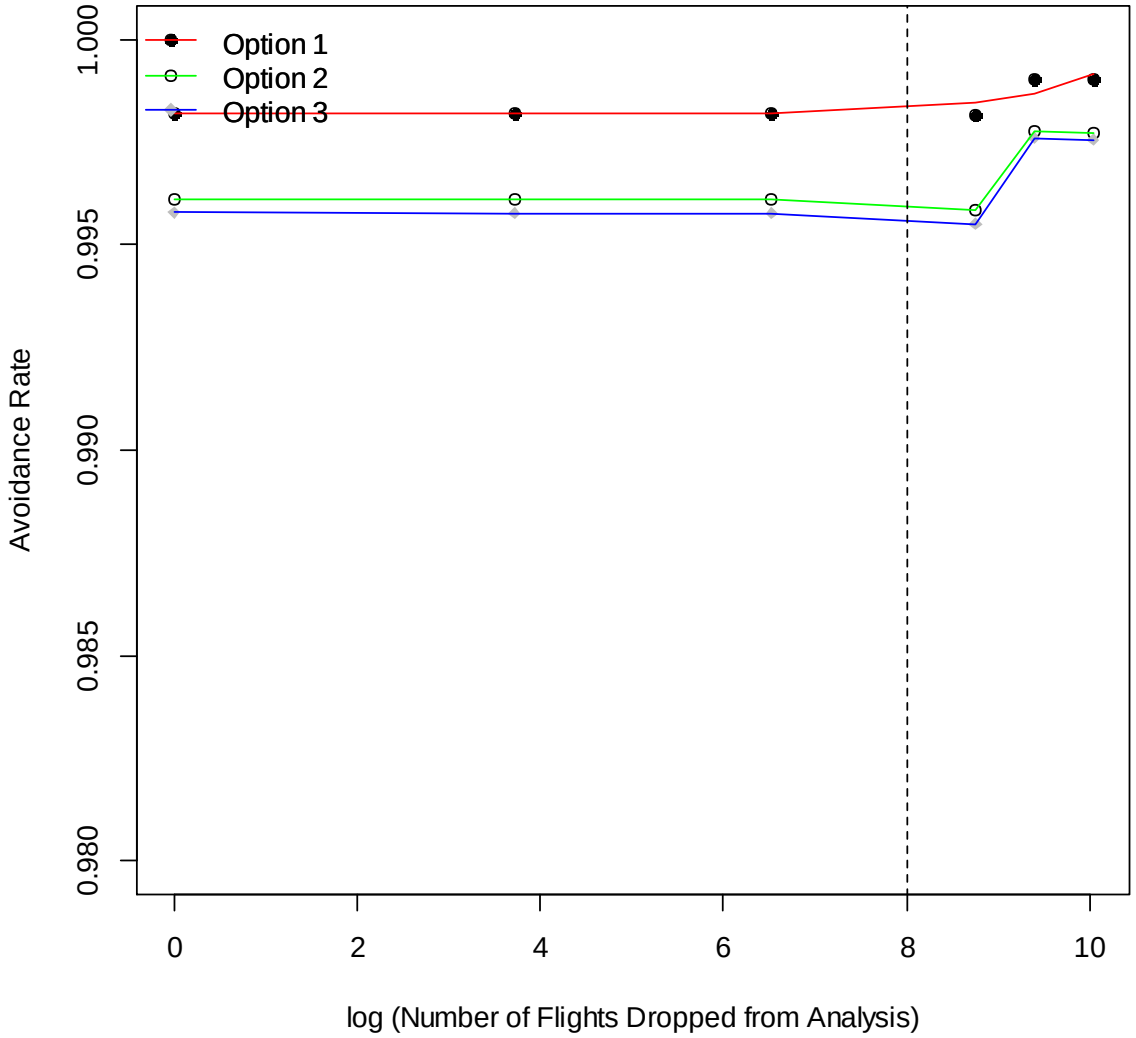


Figure 5.11 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for lesser black-backed gull.

Using option 1 of the Band model to derive within-windfarm avoidance rates, values remain fairly stable regardless of the number of birds recorded flying through the study sites (Figure 5.11). Using options 2 and 3 the final value remains relatively

stable until the first 6,000 flights have been removed. This is likely to reflect that fact that whilst a relatively high number of birds were predicted to have flown through the final two sites, only a single collision was recorded.

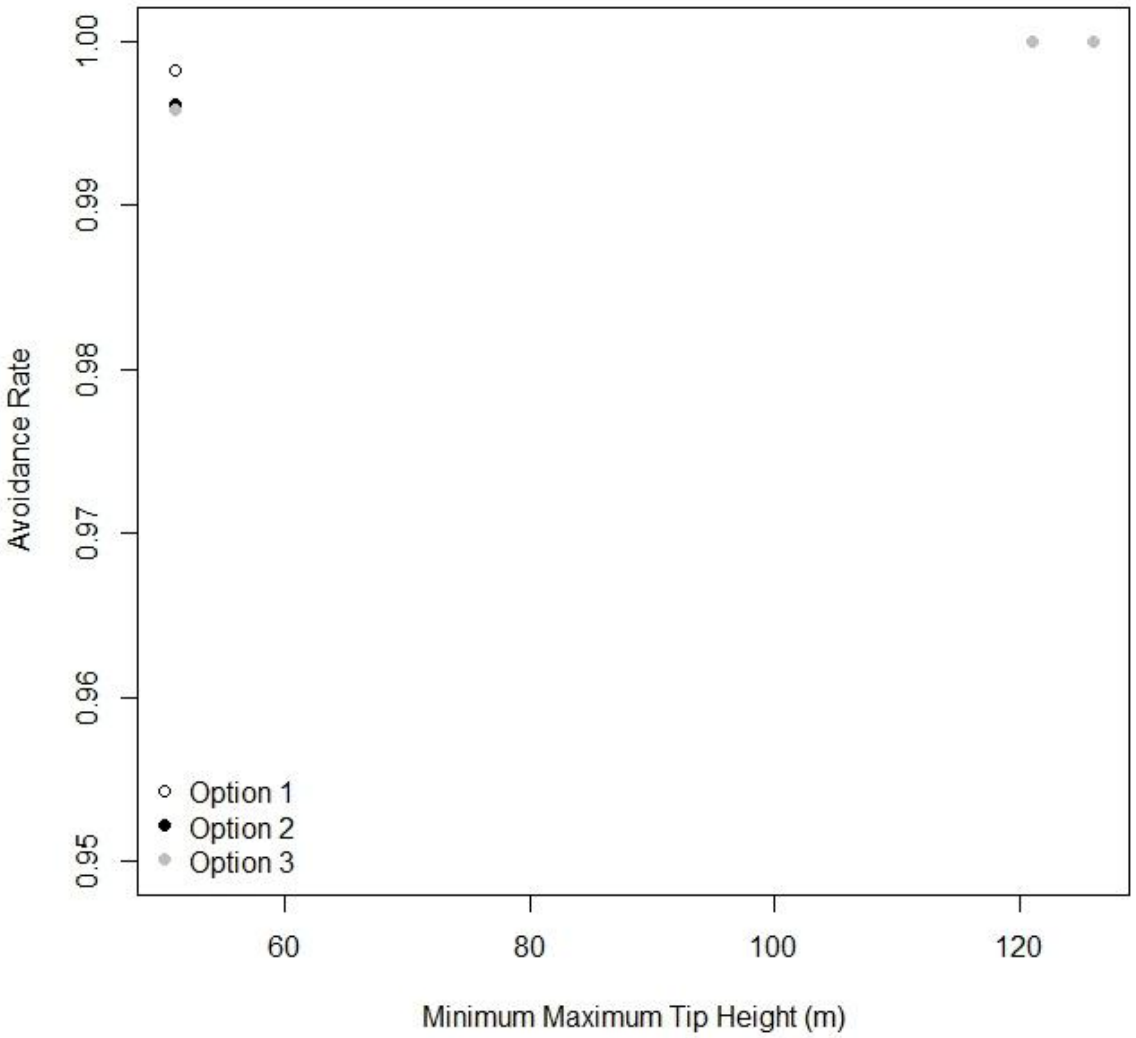


Figure 5.12 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for lesser black-backed gull.

Excluding smaller turbines did not appear to have a significant impact on the final within-windfarm avoidance rate derived for lesser black-backed gull using any of the three model options (Figure 5.12).

Whilst data from Zeebrugge in September-October 2001 had a relatively high leverage on the final within-windfarm avoidance rates derived, we did not feel there was a compelling reason to exclude these data from our analysis. Based on the data available for lesser black-backed gull, we consider within-windfarm avoidance rates

of 0.9982 (±0.0005 SD) for the basic Band model and 0.9957 (±0.0011 SD) for the extended Band model to be realistic, precautionary values given the data available. However, given the data come from only three sites and incorporate a relatively small number of flights through the windfarm, we feel these values should be treated with caution. Whilst we identified several sites as having a strong influence over the final values derived, we do not feel there is sufficient reason to exclude these data from our analysis. It should be noted that the influence of these sites occurs in similar magnitudes in both positive and negative directions. We did not identify any strong impact of turbine size on the final within-windfarm avoidance rate derived.

Small gulls

A total of 1,589,953 small gulls were expected to have passed through eight sites over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 5,263 collisions based on option 1,801 collisions based on option 2, and 427 based on option 3. However, in total only 42 small gull collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9921 (±0.0015 SD) using option 1 of the Band model, 0.9481 (±0.0032 SD) using option 2 of the Band model and 0.9027 (±0.0068 SD) using option 3 of the Band model.

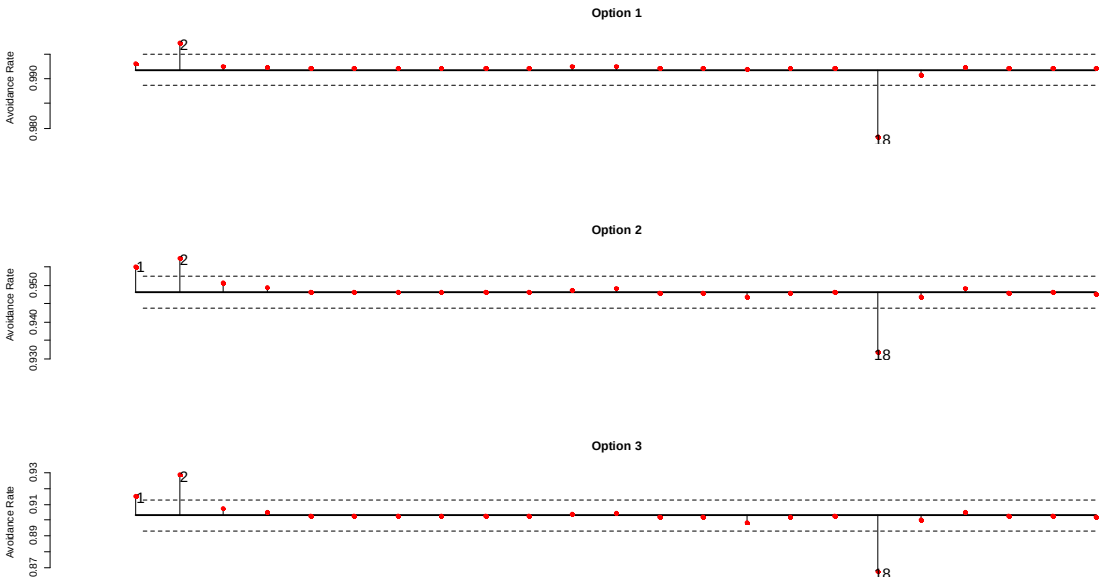


Figure 5.13 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for small gulls. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard

deviation in the overall mean within-windfarm avoidance rate. Sites with high leverage are 1 – black-headed gull at Boudwijnkanaal in October 2015, 2 – black-headed gull at Bouin and 18 – common gull at Hellrigg in 2012/13.

For all three model options, the exclusion of data from black-headed gull at Bouin results in an increased within-windfarm avoidance rate (Figure 5.13). This is likely to be due to the presence of the turbines on the edge of a black-headed gull breeding colony. This may have led to a greater number of flights through the rotor-swept area of turbines by adult birds returning to provision chicks and/or newly fledged chicks less experienced at flying. As a result of the relatively high collision rate, including this site in our analysis reduced the overall rate derived using option 1. The leverage of data from Hellrigg in 2012/13 was of a similar magnitude, but in the opposite direction. Despite having the highest level of small gull activity and a high proportion of birds flying at collision risk height, no collisions were recorded at this site, in this year. However, we did not consider there to be a valid reason for excluding these sites from our analysis.

Using options 2 and 3, excluding data for black-headed gull from Boudwijnkanaal in October 2005 also resulted in an increase in the overall within-windfarm avoidance rates. The reason for this differing from the results for option 1 is that the modelled flight height distribution predicts a lower proportion of birds at collision risk height. As a consequence, the predicted collision rate, and therefore the within-windfarm avoidance rate, is reduced.

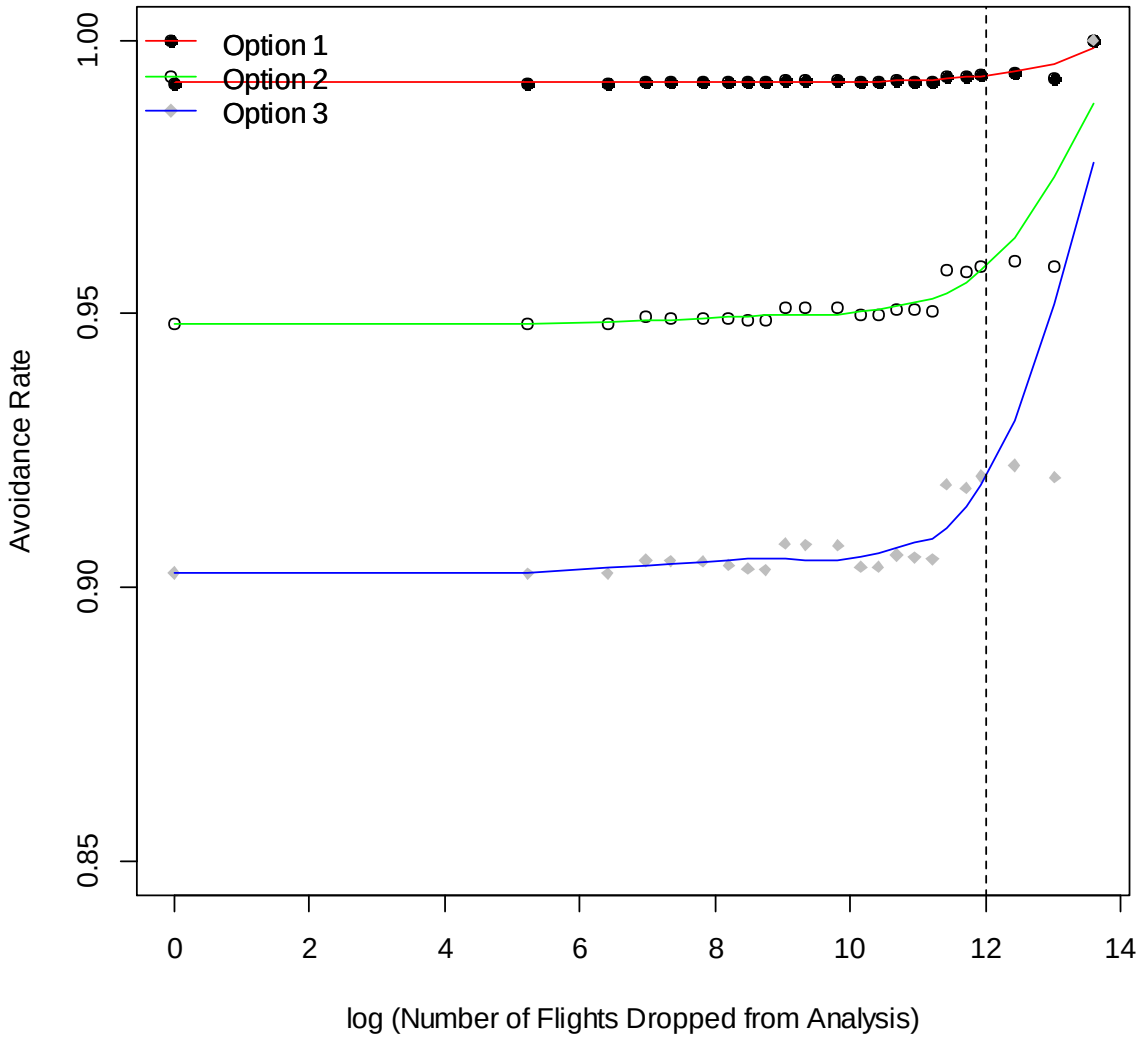


Figure 5.14 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for small gulls.

Within-windfarm avoidance rates derived using all three model options remain relatively stable as the first 160,000 flights through windfarms were dropped from the analysis (Figure 5.14), before increasing as only the sites with the highest levels of gull activity remain. This reflects the fact that at several of the sites with the highest levels of gull activity, no collisions were recorded, resulting in an overall increase in the within-windfarm avoidance rates as other sites were dropped.

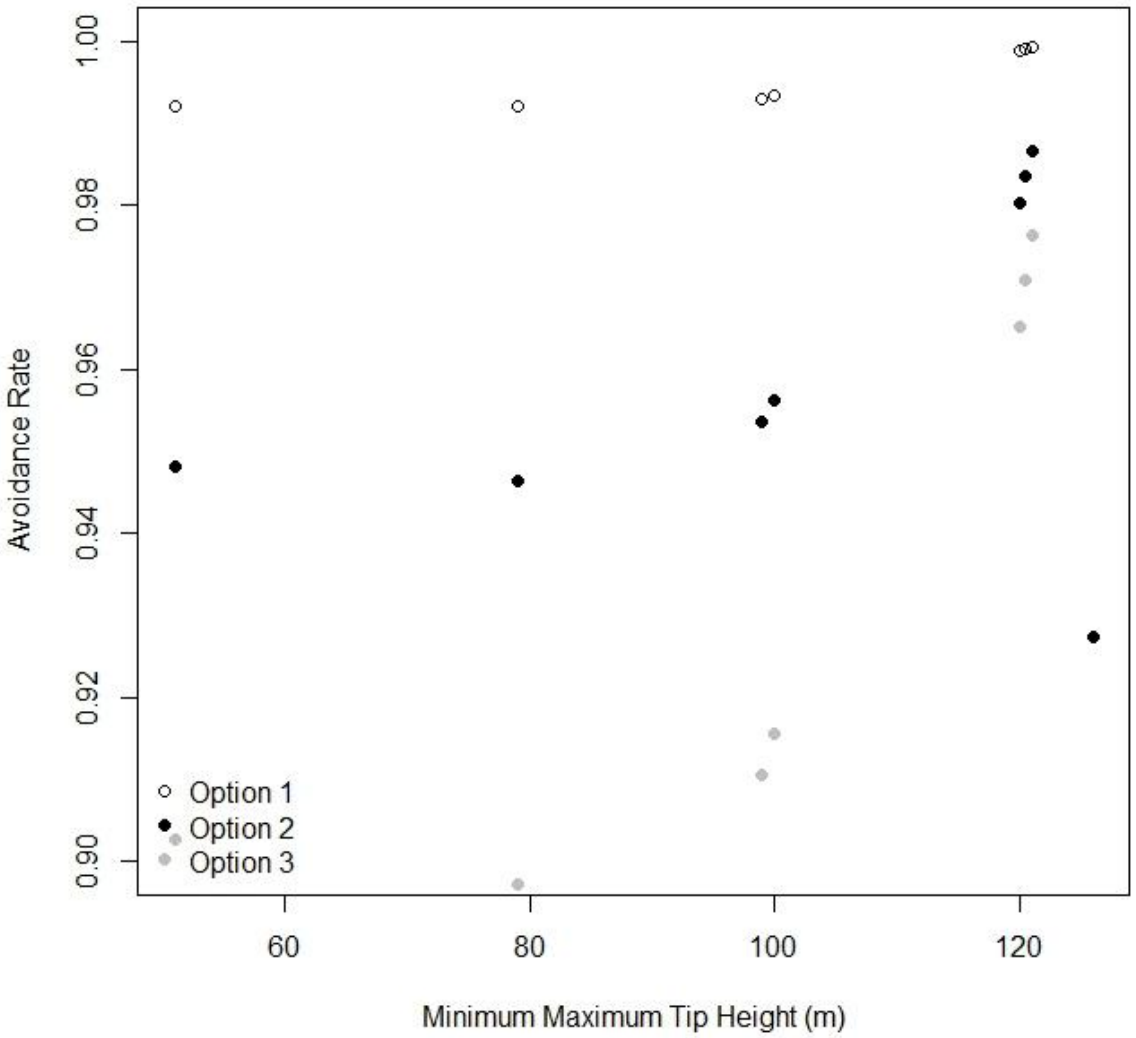


Figure 5.15 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for small gulls.

Using option 1 of the Band model, there does not appear to be a relationship between turbine size and the within-windfarm avoidance rates derived using ratio estimators (Figure 5.15). However, in the case of options 2 and 3, there is a trend for higher within-windfarm avoidance rates with larger turbines. The reason for this discrepancy is unclear, although it may reflect differences in the proportion of birds at collision risk height between the observed data and modelled distributions.

We consider within-windfarm avoidance rates of 0.9921 (±0.0015 SD) for the basic Band model, and 0.9027 (±0.0068 SD) for the extended Band model to be realistic, precautionary values given the data available. Whilst we identified several sites as having a strong influence over the final values derived, we do not feel there is sufficient reason to exclude these data from our analysis. It should be noted that the influence of these sites occurs in similar magnitudes in both positive and negative

directions. We did not identify any strong impact of turbine size on the final within-windfarm avoidance rate derived using option 1 of the Band model.

Large gulls

A total of 639,560 large gulls were expected to have passed through seven sites – Avonmouth (four studies, one species), Boudwijnkanaal (two studies, two species), Bouin (one study, one species), Gniezdzewo (three studies, three species), Hellrigg (three studies, three species), Kessingland (one study, three species) and Zeebrugge (three studies, two species) – over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 3,368 collisions based on option 1, 1,684 collisions based on option 2, and 1,452 based on option 3. However, in total only 42 large gull collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9956 (±0.0004 SD) using option 1 of the Band model, 0.9912 (±0.0007 SD) using option 2 of the Band model and 0.9898 (±0.0009 SD) using option 3 of the Band model.

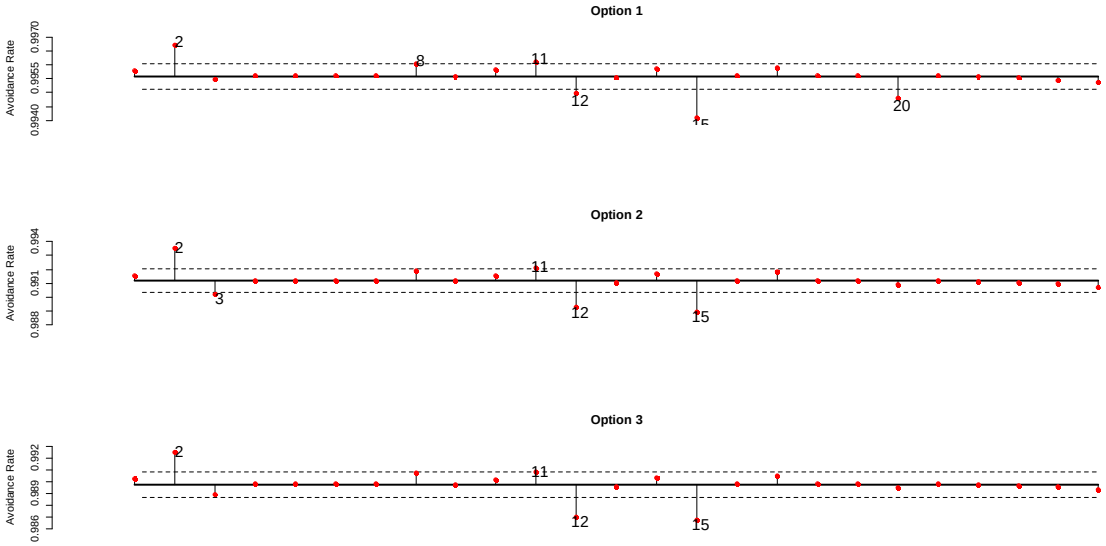


Figure 5.16 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for large gulls. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Sites with high leverage are 2 – herring/lesser black-backed gull, Boudwijnkanaal (October 2005), 3 – herring gull, Bouin; 8 – herring gull, Kessingland; 11 – herring gull, Zeebrugge (June-July 2001); 12 – herring gull, Zeebrugge (September-October 2001); 15 – lesser black-backed gull, Zeebrugge (September-October 2001); and 20 – herring gull, Hellrigg (2012/13).

There is no obvious pattern to the sites which have high leverage over the final derived within-windfarm avoidance rates (Figure 5.16). Excluding the data for herring/lesser black-backed gull at Boudwijnkanaal in October 2005, herring gull for Kessingland and herring gull for Zeebrugge in June-July 2001 results in an increase in the overall within-windfarm avoidance rate. The size of turbines at these sites varies from small (51 m maximum turbine height at Zeebrugge) to large (126 m maximum turbine height at Kessingland) so the inclusion of different sizes of turbines does not appear to have influenced the within-windfarm avoidance rates derived. In contrast, the inclusion of date for herring gull and lesser black-backed gull at Zeebrugge in September-October 2001 and for herring gull at Hellrigg in 2012/13 results in an increase in the overall within-windfarm avoidance rate derived. In these cases, the increase in the within-windfarm avoidance rates is likely to be linked to the relatively high activity levels at these sites and relatively low collision rates. We do not consider there to be a valid reason for excluding these sites from the analysis.

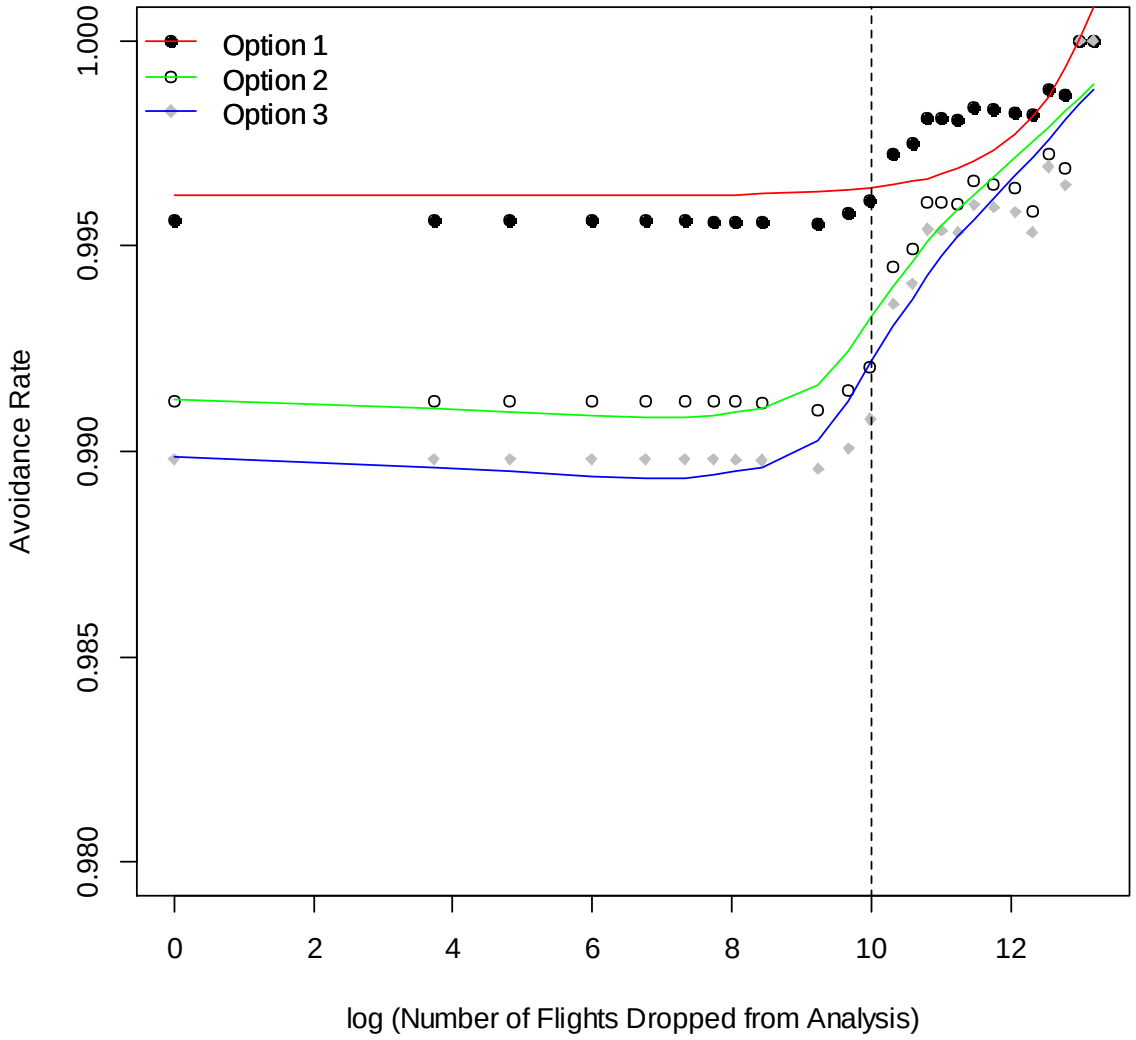


Figure 5.17 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for large gulls.

Within-windfarm avoidance rates derived using all three model options remain relatively stable as the first 22,000 flights through windfarms are dropped from the analysis (Figure 5.17), before increasing as only the sites with the highest levels of gull activity remain. This reflects the fact that at several of the sites with the highest levels of gull activity, no collisions were recorded, resulting in an overall increase in the within-windfarm avoidance rates as other sites were dropped.

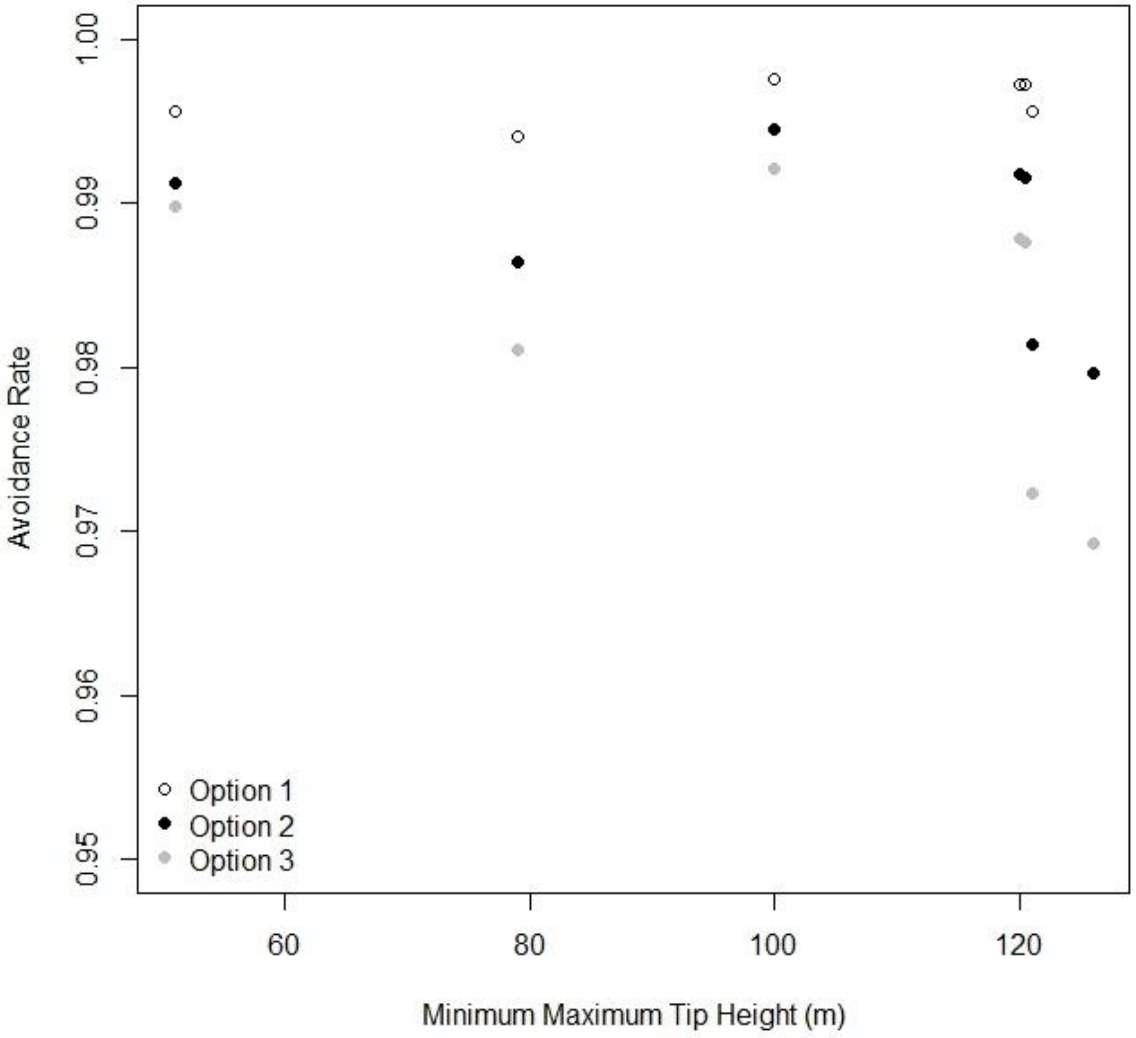


Figure 5.18 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for large gulls.

Using option 1 of the Band model, there does not appear to be a relationship between turbine size and the within-windfarm avoidance rates derived using ratio estimators (Figure 5.18). However, in the case of options 2 and 3, there is a trend for lower within-windfarm avoidance rates with larger turbines. This apparent

discrepancy is likely to reflect differences between the proportion of birds observed flying at collision risk height and the proportion of birds estimated to fly at collision risk height from generic distributions. The generic distributions estimated a lower proportion of birds flying at collision risk height for the larger turbines, meaning the predicted collision rate, and therefore overall within-windfarm avoidance rate, was reduced.

We consider within-windfarm avoidance rates of 0.9956 (±0.0004 SD) for the basic Band model, and 0.9898 (±0.0009 SD) for the extended Band model to be realistic, precautionary values given the data available. Whilst we identified several sites as having a strong influence over the final values derived, we do not feel there is sufficient reason to exclude these data from our analysis. It should be noted that the influence of these sites occurs in similar magnitudes in both positive and negative directions. We did not identify any strong impact of turbine size on the final within-windfarm avoidance rate derived.

All gulls

A total of 2,567,124 gulls were expected to have passed through seven sites over the course of their respective study periods. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 10,052 collisions based on option 1, 4,054 collisions based on option 2, and 3,271 based on option 3. However, in total only 107 gull collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9893 (±0.0007 SD) using option 1 of the Band model, 0.9735 (±0.0014 SD) using option 2 of the Band model and 0.9672 (±0.0018 SD) using option 3 of the Band model.

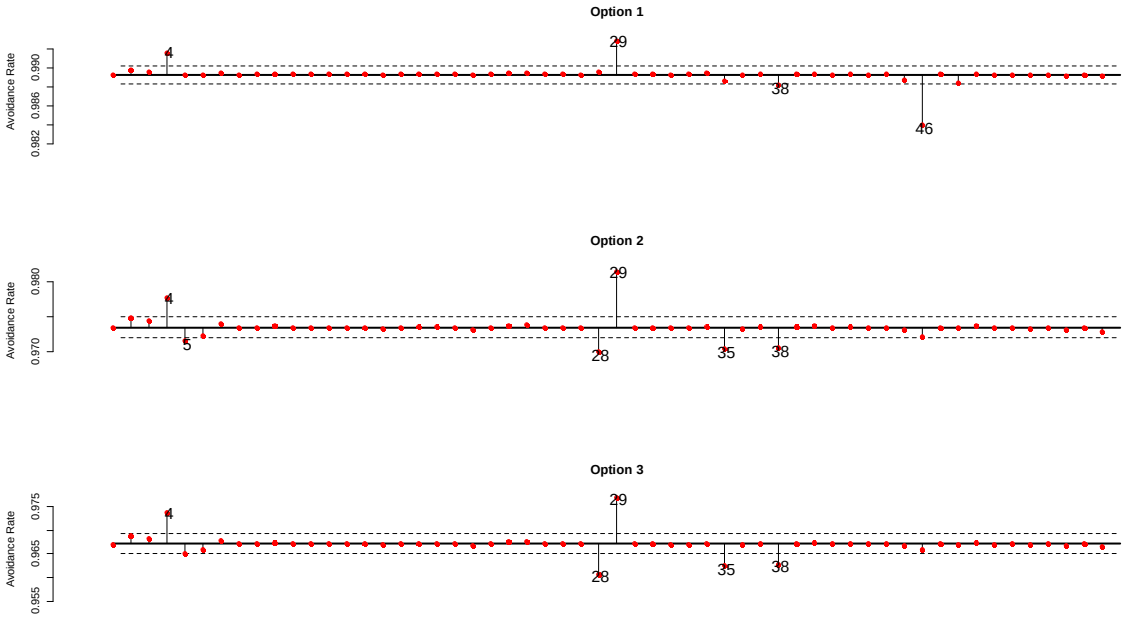


Figure 5.19 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for all gulls. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Sites with high leverage are 4 – black-headed gulls, Bouin; 5 – gull spp, Bouin; 28 – gull spp, Oosterbierum (autumn 1990); 29 – gull spp, Oosterbierum (spring 1991); 35 – herring gull, Zeebrugge (September-October 2001); 38 – lesser black-backed gull in Zeebrugge (September-October 2001); 46 – common gull, Hellrigg (2012/13).

For all three model options, excluding data for black-headed gulls at Bouin and gull spp at Oosterbierum in autumn 1990, results in an increase in the overall within-windfarm avoidance in the final derived within-windfarm avoidance rates (Figure 5.19). There are no obvious commonalities between these sites. The turbines at Oosterbierum are relatively small with a maximum tip height of 50 m, but those at Bouin are more typical of the sites in our study, with maximum tip heights of 100 m. Using option 1, the exclusion of data from lesser black-backed gull at Zeebrugge in September-October 2001 and common gull at Hellrigg in 2012/13 resulted in decreased within-windfarm avoidance rates. This is likely to reflect relatively high levels of bird activity in combination with very few recorded collisions at these sites, meaning they have a negative bias on the final, derived figures. This pattern was repeated for gull spp at Oosterbierum in spring 1991 and herring gulls and lesser black-backed gulls at Zeebrugge in September-October 2001 using options 2 and 3 and gull spp at Bouin using option 2.

It should be noted that for all three model options, leverage occurred in both directions. We did not feel there was a valid justification for excluding any of these data points from our analysis.

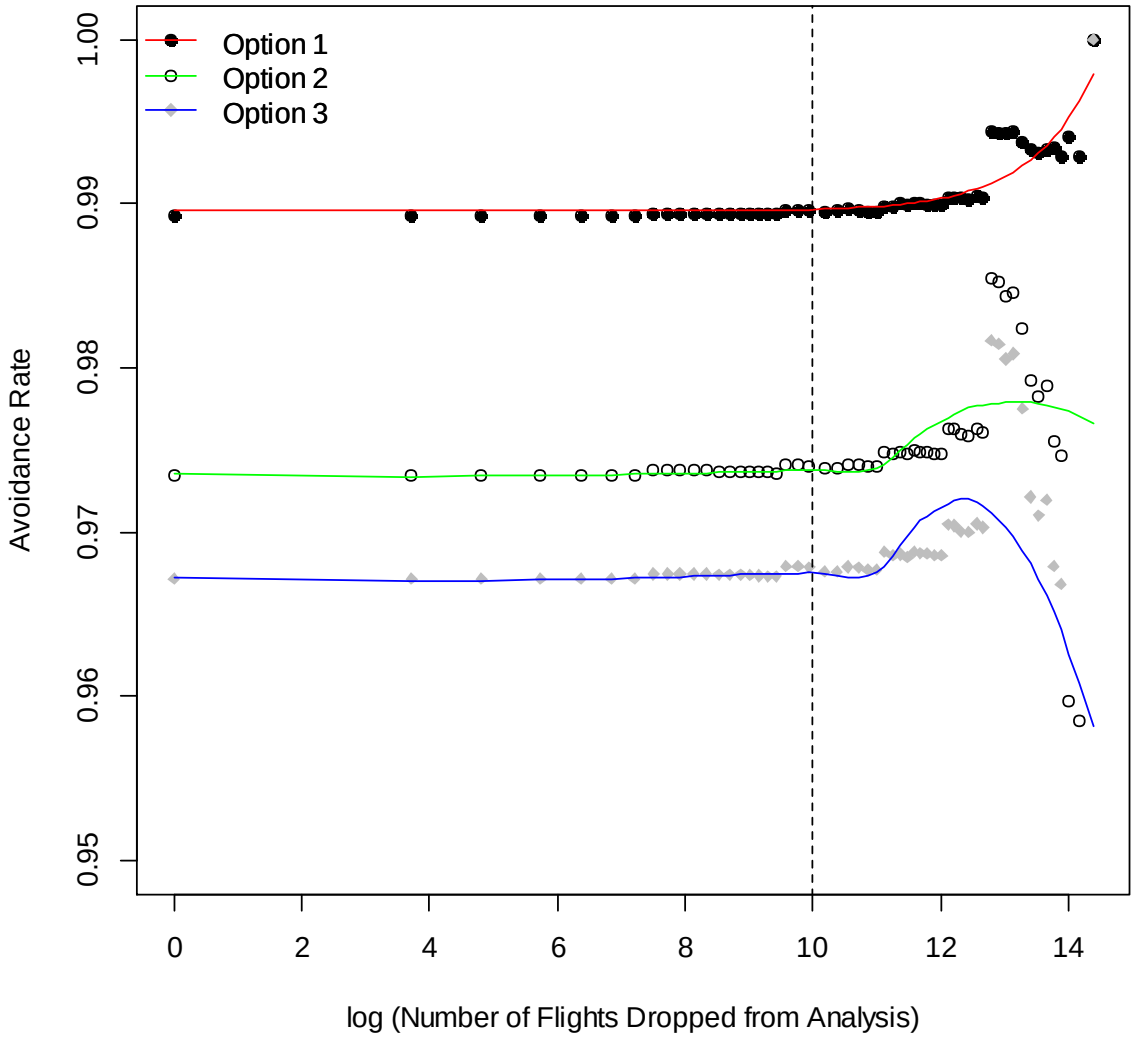


Figure 5.20 Impact of dropping data points (each site-year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model.

As might be expected, dropping sites from the analysis can influence the final within-windfarm avoidance rates. Only sites at which there is a relatively limited level of flight activity can be dropped from the analysis before the within-windfarm avoidance rates derived become less stable (Figure 5.20). In all three model options, this is noticeable after around 22,000 of the 2,605,681 flights through the windfarms have been removed (Figure 5.20).

Using option 1 of the Band model, dropping sites from the analysis results in an increase in the overall within-windfarm avoidance. This result suggests that, for

option 1, a higher flux rate is associated with a higher within-windfarm avoidance rate. Collisions between birds and turbines are relatively rare events, so studies carried out over a month or two may under-estimate mean annual within-windfarm avoidance rates if they are targeted to specific times of year when collisions are more likely. Amongst our datasets, there was a propensity for studies carried out during the breeding season. At Zeebrugge, both herring and lesser black-backed gulls showed a marked increase in their within-windfarm avoidance rates during the autumn than during the breeding season. At present, data are not robust enough to enable detailed analysis of seasonal patterns in within-windfarm avoidance behaviour, but this is an area that would benefit from such analyses as better data become available.

Initially a similar pattern is evident with option 3 of the Band model. However, when only the last few sites are included in the analysis, the final within-windfarm avoidance rates derived using ratio estimators start to fall (Figure 5.20). The decline is driven by breeding season data from Zeebrugge and Bouin, sites where turbines are situated close to the edge of breeding colonies. The reason the pattern is not evident in the within-windfarm avoidance rates derived using option 1 is the variation in the difference between the proportion of birds observed at rotor height in each study and those predicted to occur at rotor height based on the modelled flight height distribution. This is apparent when the differences between within-windfarm avoidance rates derived using options 1 and 2 are considered. Options 1 and 2 differ only in the proportion of birds predicted to fly at collision risk height. The proportion of birds estimated to fly at rotor height tended to be lower than the proportion of birds observed flying at rotor height (Appendix 7). As a result, the predicted collision rate, and therefore mean within-windfarm avoidance rate, was lower using option 2 than option 1. This difference becomes exaggerated under option 3 because, in addition to accounting for a lower proportion of birds flying at rotor height, fewer of these birds are predicted to collide.

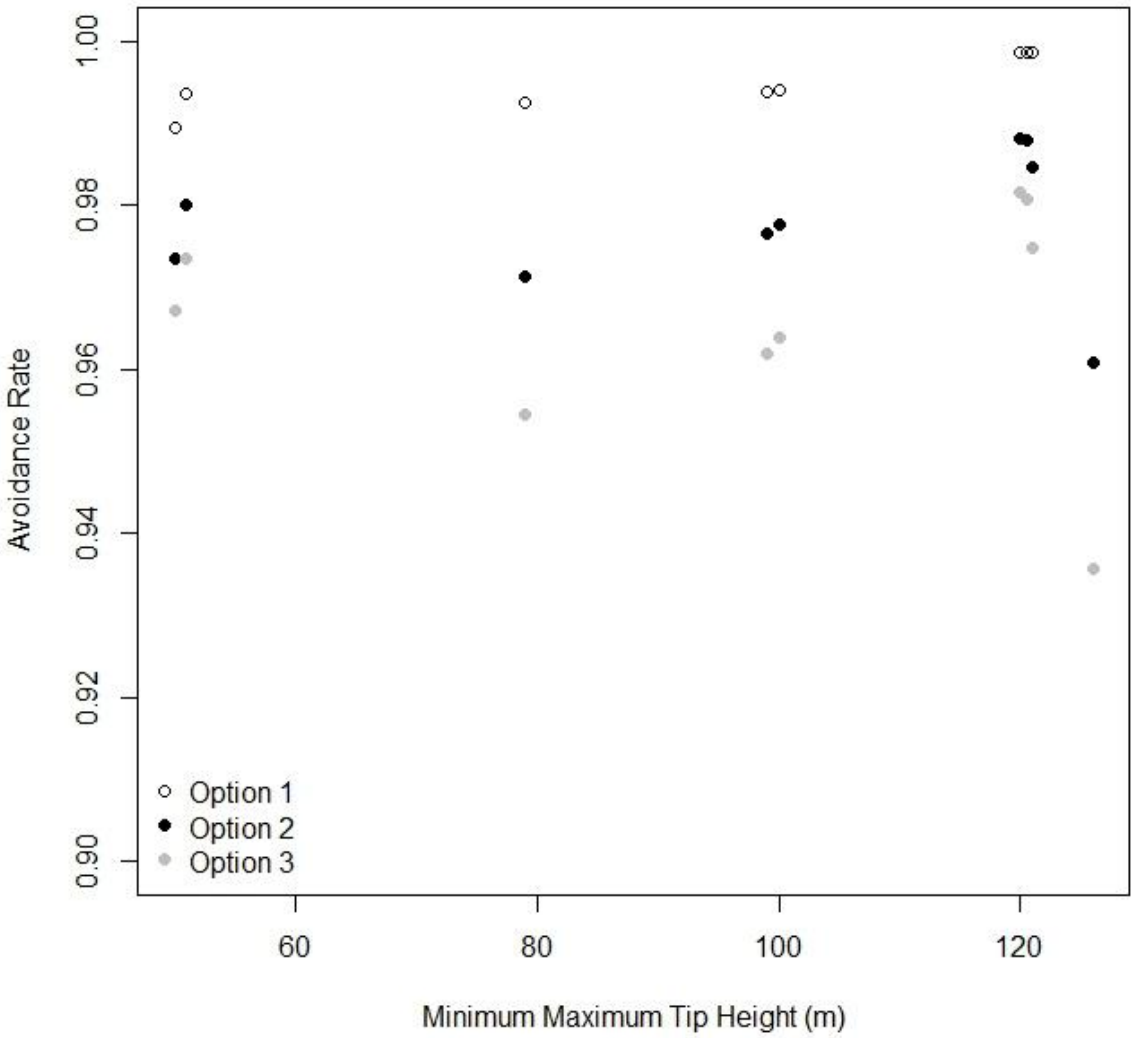


Figure 5.21 Impact of excluding sites with smaller turbines on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for large gulls.

Across all three model options there did not appear to be any consistent effect of excluding data collected from sites with smaller turbines on the final within-windfarm avoidance rates derived (Figure 5.21).

We consider that within-windfarm avoidance rates of 0.9893 (±0.0007 SD) for the basic Band model and 0.9672 (±0.0018 SD) for the extended Band model are realistic precautionary within-windfarm avoidance rates given the data available. Whilst we identified several sites as having a strong influence over the final values derived, we do not feel there is sufficient reason to exclude these data from our analysis. It should be noted that the influence of these sites occurs in similar magnitudes in both positive and negative directions. We did not identify any strong impact of turbine size on the final within-windfarm avoidance rate derived.

All terns

A total of 1,286,562 terns were expected to have passed through one site – Zeebrugge – during June 2004 and June 2005. After adjustments were made to this total to account for the proportion of birds flying at rotor height, the size of the rotor swept area and the probability of birds passing through the rotor-swept area without colliding, this was predicted to result in 1,408 collisions based on option 1, 1,299 collisions based on option 2, and 1,011 based on option 3. However, in total only 21 tern collisions were recorded across all sites during their respective study periods. This corresponds to within-windfarm avoidance rates of 0.9851 (±0.0022 SD) using option 1 of the Band model, 0.9838 (±0.0031 SD) using option 2 of the Band model and 0.9792 (±0.0040 SD) using option 3 of the Band model.

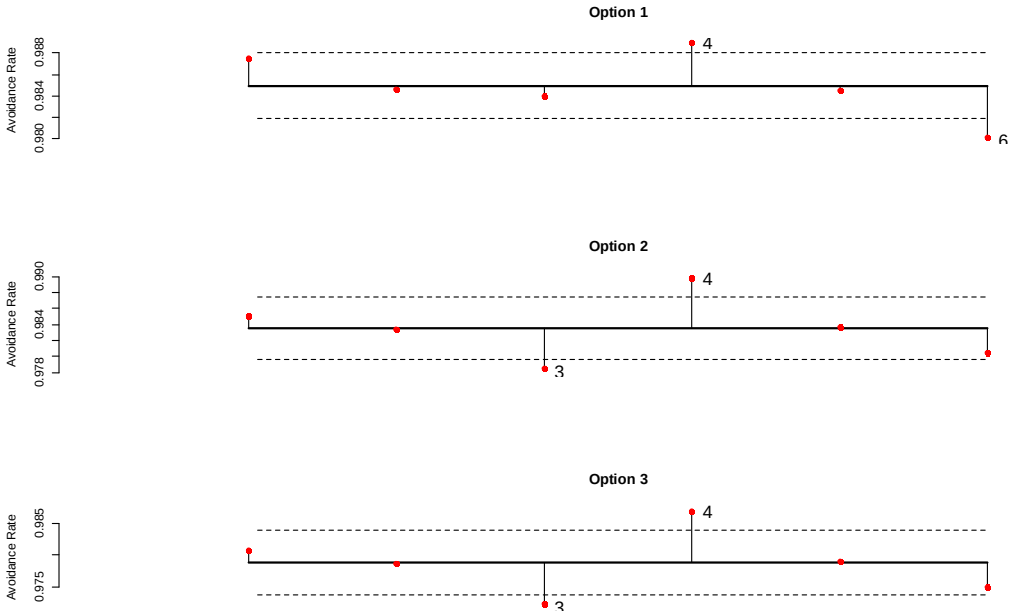


Figure 5.22 Leverage exerted by each site at which within-windfarm avoidance rates were calculated on the overall, mean within-windfarm avoidance rate derived for terns. Solid line indicates mean within-windfarm avoidance rate across all sites, broken line indicates mean within-windfarm avoidance rate across all sites ± 1 standard deviation, dots indicate mean within-windfarm avoidance rate with each site excluded from analysis. Sites are considered to have high leverage when their exclusion from the analysis leads to a change of more than 1 standard deviation in the overall mean within-windfarm avoidance rate. Points with high leverage are 3 – Sandwich tern in June 2004, 4 – common tern in June 2005, 6 – Sandwich tern in June 2005.

There was no obvious pattern in the data points with high leverage. Using all three model options, excluding common tern data from June 2005 was found to result in an increased within-windfarm avoidance rate, reflecting the relatively high collision rate involving this species in this year (Figure 5.22). Using option 1, excluding

Sandwich tern data from June 2005 resulted in a decrease in the within-windfarm avoidance rate derived. Using options 2 and 3 the same was true of Sandwich tern data in June 2004. This is likely to reflect the fact that relatively few collisions were recorded involving this species, despite a high flux rate. Differences between model options are likely to result from differences between the proportion of birds observed at collision risk height during surveys, and that estimated from the modelled distributions. We do not consider there to be a valid reason to exclude any of these data from our analysis when deriving within-windfarm avoidance rates.

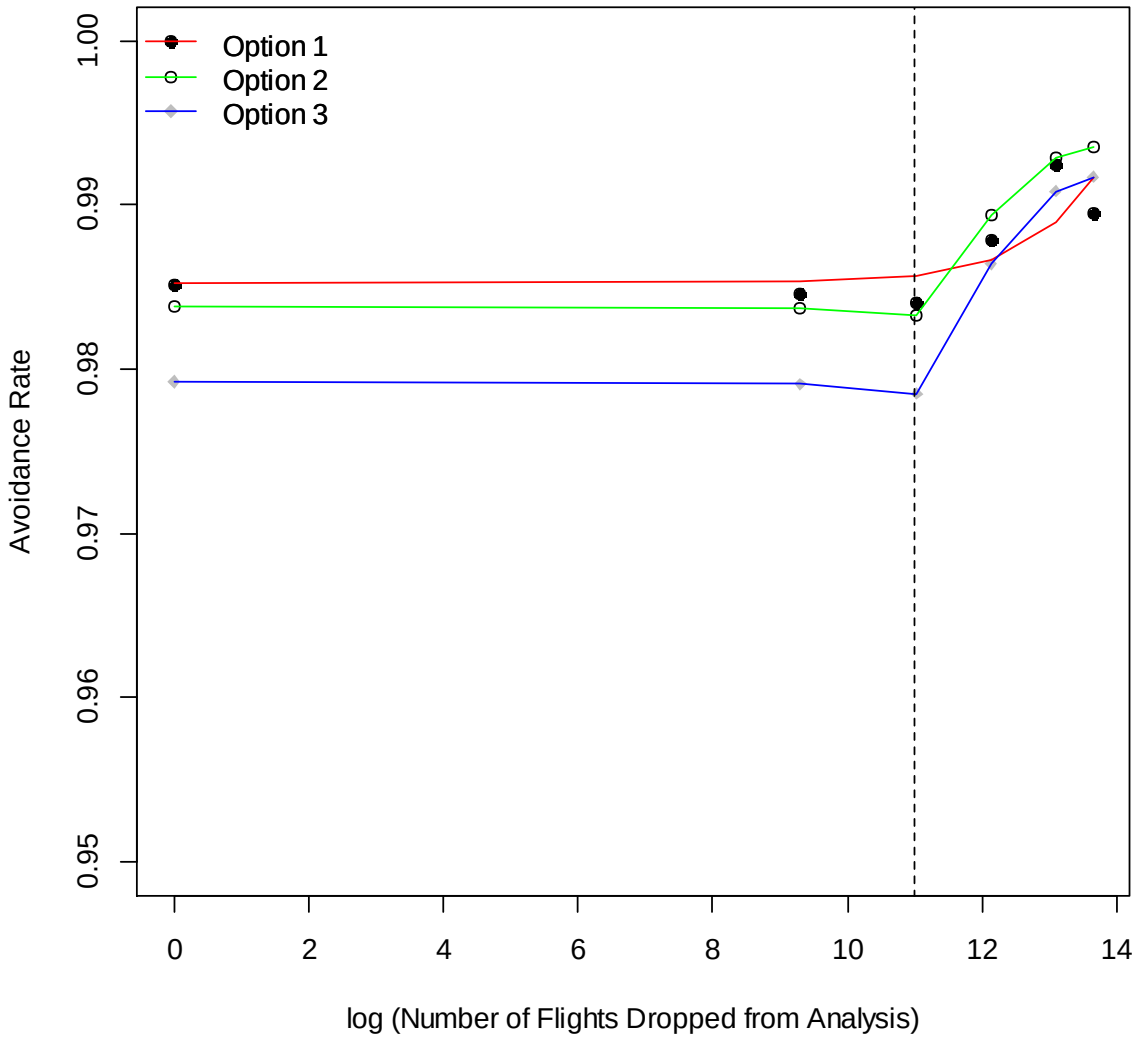


Figure 5.23 Impact of dropping data points (each year-species combination) on the within-windfarm avoidance rates derived using ratio estimators for options 1, 2 and 3 of the Band model for terns.

Within-windfarm avoidance rates derived using all three model options remain relatively stable as the first 660,000 flights through windfarms are dropped from the analysis (Figure 5.23), before increasing as only the species with the highest levels of activity remain. This reflects the fact that Sandwich terns, the species with the highest levels of activity were involved in relatively few collisions, resulting in an

overall increase in the within-windfarm avoidance rates as other species were dropped from the analysis.

We consider that within-windfarm avoidance rates of 0.9851 (±0.0022 SD) for the basic Band model and 0.9792 (±0.0040 SD) for the extended Band model are realistic precautionary within-windfarm avoidance rates given the data available. Whilst we determined that some data points had a high level of leverage on the final values derived, we did not feel that there was sufficient justification for excluding them from our analysis. It should be noted that this leverage occurred in both positive and negative directions. However, as data come from only a single site, it is unclear how transferable they are to novel sites.

6. SENSITIVITY ANALYSIS

Within windfarm avoidance rates can be derived from sites at which estimates of collision rates and bird activity are available using the parameters listed in Table 6.1 and following equation 6 (see section 3.1). However, many of these parameters are incorporated into the calculations as mean values, or a range of values, and others must be estimated. Therefore, in order to understand how transferable these values may be between different models and situations, it is important to understand how sensitive the final avoidance rates are to each of the model input parameters. If avoidance rates are found to be highly sensitive to variation in one or more of the input parameters, it may raise questions about whether or not it is appropriate to apply the avoidance rates derived to novel sites.

For this reason, we assess the sensitivity of each of the avoidance rates presented in Appendix 7 to different input parameters. These parameters include corpse correction factors used to correct for the efficacy of corpse searches (observed collision rate in eq. 6), which will be influenced by scavenger behaviour and searcher efficiency, and estimates of the number of birds passing through a windfarm over a given period of time (flux rate in eq. 6). They also include parameters used to calculate collisions in the absence of avoidance behaviour (P_{coll} in eq. 6) including bird behavioural parameters such as flight speed and altitude, and turbine parameters such as rotor speed and pitch.

Avoidance may also be sensitive to a range of additional factors which cannot be easily quantified. These include time of day, weather, proximity to breeding colonies or overlap with migration routes and the size of the turbines concerned. We use a brief literature review to consider how each of these factors may influence the avoidance rates we derive.

6.1 Avoidance rates derived using the basic Band model (options 1 and 2)

The variables used to estimate P_{coll}, the first step to deriving an avoidance rate, are subject to differing levels of uncertainty. Some, such as rotor diameter, blade width and turbine height are fixed and are, therefore, known quantities with very little, if any, uncertainty surrounding them. Others, such as rotor speed and pitch and aspects of bird behaviour, such as flight speed and altitude and the propensity to fly upwind or downwind are subject to a greater degree of uncertainty. As part of the sensitivity analysis, we focus on the parameters which are not fixed and, therefore, subject to varying degrees of uncertainty, in order to determine what influence the inaccurate estimation of each of these parameters has on the final derived avoidance rates. Whilst the focus of much of the interest in collision risk modelling has been on avoidance rates, it is actually 1-avoidance rate, or the non-avoidance rate which is applied in the final step of the Band collision risk model (Band *pers. comm.*, Masden *et al.* in prep). For this reason, we focus our sensitivity analysis on this factor, rather than the avoidance rate.

For each of the sites and species combinations presented in Appendix 7 at which collisions were recorded, we consider the impact that a 10% increase (following Chamberlain *et al.* 2006) in each of rotor speed, rotor pitch, bird flight speed, flux rate and the proportion of flights upwind would have on the avoidance rates derived

using option 1. In addition, we also consider the influence of a 10% increase in corpse detection rate.

Table 6.1 Input parameters for the Band (basic and extended model)

Parameter	Sensitivity assessed
Species name	No
Bird length	No
Wingspan	No
Flight speed	Yes – 10% increase considered following Chamberlain *et al.* (2006)
Nocturnal activity factor (1-5)	Considered as part of increase in flux rate
Flight type, flapping or gliding	No – Seabirds most likely to engage in flapping flight, which is the higher risk activity
Daytime bird density	Yes – considered as part of increase in flux rate
Proportion at rotor height	Yes – 10% increase in birds flying at risk height considered for basic model, 200 randomly simulated distributions considered for the extended model
Proportion of flights upwind	Yes – 10% increase in the proportion of birds flying upwind considered
Name of windfarm site	No – Fixed parameter
Latitude	No – Fixed parameter
Number of turbines	No – Fixed parameter
Width of windfarm	No – Fixed parameter
Tidal offset	No – Suitable datasets were only available for onshore windfarms
Turbine model	No – Fixed parameter
No. of blades	No – Fixed parameter
Mean rotation speed	Yes – 10% increase considered following Chamberlain *et al.* (2006)
Rotor radius	No – Fixed parameter
Hub height	No – Fixed parameter
Monthly proportion of time operational	Yes – considered as part of increase in flux rate
Max blade width	No – Fixed parameter
Pitch	Yes – 10% increase considered following Chamberlain *et al.* (2006)

6.1.1 Sensitivity to the assumed flux rate at the windfarm

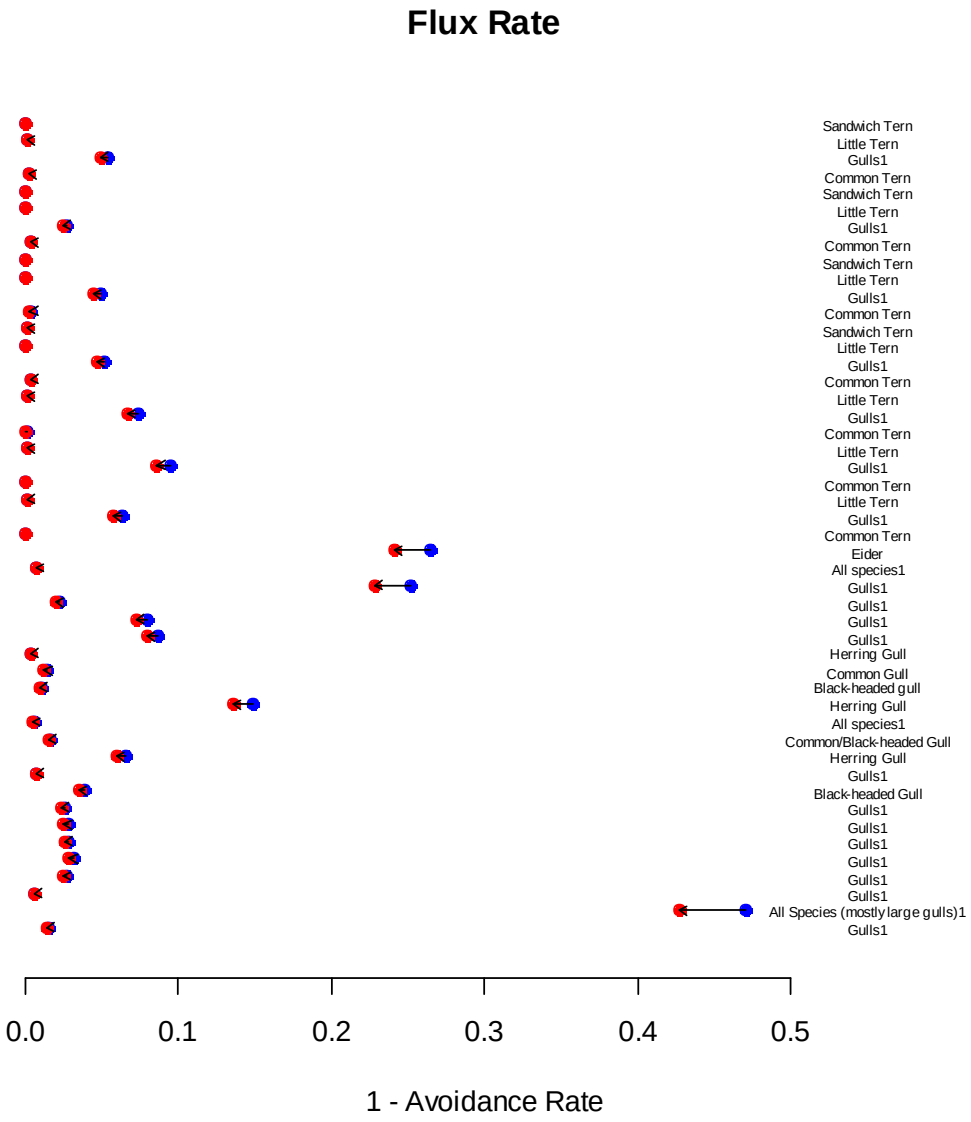


Figure 6.1 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the basic Band model, to the assumed flux rate at each site. Blue dots indicate the non-avoidance rate derived assuming the flux rate presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the flux rate at each site.

Bird flux rate is an estimate of the total number of birds passing through the windfarm when it is operational. As such, it combines estimates of the number of birds recorded within the windfarm, the proportion of birds at collision risk height,

corrections for nocturnal activity and an estimate of the monthly proportion of time it is operational. An increase in the flux rate derived at each site results in a decrease in the derived non-avoidance rates (Figure 6.1). This is because, whilst the observed number of collisions remains constant, the number of birds passing through the windfarm increases, meaning that a greater proportion of them are assumed to have avoided collision. These changes are approximately inversely proportional to the increase in the numbers of birds passing through the site. A comparison of the mean non-avoidance rates based on the flux rate presented in Appendix 7, with the mean non-avoidance rates assuming a 10% increase in this flux rate suggests that such an increase may result in a 9.1% decrease in the non-avoidance rate.

6.1.2 Sensitivity to the corpse detection rate at the windfarm

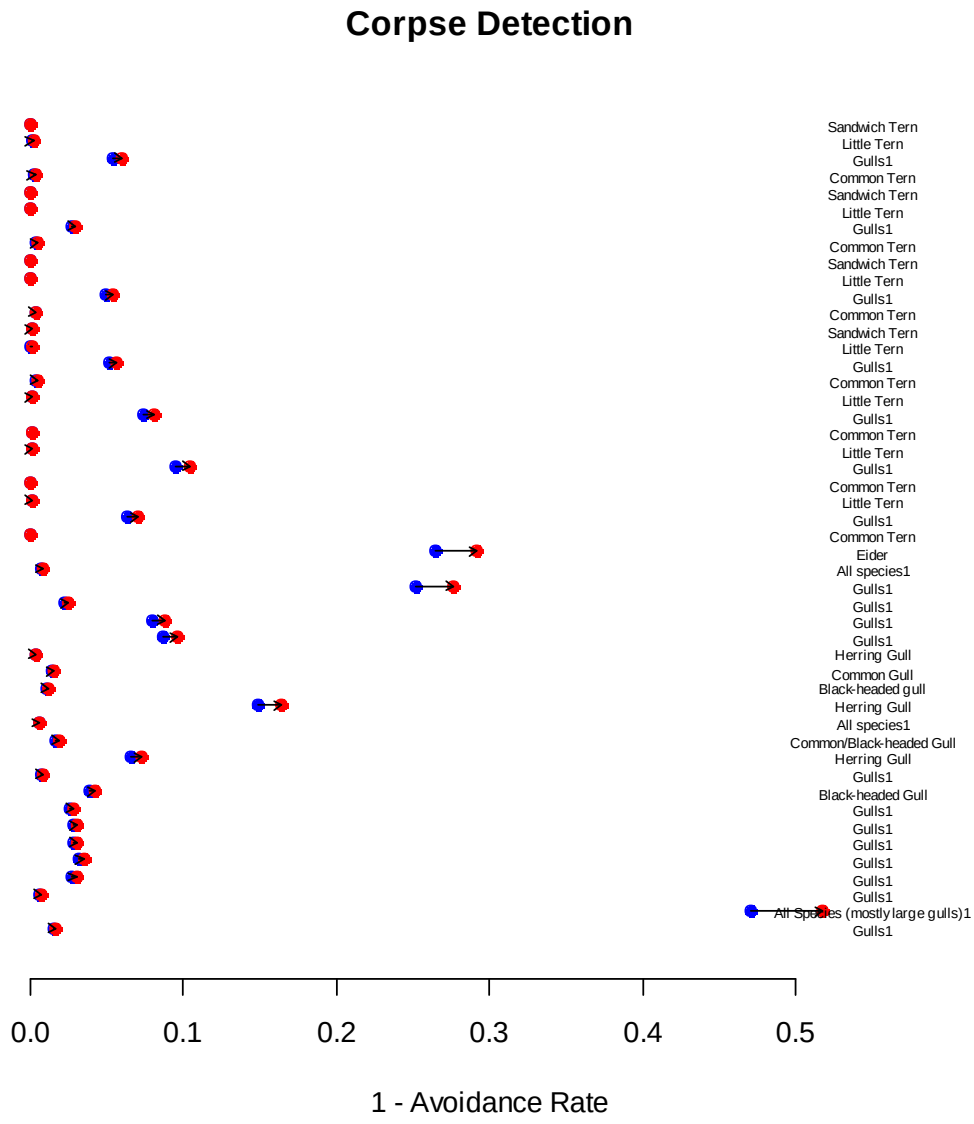


Figure 6.2 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the basic Band model, to the assumed corpse detection rate at each site. Blue dots indicate the non-avoidance rate derived assuming the

number of collisions presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the number of collisions detected at each site.

During the search for collision victims, corpses may be missed either as a result of searcher inefficiency, or through the removal of carcasses by predators (Winkelman 1992). As a result it is often necessary to correct observed collision rates to account for these missing corpses. Assuming an increase in the total number of victims leads to an increase in the derived non-avoidance rate because the total number of birds passing through the windfarm remains constant and it is assumed a higher proportion of them collide with the turbines. These increases in the non-avoidance rate are proportional with the increase in corpse detection (Figure 6.2), with a 10% correction in the number of collisions to account for a failure to detect corpses resulting in 10% increase in the non-avoidance rate.

6.1.3 Sensitivity to the proportion of birds flying upwind

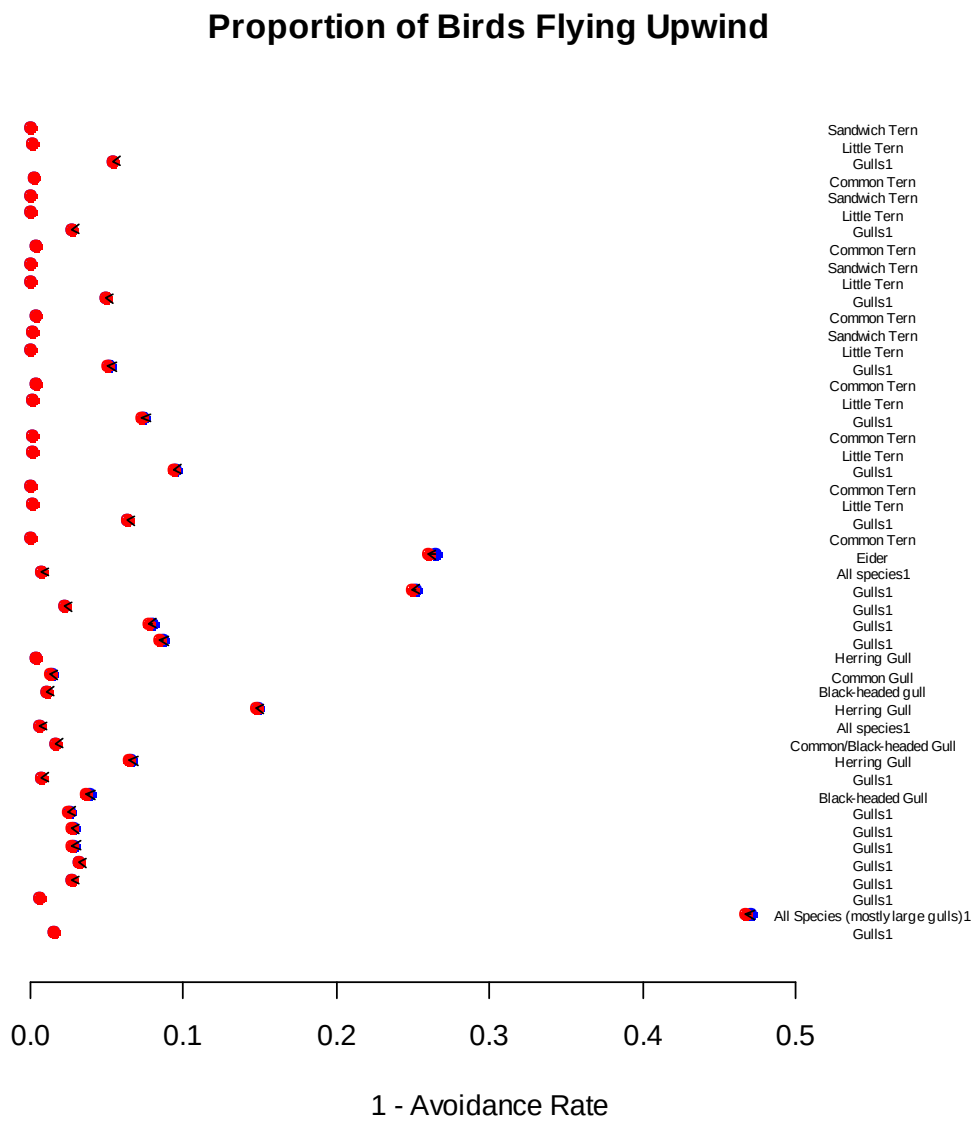


Figure 6.3 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the basic Band model, to the proportion of birds flying upwind at each site. Blue dots indicate the non-avoidance rate derived assuming the number of collisions presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the proportion of birds flying upwind detected at each site.

A 10% change to the proportion of birds flying upwind resulted in a small decrease in the derived non-avoidance rates (Figure 6.3) of 1.17%. These results suggest that the proportion of birds estimated to fly up or downwind has a relatively small effect on the final, derived non-avoidance rate.

6.1.4 Sensitivity to the mean turbine rotor speed

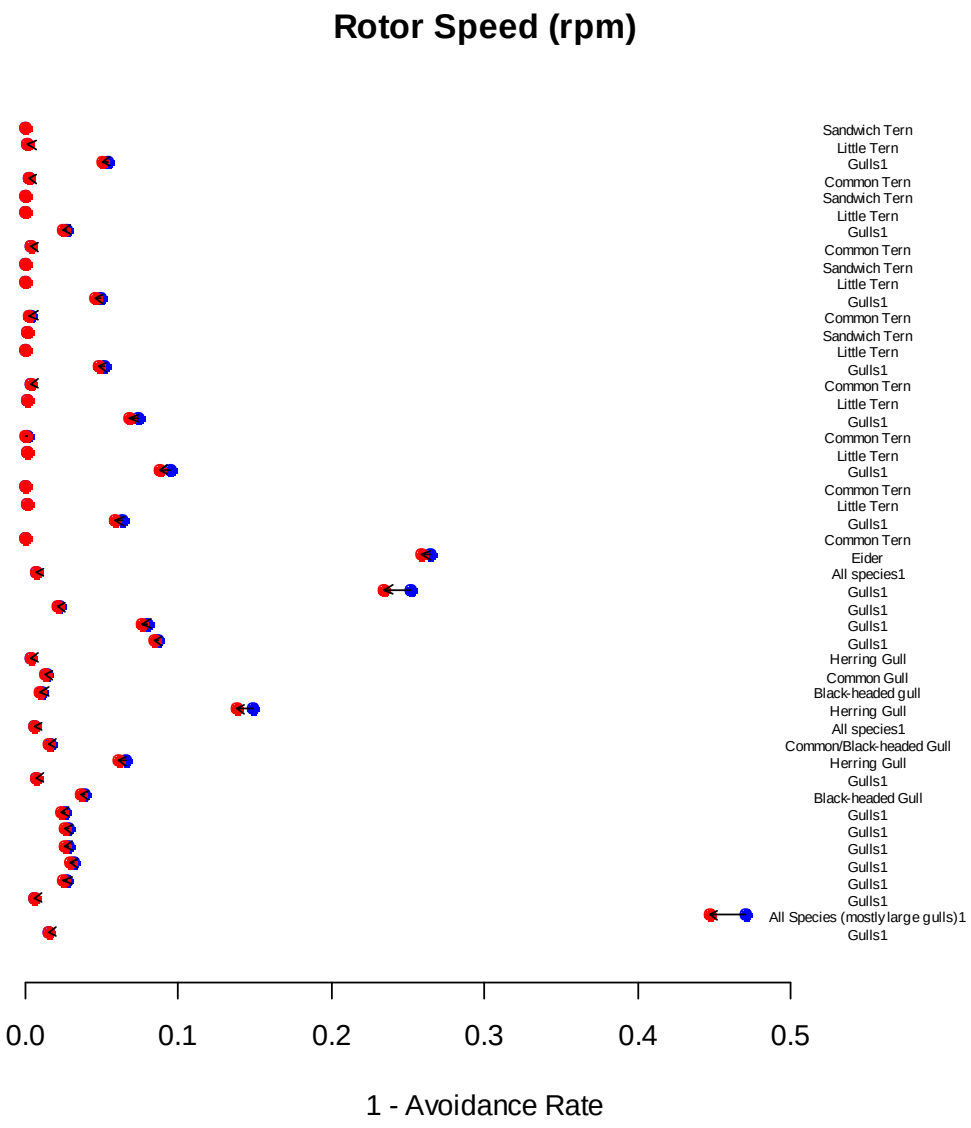


Figure 6.4 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the

basic Band model, to the turbine rotor speed at each site. Blue dots indicate the non-avoidance rate derived assuming the number of collisions presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the turbine rotor speed at each site.

A 10% increase in the mean turbine rotor speed assumed typically resulted in a decrease in the derived non-avoidance rates of approximately 5.5% (Figure 6.4). The reason for this decrease is that as the rotor speed increases, the time available for a bird to pass through unharmed decreases, meaning that the predicted collision rate increases whilst the recorded number of collisions remains constant. Based on the turbines we considered, a 10% increase in mean rotor speed reflects an increase of between 1 and 4 rotations per minute. Published data from turbine manufacturers (http://www.4coffshore.com/) suggests the range of operational speeds for turbines is like to vary by between 5 and 15 rpm. As such, the increase in rotation speed we consider may be somewhat conservative but, without more detailed curves showing the range of operational speeds used by different turbines, assessing this is difficult.

6.1.5 Sensitivity to the turbine pitch

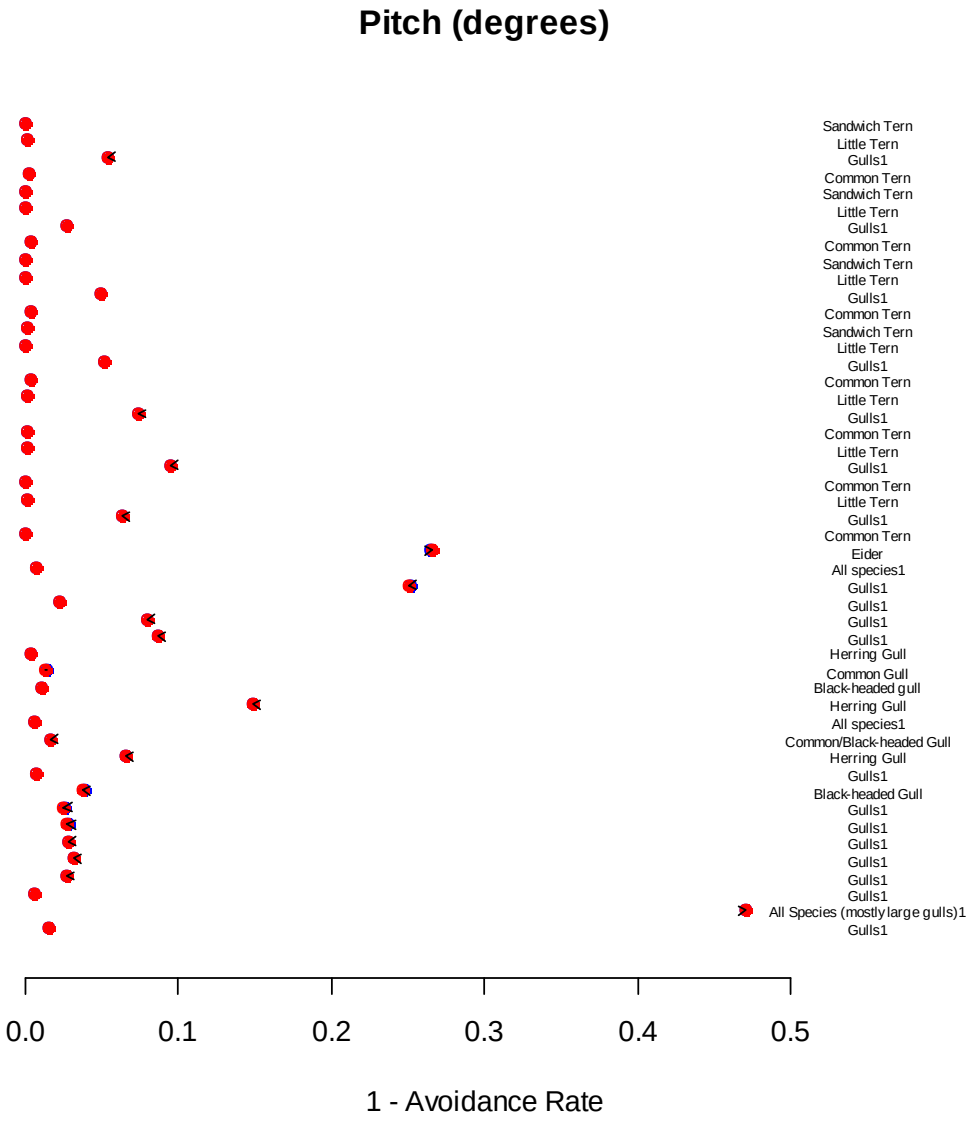


Figure 6.5 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the basic Band model, to the turbine pitch at each site. Blue dots indicate the non-avoidance rate derived assuming the number of collisions presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the turbine pitch at each site.

A 10% change in the assumed turbine pitch resulted in a fairly negligible decrease in the derived non-avoidance rates (Figure 6.5) of 0.2%. Our calculations were based on an assumption of a 10˚ pitch for each turbine, so a 10% increase reflects an 11˚ pitch. Available data describing the pitch of operational turbines are extremely limited. As a consequence, it is not possible to determine how well these values reflect reality at operational turbines.

6.1.6 Sensitivity to the bird flight speed

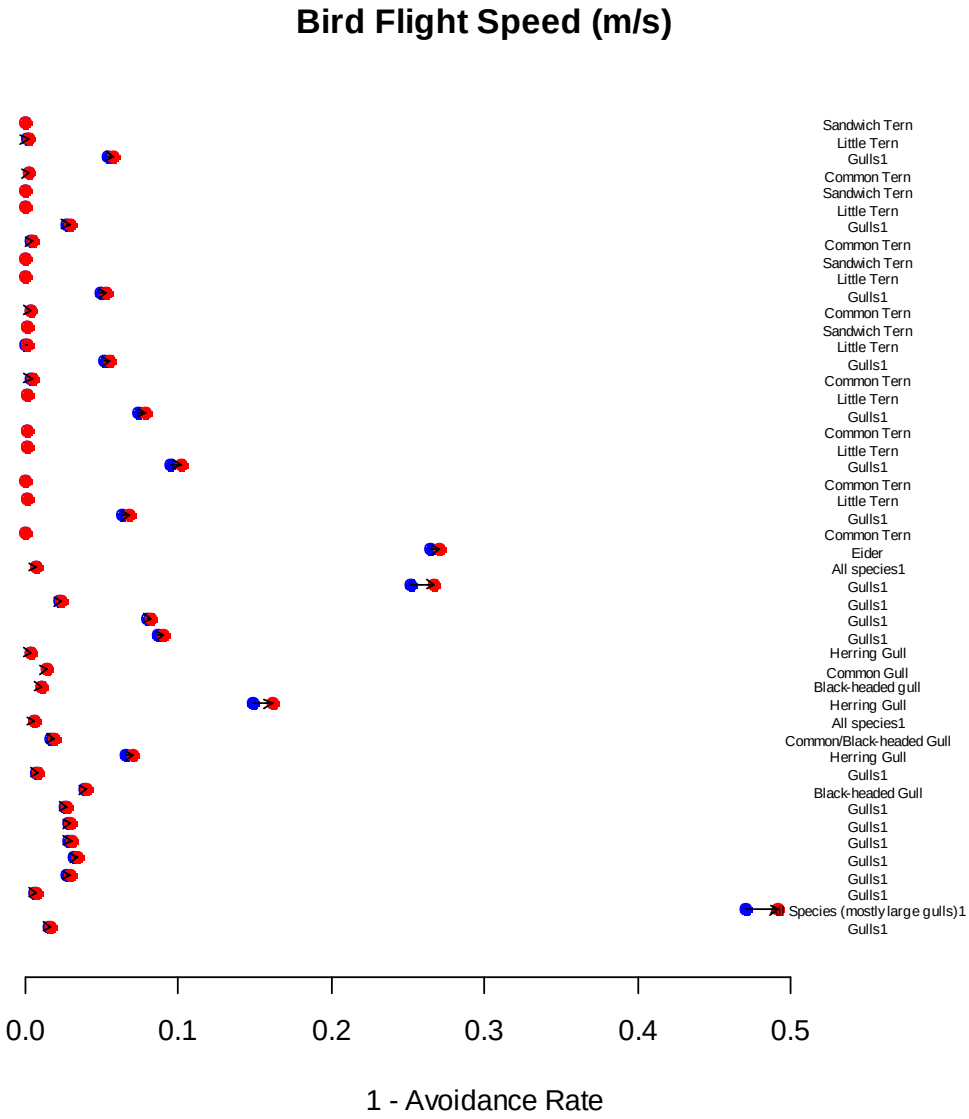


Figure 6.6 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the basic Band model, to the bird flight speed at each site. Blue dots indicate the non-avoidance rate derived assuming the number of collisions presented in Appendix 7, red dots indicate the avoidance rate derived assuming a 10% increase in the bird flight speed at each site.

A 10% increase in the assumed bird flight speed resulted in an increase in the derived non-avoidance rates (Figure 6.6) of 5.5%. This increase reflects the fact that the faster a bird passes through the rotor swept-area, the less likely it is to be hit. As a result an increase in flight speed results in a decrease in the predicted number of collisions whilst the observed number of collisions remains constant. For our study species a 10% increase in flight speed reflects an increase of 1-1.3 m/s. Alerstam *et al.* (2007) suggest that the standard deviations around the mean flight speeds for our study species are in the region of 1-2 m/s, suggesting that a 10% increase in flight speed may be a realistic, precautionary assumption.

6.1.7 Basic Band model sensitivity conclusions

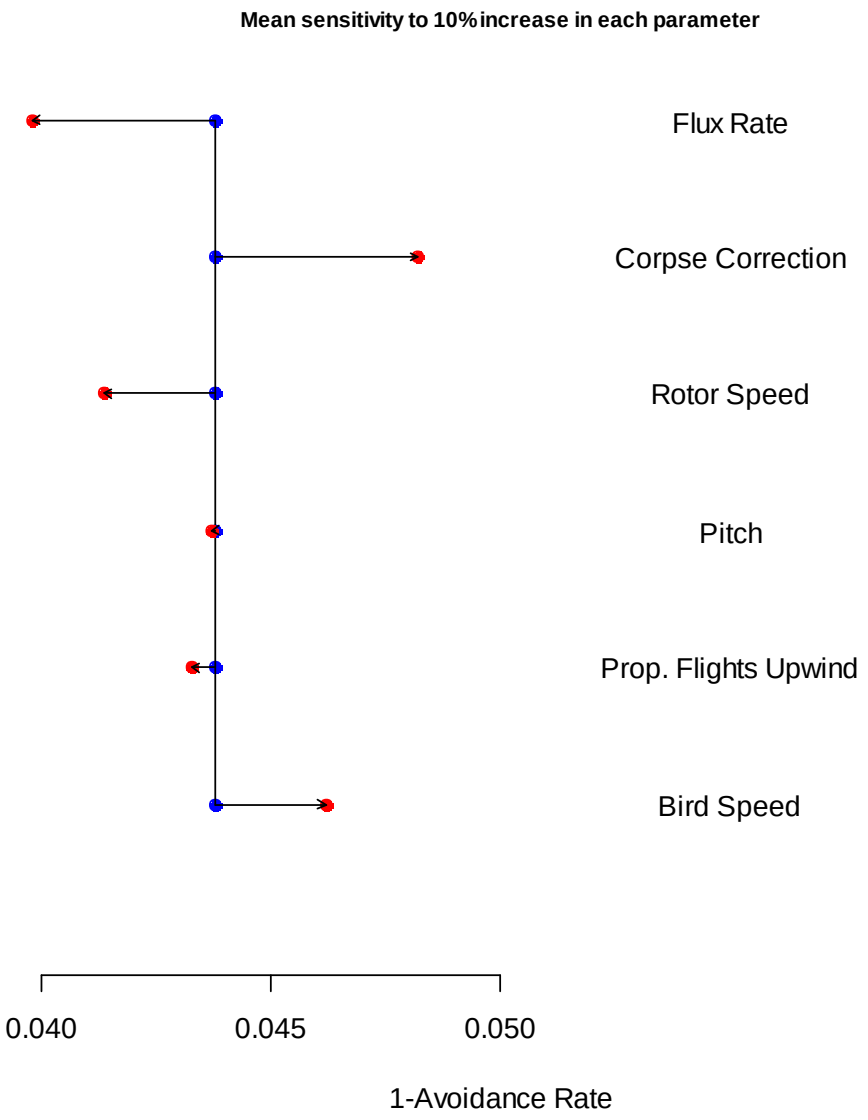


Figure 6.7 Sensitivity of derived non-avoidance rates derived using the basic Band model to a 10% increase in each of the Band model parameters.

Of the parameters considered, the final derived non-avoidance rates were most sensitive to flux rate and the corpse correction (Figure 6.7). An increase in the flux rate meant that the predicted collision rate increased, whilst the observed collision rate remained constant (see eq. 6, section 3.1); as a consequence, the non-avoidance rate decreased in response to an increase in the flux rate. For similar reasons, an increase in the number of corpses detected resulted in an increase in the non-avoidance rate derived. The impacts of assumed rotor speed and bird speed on the derived non-avoidance rates were of a similar magnitude, but in opposite directions. An increased assumed rotor speed results in a decreased non-avoidance rate because faster turbines result in an increased risk of collision. As a consequence, a faster rotor speed would result in an increase in the predicted collision rate, whilst the observed collision rate remains constant. This results in a decrease in the non-avoidance rate. In contrast, an increase in the assumed speed

of the birds passing through the rotor swept area of a turbine decreases the risk of collision. As a consequence, the predicted collision rate decreases and, for the reasons stated above, the non-avoidance rate derived increases. Whilst increases in both the assumed pitch and the proportion of flights upwind resulted in decreases in the derived non-avoidance rates, the impact of both parameters was negligible.

6.2 Avoidance rates derived using the extended Band model (option 3)

In addition to the variables described above (section 6.1), non-avoidance rates derived using the extended Band model are also likely to be sensitive to the assumed flight height distributions. Collision risk is not evenly distributed within the rotor swept area of turbines, and is greatest towards the centre of the rotor disk. The extended Band model makes use of flight height distributions, such as those derived by Johnston *et al.* (2014a) to account for this variable risk. However, as these are continuous distributions, it is not appropriate to simply assume, for example, that an additional 10% of birds fly at rotor height as this will have implications for the overall shape of the distribution. Therefore, in addition to the parameters considered for the basic Band model, for each species/site combination we consider, we use 200 random distributions estimated following the methodology of Johnston *et al.* (2014a) to investigate sensitivity to the assumed distribution (Figure 6.8). It is important to note that by comparing between different distributions, the outputs of the sensitivity analysis will not be strictly comparable to the outputs of the sensitivity analyses described above.

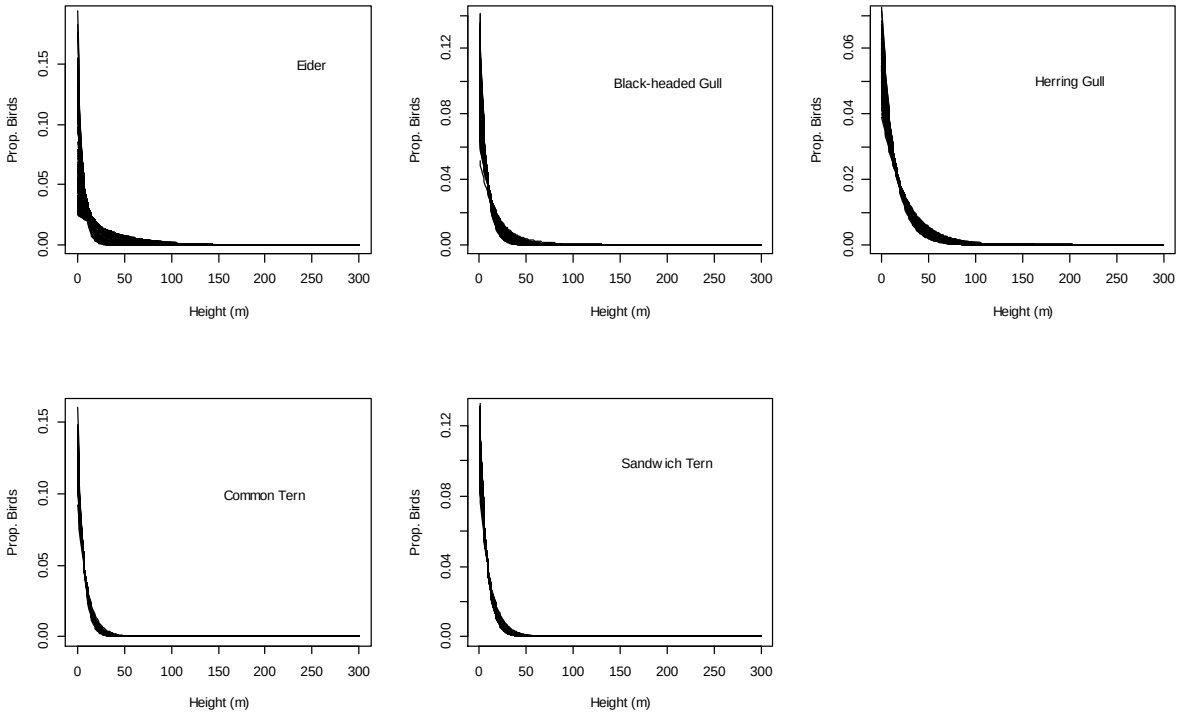


Figure 6.8 200 Random flight height distributions estimated for each of eider, black-headed gull, herring gull, common tern and Sandwich tern, species for which avoidance rates could be derived from a combination of recorded collisions and recorded levels of bird activity, using the methodology set out in Johnston *et al.* (2014a) and used to assess the sensitivity of derived avoidance rates to the assumed flight height distribution.

6.2.1 Sensitivity to assumed flight height distribution

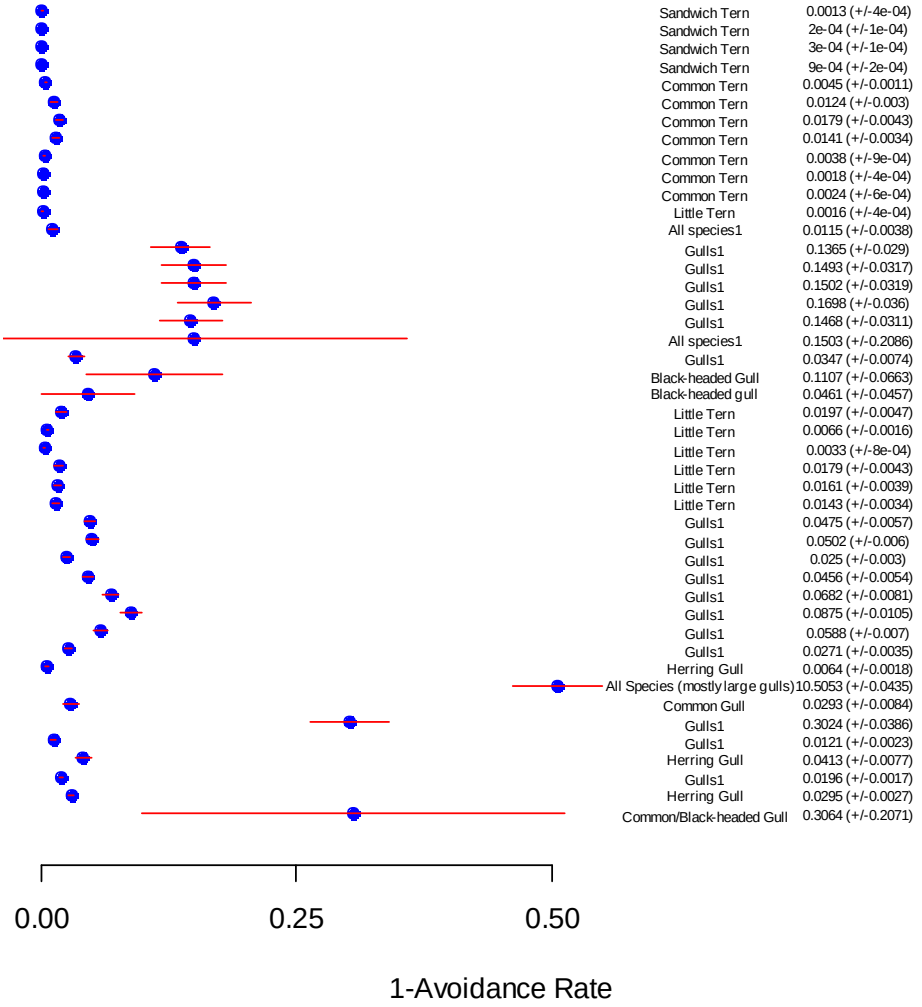


Figure 6.9 Sensitivity of non-avoidance rates derived for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the assumed flight height distribution at the site. Blue dots indicate the mean non-avoidance rate values derived from 200 random flight height distributions at each site, red lines indicate the standard deviation around these values, actual values shown alongside plot.

The sensitivity of the derived non-avoidance rates to different flight height distributions appears to be highly variable (Figure 6.9). The greatest sensitivity appears to occur where derived non-avoidance rates are highest. This relationship is likely to reflect the level of activity at any given site. For example, consider two sites, at the first of which 1 flight out of 100 at rotor height results in a collision and at the second of which 1 flight out of 1000 results in a collision. If the estimate of the proportion of birds flying at rotor height increases at each site by 10%, whilst the recorded number of collisions remains constant, this becomes 1 flight out of 110 at

the first site and 1 flight out of 1,100 at the second. At the first site the non-avoidance rate decreases from 0.0100 to 0.0091, whilst at the second it decreases from 0.0010 to 0.0009. The overall decrease is therefore greater at the first site, with the lower level of flight activity.

6.2.2 Sensitivity to the assumed flux rate at the windfarm

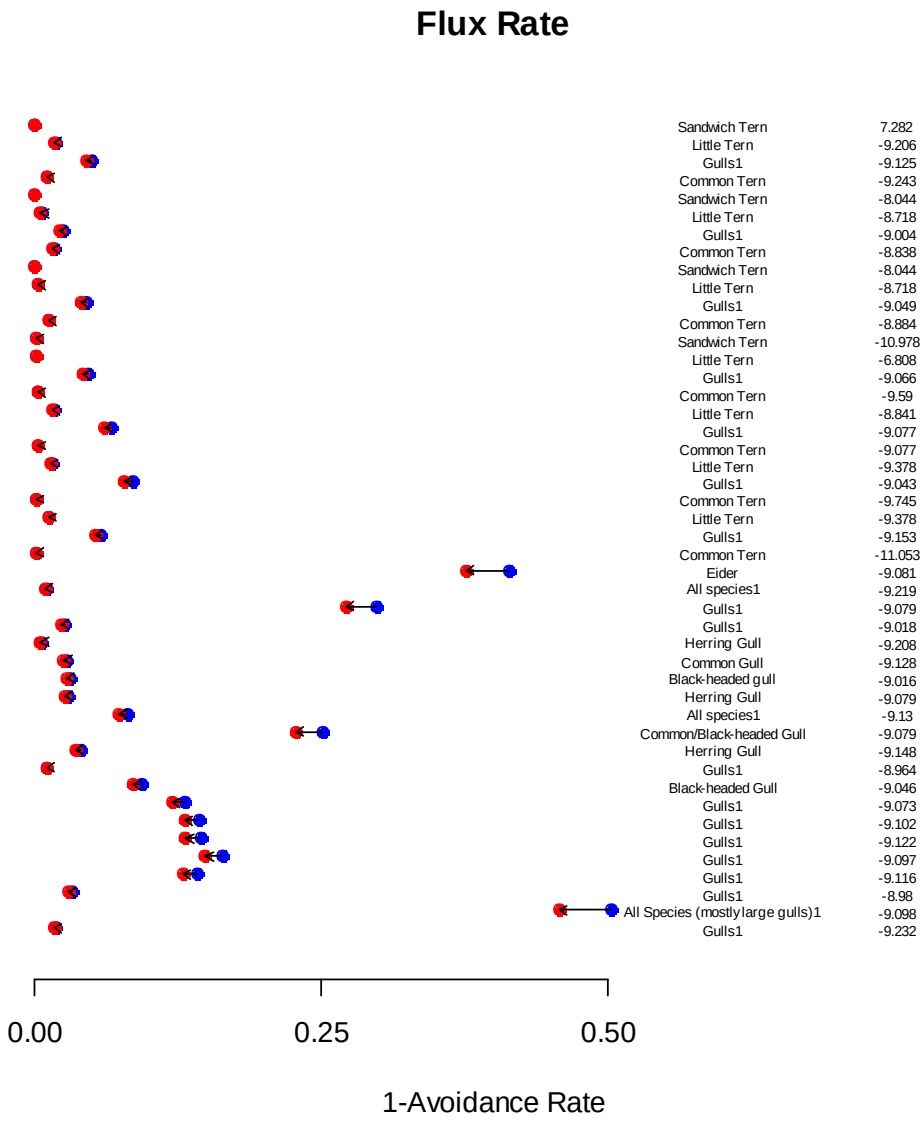


Figure 6.10 Sensitivity of non-avoidance rates derived for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the assumed flux rate at each site. Blue dots indicate the non-avoidance rate derived assuming the flux rate presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the flux rate at each site. % change in the non-avoidance rates following a 10% increase in flux rate shown alongside graph.

Bird flux rate is an estimate of the total number of bird passing through the windfarm when it is operational. As such, it combines estimates of the number of birds

recorded within the windfarm, corrections for nocturnal activity and an estimate of the monthly proportion of time it is operational. An increase in the flux rate derived at each site results in a decrease in the derived non-avoidance rates (Figure 6.10). This is because, whilst the observed number of collisions remains constant, the number of birds passing through the windfarm increases, meaning that a greater proportion of them are assumed to have avoided collision. These decreases are roughly inversely proportional to the increase in flux rate, although in contrast to the case of the basic Band model, this value will vary across sites as a consequence of the different height distributions assumed. A comparison of the mean avoidance rates based on the flux rate presented in Appendix 7, with the mean avoidance rates assuming a 10% increase in this flux rate suggests that such an increase may result in a mean 8.73% decrease in the non-avoidance rate.

6.2.3 Sensitivity to the corpse detection rate at the windfarm

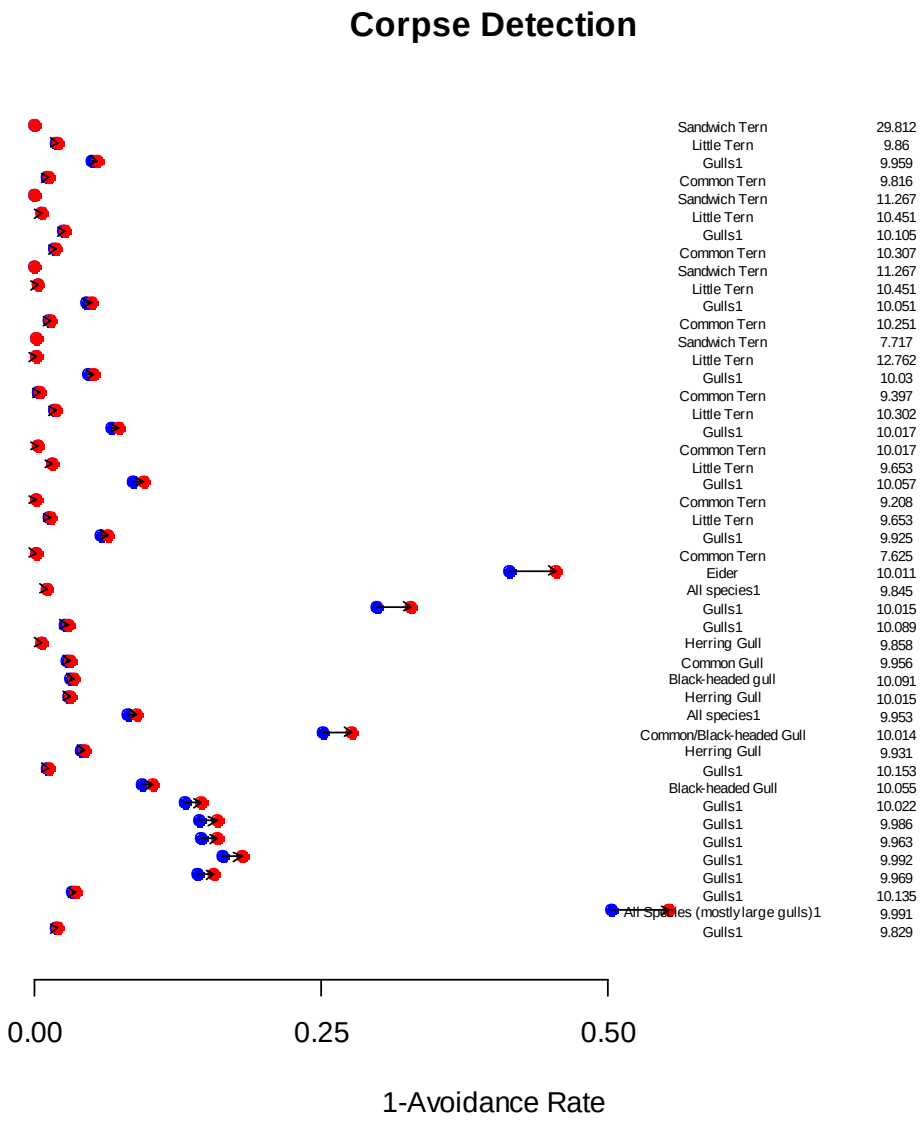


Figure 6.11 Sensitivity of non-avoidance rates for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the assumed corpse detection rate at each

site. Blue dots indicate the non-avoidance rate derived assuming the number of collisions presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the number of collisions detected at each site. % change in the non-avoidance rates following a 10% increase in the number of collisions detected shown alongside graph.

During the search for collision victims, corpses may be missed either as a result of searcher inefficiency, or through the removal of carcasses by predators (Winkelman 1992). As a result it is often necessary to correct observed collision rates to account for these missing corpses. Assuming an increase in the total number of victims leads to an increase in the derived non-avoidance rate because the total number of birds passing through the windfarm remains constant and it is assumed a higher proportion of them collide with the turbines (Figure 6.11). This increase is broadly proportional with the increase in the flux rate across sites, with a mean 10.43% increase in the non-avoidance rate following a 10% increase in the flux rate.

6.2.4 Sensitivity to the proportion of birds flying upwind

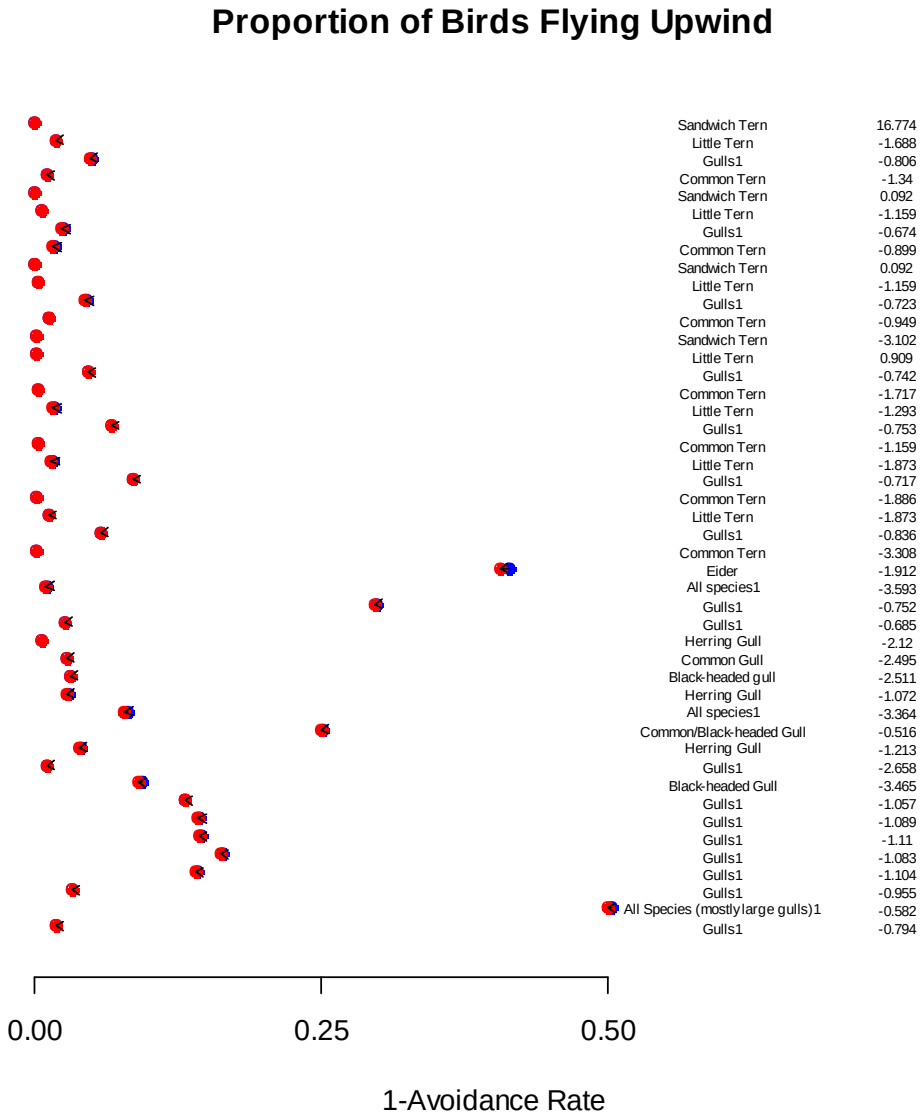


Figure 6.12 Sensitivity of non-avoidance rates derived for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the proportion of birds flying upwind at each site. Blue dots indicate the non-avoidance rate derived assuming 50% of birds flying upwind, red dots indicate the avoidance rate derived assuming a 10% increase in the proportion of birds flying upwind at each site. % change in the non-avoidance rates following a 10% increase in the proportion of birds flying upwind shown alongside graph.

A 10% change to the proportion of birds flying upwind resulted in a fairly negligible decrease in the derived avoidance rates (Figure 6.12). The % increases were typically <1%, and across all sites a 10% increase in the proportion of birds flying upwind resulted in a decrease in the non-avoidance rate of approximately 0.97%. These results suggest that the proportion of birds estimated to fly up or downwind has a negligible effect on the final, derived non-avoidance rate.

6.2.5 Sensitivity to the turbine rotor speed

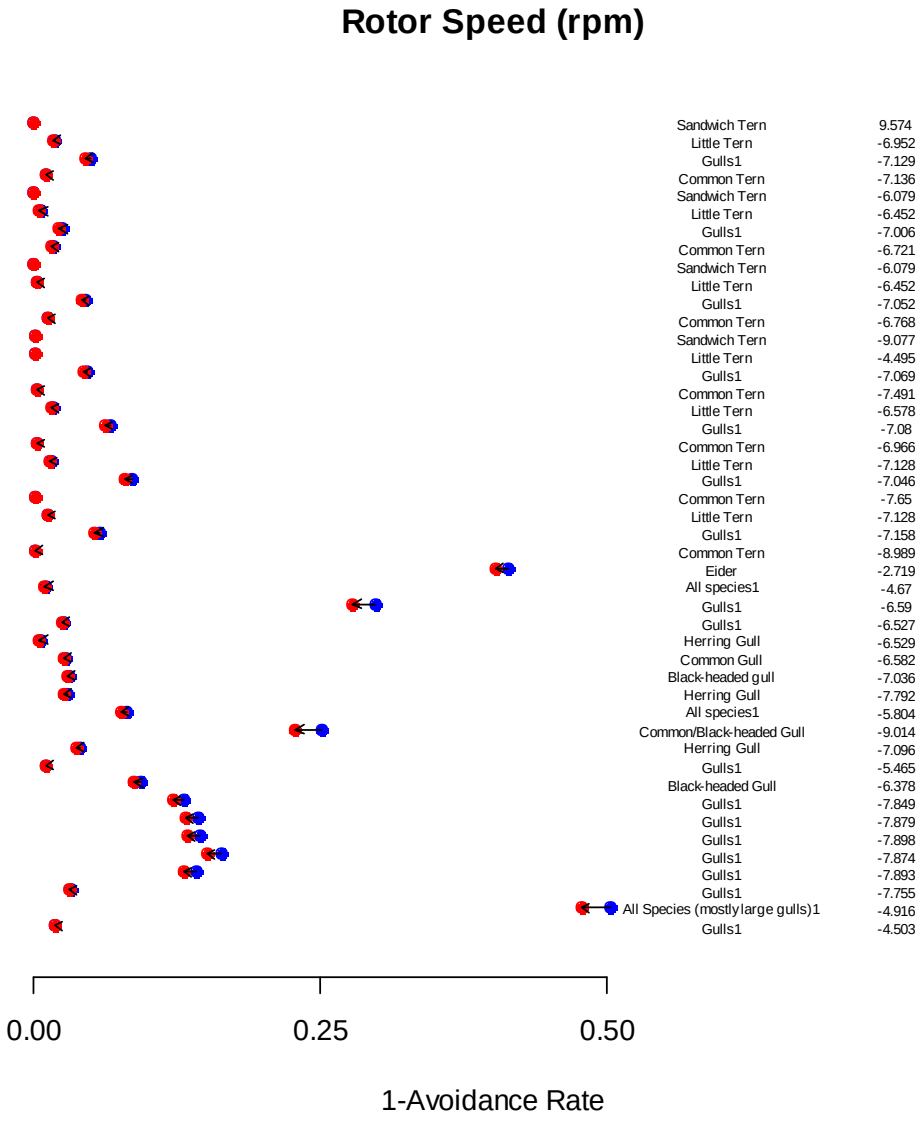


Figure 6.13 Sensitivity of non-avoidance rates derived for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the turbine rotor speed. Blue dots indicate the non-avoidance rate derived based on the rotor speed values presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in these rotor speeds. % change in the non-avoidance rates following a 10% increase in the turbine rotor speed shown alongside graph.

A 10% increase in the assumed turbine rotor speed typically resulted in a decrease in the derived non-avoidance rates (Figure 6.13). Across all sites a 10% increase in the rotor speed resulted in a decrease in the non-avoidance rate of approximately 6.45%.

6.2.6 Sensitivity to the turbine pitch

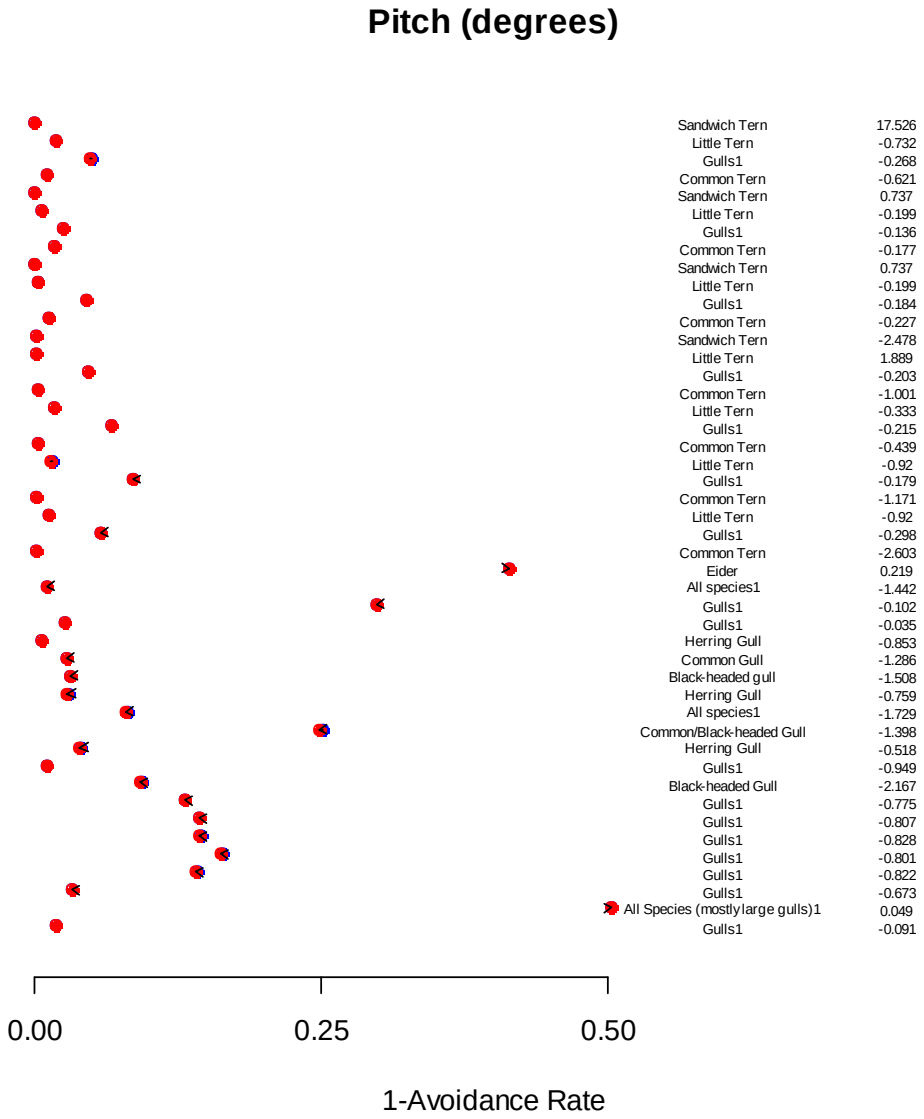


Figure 6.14 Sensitivity of non-avoidance rates derived for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the turbine pitch. Blue dots indicate the non-avoidance rate derived based on the rotor speed values presented in Appendix 7, red dots indicate the non-avoidance rate derived assuming a 10% increase in the pitch. % change in the non-avoidance rates following a 10% increase in the turbine rotor speed shown alongside graph.

A 10% change in the assumed turbine pitch resulted in a fairly negligible decrease in the derived non-avoidance rates (Figure 6.14). The % decreases were typically <1%, and across all sites a 10% increase in the turbine pitch resulted in a decrease in the non-avoidance rate of approximately 0.21%.

6.2.7 Sensitivity to the bird flight speed

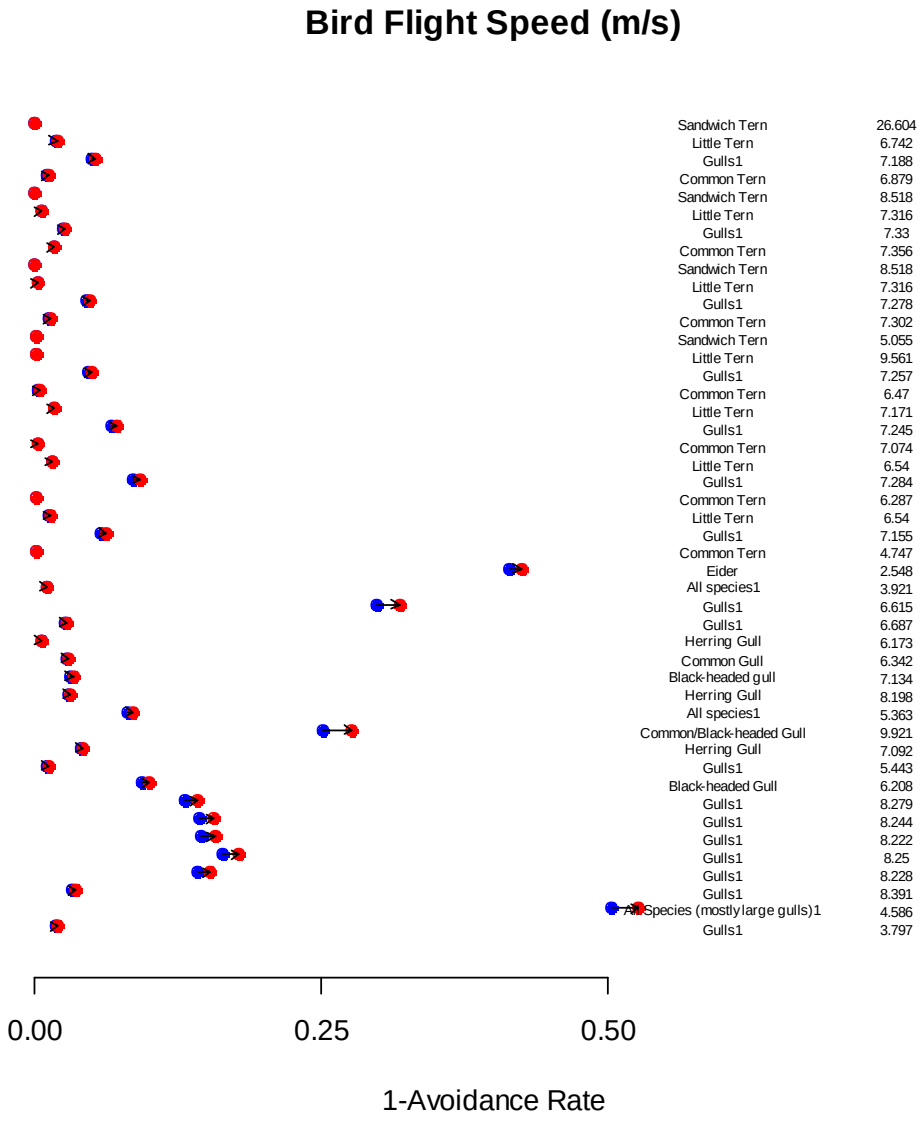


Figure 6.15 Sensitivity of non-avoidance rates derived for each species and site in Appendix 7 at which a collision was recorded and derived using the extended Band model, to the bird flight speed. Blue dots indicate the non-avoidance rate derived based on the bird flight speed values presented in Table 5.4, red dots indicate the non-avoidance rate derived assuming a 10% increase in the bird flight speed. % change in the non-avoidance rates following a 10% increase in the bird flight speed shown alongside graph.

A 10% increase in the assumed bird flight speed typically resulted in an increase in the derived non-avoidance rates (Figure 6.15). Across all sites a 10% increase in the bird flight speed resulted in an increase in the non-avoidance rate of approximately 7.31%.

6.2.8 Extended Band model sensitivity conclusions

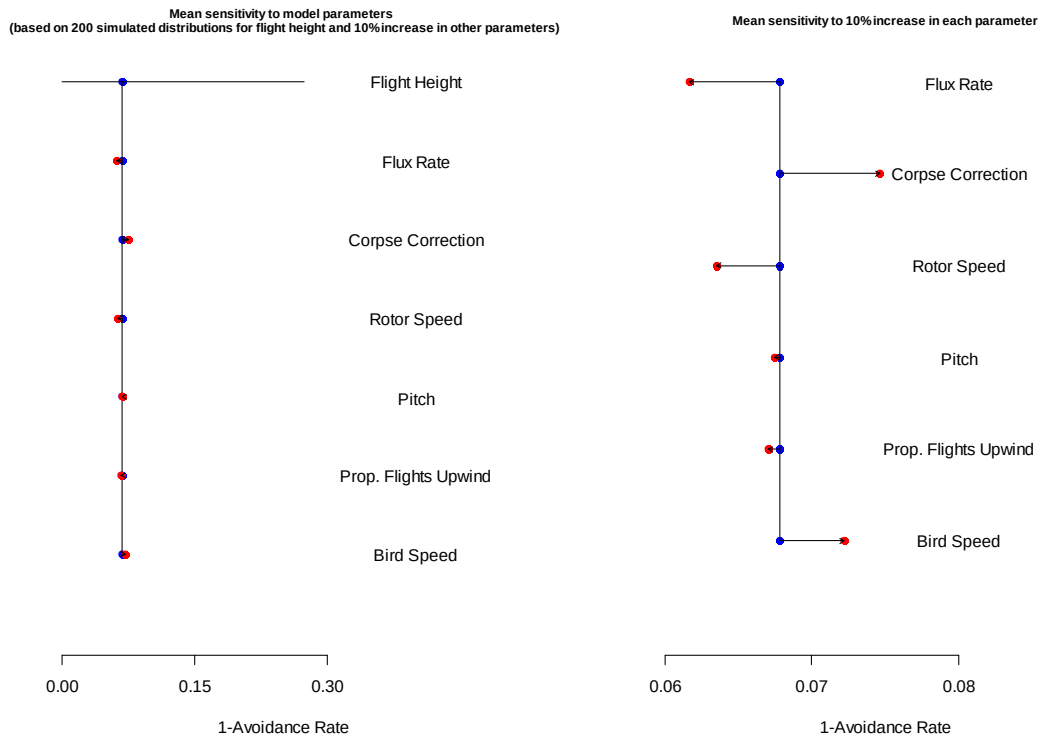


Figure 6.16 Sensitivity of non-avoidance rates derived using the extended Band model to Band model parameters. Sensitivity to flight height distribution is assessed by considering the standard deviation calculated from non-avoidance rates derived using 200 randomly simulated flight height distributions and sensitivity to the remaining parameters is derived from a 10% increase in the values presented in Appendix 7 and Table 5.4.

Of the parameters considered, the derived non-avoidance rates appear to be most sensitive to the assumed flight height distribution (Figure 6.16). However, the assessment of sensitivity for this parameter is not strictly comparable to that for the other parameters as it is not possible to make a simple assumption about a change in a continuous distribution in the same way it is about a change in, for example, rotor speed or bird numbers. Furthermore, the magnitude of the sensitivity in this parameter may be strongly influenced by 11 of the 45 data points, for which there was particularly high variation around the mean values (Figure 6.9). On closer examination, this variation appears to be strongly linked to sites with relatively low levels of bird activity (Figure 6.17).

Of the remaining parameters, the derived non-avoidance rates were most sensitive to changes in the flux rate at the windfarm (the number of birds passing through over the course of the study period) and the accuracy with which corpses were detected.

Both rotor speed and bird speed also appeared to have a moderate influence on the derived non-avoidance rates (Figure 6.18). The sensitivity of the non-avoidance rates to the input parameters appeared to be relatively consistent between option 1 and option 3.

Sensitivity to each parameter also appeared to be strongly linked to the number of birds estimated flying through each monitored windfarm (Figures 6.10 and 6.17). As the number of birds passing through a site increases, the sensitivity of the derived non-avoidance rates to each of the model parameters, including the assumed flight height distribution, drops markedly. This finding is consistent with that of Douglas *et al.* (2012) who found that the sensitivity of predicted collision rates to input parameters dropped as the quantity of observational data increased. In the case of sensitivity to the assumed flight height distributions used, at sites where flight activity is greatest, the derived avoidance rates have a similar level of variability to this and to other parameters. This is because for two sites where similar numbers of collisions are recorded, but at which the levels of bird activity differ, the non-avoidance rate will be higher at the site with the lowest level of bird activity. As a consequence, where an identical change occurs at both sites, the total change in the non-avoidance rate will be greatest at the site with the lowest level of bird activity.

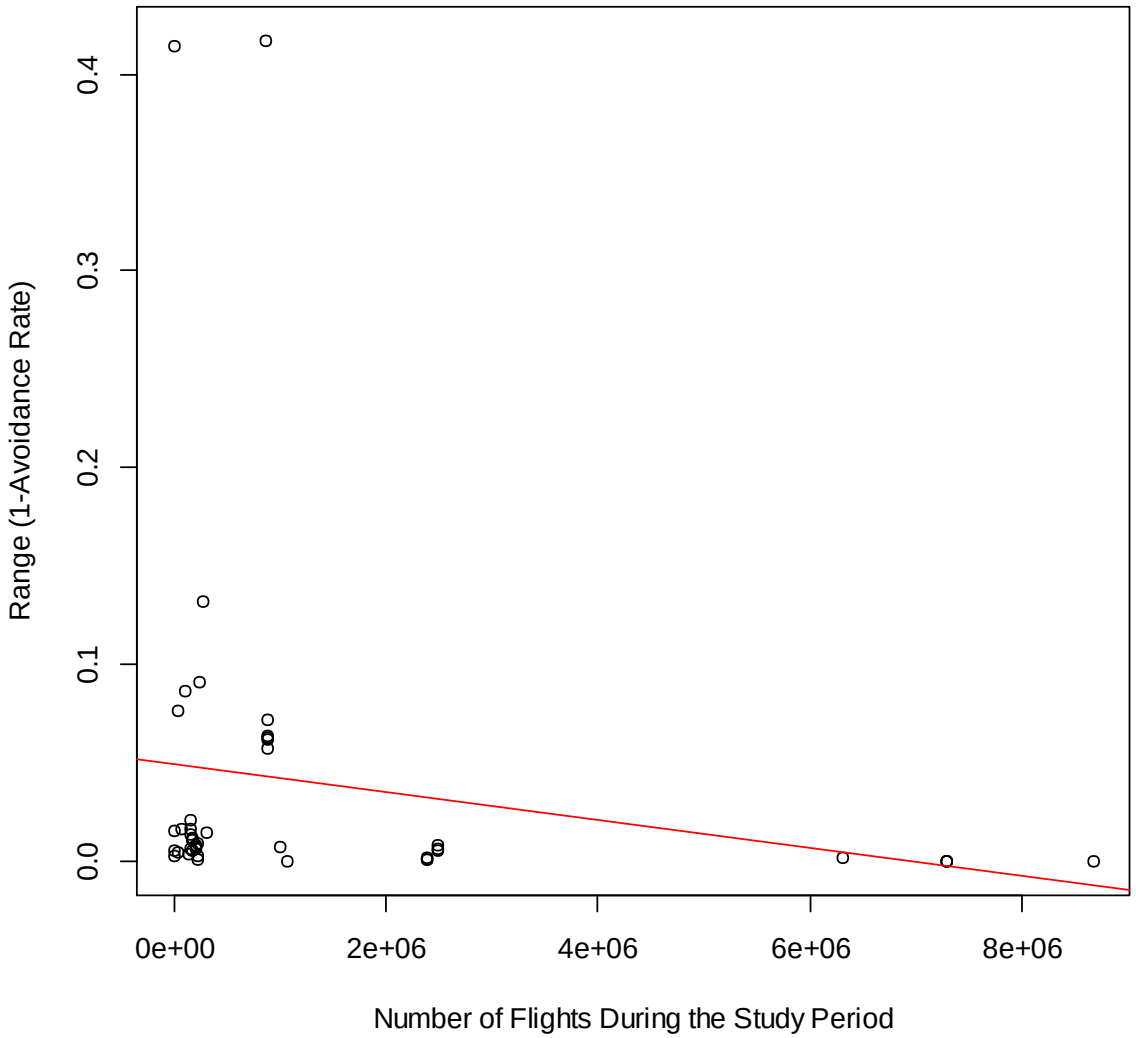


Figure 6.17 Sensitivity of the non-avoidance rate derived using option 3 of the Band model to the assumed flight height distribution.

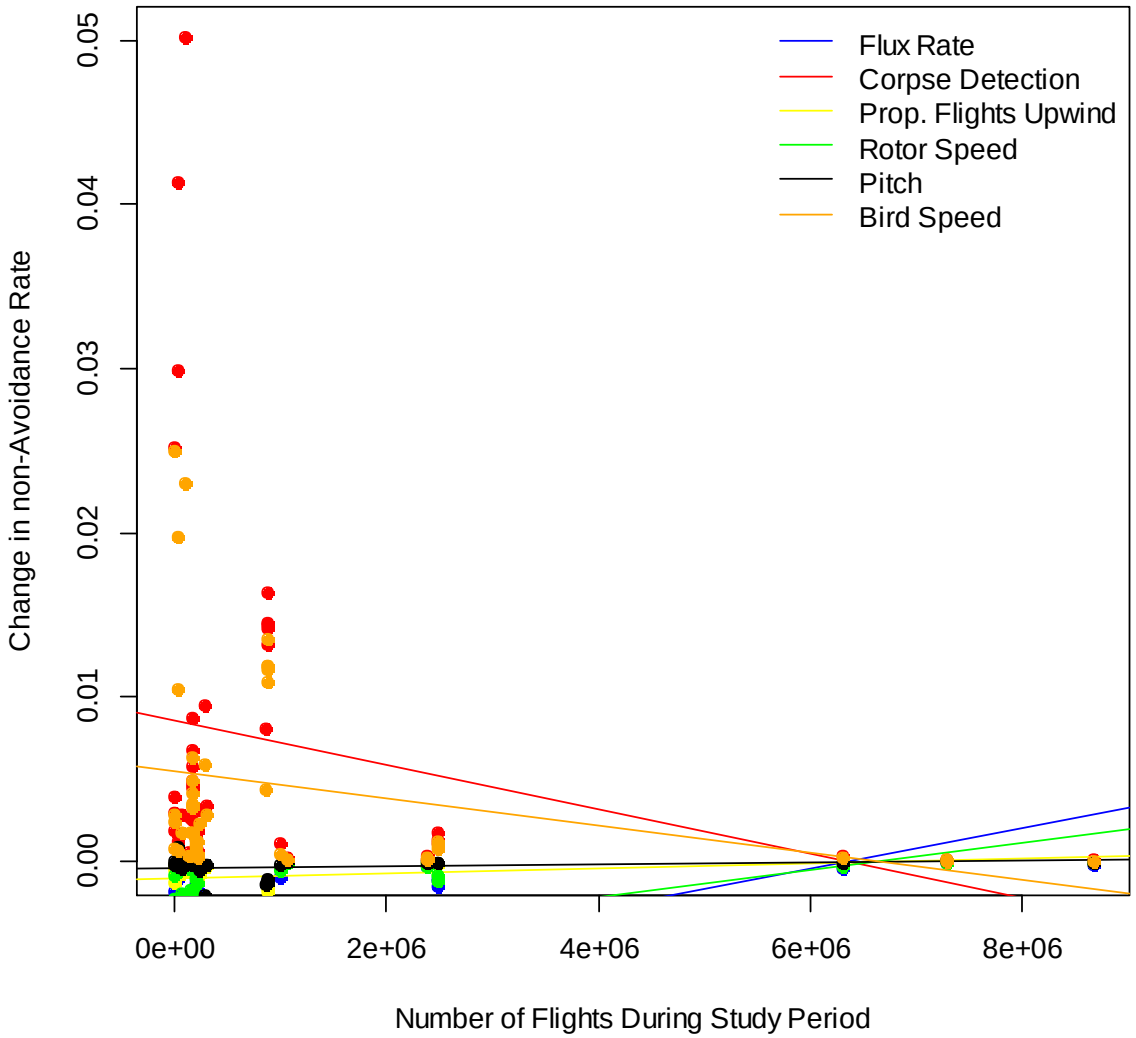


Figure 6.18 Sensitivity of non-avoidance rates derived using option 3 of the Band model to a 10% increase in each of the following parameters – flux rate, corpse detection, proportion of flights upwind, rotor speed, pitch and bird speed.

6.3 Sensitivity analysis conclusions

Avoidance rates derived using both the extended and basic Band models were sensitive to uncertainty surrounding the flux rate, corpse correction factor, rotor speed and bird speed. Whilst we considered a 10% increase in each of these parameter values to test the sensitivity of the models to the underlying assumptions, it would be valuable to consider how this compares to the actual range in each of these parameters experienced at each site. This would enable us to better quantify the uncertainty surrounding the derived avoidance rates. However, such an analysis would be complex, especially given that some parameters may co-vary, or be influenced by factors not included in the model, for example, both rotor speed and

bird speed are likely to be influenced by wind speed. Such an analysis would be beyond the scope of this project and has not been considered here.

6.4 Sensitivity to other external factors

6.4.1 Weather

The flight behaviour of birds may be strongly influenced by weather conditions. However, much of the research on this subject has been carried out in relation to migration (e.g. Larkin & Thompson 1980, Gauthreaux 1991, Zehnder *et al.* 2001, Dokter *et al.* 2011). Weather is likely to influence avoidance behaviour in two ways. Firstly, by reducing visibility, making it harder to detect hazards and, therefore, increasing the risk of collision and, secondly, by affecting the manoeuvrability of birds as a result of strong winds or the presence of thermals (Spear & Ainley 1997, Shamoun-Baranes *et al.* 2006, Shamoun-Baranes & van Loon 2006).

Increases in the numbers of recorded collisions between birds and wind turbines, or other man-made objects, have been widely reported following periods of dull, overcast weather (Crawford 1981, Winkelman 1992, Bevanger 1994). This is likely to be because poor visibility reduces the ability of birds to detect turbines, and may lead to them becoming disorientated (Williams *et al.* 1974, Able 1982, Richardson 1990). As a result, the avoidance rates of individual birds are likely to be lower during periods of poor visibility. However, data used for collision risk modelling are based on the abundance of birds in flight within the windfarm, during conditions with good visibility (Camphuysen *et al.* 2004).

In contrast, there is some, limited, evidence that some bird species may be more likely to forage inland, and less likely to fly during periods of poor visibility (Williams *et al.* 1974, Pinder 1989), reducing the number of birds in flight within the windfarm in comparison to baseline survey data used in collision risk modelling. Such a potential reduction in the number of birds in flight needs to be factored into the avoidance rates used in collision risk modelling.

As a result, it is unclear as to the extent to which conditions with poor visibility may affect the avoidance rates necessary for use in offshore windfarms. To understand the potential importance of this, it is necessary to quantify the proportion of birds likely to be in flight, at sea when visibility is poor. Data collected using modern GPS tags has the potential to answer this problem and also inform on nocturnal flight activity.

Wind speed and direction both influence bird flight behaviour (e.g. Spear & Ainley 1997, Safi *et al.* 2013), with potential implications for avoidance rates. At onshore windfarms, birds have been observed to exhibit less risky flight behaviour during periods of increasing wind (Barrios & Rodriguez 2004). During periods of strong winds, Krijgsveld *et al.* (2011) noted a decrease in the number of birds in flight around Egmond aan Zee. However, as these data were collected using radar, they emphasise that these observations may reflect increased clutter from waves, rather than a decrease in the total number of birds.

Studies have demonstrated that birds make use of wind conditions to minimise the energetic cost of flight and optimise the trade-off between the maximum range they can reach and the energy they expend in reaching it (Williams *et al.* 1974, Spear & Ainley 1997, de Lucas *et al.* 2012). They achieve this in two ways. Firstly, birds fly faster into headwinds than tail or crosswinds (Tucker & Schmidt-Koenig 1971, Larkin & Thompson 1980, Wakeling & Hodgson 1992, Spear & Ainley 1997). This would lead to a decrease in the avoidance rates derived above, as the probability of a bird colliding with a turbine would be reduced, reducing the ratio of predicted to observed collisions (see sections 6.1.6 and 6.2.7). Secondly, during stronger winds, birds have a tendency to fly more slowly (Larkin & Thompson 1980, Spear & Ainley 1997). This would lead to an increase in the avoidance rates derived above, as the probability of a bird colliding with a turbine would be increased, increasing the ratio of predicted to observed collisions (see sections 6.1.6 and 6.2.7). As with the influence of visibility, the relative importance of wind direction and speed on avoidance behaviour is hard to quantify. The situation is further complicated as birds may be less likely to fly during periods of heavy wind (Stienen *et al.* 2000). Again, the growth of modern tracking technology has the potential to help address some of these issues.

6.4.2 Habitat use

The avoidance behaviour of birds in relation to an offshore windfarm may relate to how the habitat surrounding the turbines is used – for example, are turbines close to a breeding colony, are turbines situated on a commuting route, or are turbines situated on a key foraging area. Varying responses to the surrounding habitat are likely to manifest themselves in different flight modes, and these different flight modes are likely to have different levels of collision risk associated with them (Martin 2010, 2011). When foraging or searching for roost sites and conspecifics, birds can considerably reduce their detection of obstacles, and therefore increase their risk of collision, by moving their heads vertically (Martin & Shaw 2010). Collision risk at turbines surrounding colonies, as was the case for several of the sites included in our review, may therefore be inflated by birds arriving at the colony searching for their nests. Collision risk at breeding colonies may be further inflated by the display flights undertaken by males at the start of the breeding season (May *et al.* 2013) and by the presence of young birds, whose flight behaviour may place them at greater risk of collision (Henderson *et al.* 1996) at the end of the breeding season.

It is unclear whether foraging may confer a greater collision risk than searching for conspecifics on arrival at breeding colonies. It is, therefore, difficult to say with any certainty whether birds foraging within the area of offshore windfarms may be at lesser or greater risk of collision than those returning to breeding colonies and searching for conspecifics. However, when at sea, species such as northern gannets may restrict their foraging behaviour to relatively discrete areas (Hamer *et al.* 2009, Pettex *et al.* 2010). Therefore, the majority of the area covered at sea is likely to fall within the less risky category of commuting flights. As a consequence, relying on avoidance rates derived from turbines next to breeding colonies, such as those at Bouin and Zeebrugge, for birds at sea is likely to result in an overestimate of the true risk of collision. New technology, for example camera-loggers (e.g. Votier *et al.* 2013), has the potential to help gain a better understanding of collision risk at sea both by revealing more details about activity budgets, and also by allowing

quantification of the proportion of flight time spent by birds looking straight ahead, and therefore at less risk of collision, as opposed to looking below.

6.4.3 Turbine Size

Initial analyses suggested that there was no strong relationship between turbine size and the avoidance rates derived for each of the species and groups we considered in our review (see section 5.3.3.2). Plots of avoidance rate against maximum turbine tip height appear to support this conclusion (Figure 6.19).

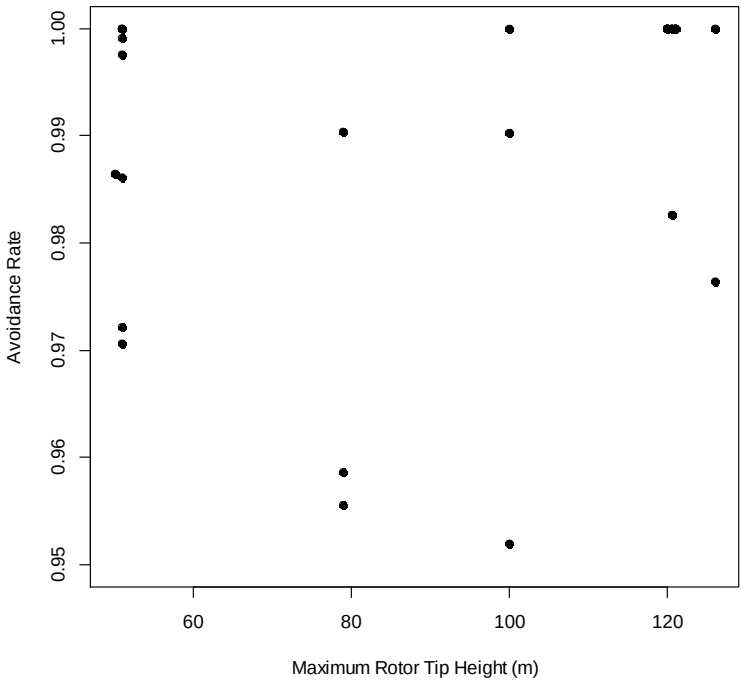


Figure 6.19 Relationship between maximum rotor tip height and the avoidance rate derived using option 1 of the Band model for all gulls.

6.4.4 Seasonality

Our analysis of the data from Zeebrugge present limited evidence that there may be a seasonal aspect to collision risk (see Section 5.1). These data suggest that avoidance rates may be higher in the autumn than in the breeding season. This may be related to two factors. Firstly the presence of younger, inexperienced birds which may have riskier flight behaviour (e.g. Henderson *et al.* 1996). Secondly, given that several of our study sites were located on the edge of breeding colonies, it may be that during the breeding season birds arriving at colonies focus on locating their nests and are therefore less likely to see turbines, increasing the collision risk.

6.4.5 Applicability of avoidance rates between species

Avoidance rates are likely to be linked to a bird's ability to detect a turbine and perceive it as a potential threat in sufficient time to take action to avoid collision. Whilst we have able to derive a within-windfarm avoidance rate for gulls, we have been unable to come up with a suitable value for northern gannet due to lack of data.

Therefore we consider other supporting evidence to evaluate whether for northern gannet total avoidance rates are likely to be higher or lower than those for gulls.

Total avoidance rates are likely to be a combination of the probability of a bird detecting a turbine and its ability to take last-second action to avoid collision. Ability to take last-second avoidance action is likely to be linked to a species manoeuvrability and a previous review used this as the basis for recommending avoidance rates for different species (Maclean *et al.* 2009). In general, expert opinion suggests that the flight manoeuvrability of northern gannets may be less than that of gulls (Garthe and Hüppop 2004, Furness *et al.* 2013), suggesting that they need more time to react to the presence of a turbine, and may therefore need to detect it earlier. Evidence from our review suggests that a high proportion of northern gannets avoid entering windfarms (Krijgsveld *et al.* 2011, Vanermen *et al.* 2013). In addition, observations undertaken within offshore windfarms suggest that very few northern gannets pass close enough to turbines to be at risk of collision (see section 5.1).

Birds are likely to be better able to detect obstacles, such as turbines, when they are looking straight ahead, as opposed to down, towards the sea-surface (Martin 2010). At sea, it may be reasonable to assume that birds will look downwards when actively foraging, and straight ahead when migrating or commuting between their breeding colonies and foraging areas. Northern gannet typically forage using area-restricted search (ARS) behaviour (based on diving activity) resulting in a relatively small proportion of the total area covered being actively used when at sea (Hamer *et al.* 2009, Votier *et al.* 2013). These ARS zones are found solely on the outbound part of the foraging trip. In contrast, gulls are not likely to limit their foraging area to such restricted zones within foraging trips (Kubetzki and Garthe 2003, Schwemmer and Garthe 2005), and may therefore spend a greater proportion of their time at sea looking towards the sea-surface. The distance over which birds can see is strongly correlated with body size (Brooke *et al.* 1999). As a consequence, northern gannets are likely to be able to detect turbines at a greater distance than gulls. Recent evidence suggests that northern gannets may respond to the presence of fishing vessels over distances of up to 11 km (Bodey *et al.* 2014). These results suggest that, at least theoretically, northern gannets may be capable of responding to the presence of a windfarm over considerable distances.

Whilst insufficient data were available to derive within-windfarm avoidance rates for northern gannets, evidence of strong avoidance of windfarms, in contrast to gulls which appear to show no consistent response, suggests that total avoidance rates for northern gannets are unlikely to be lower than those for gulls.

6.4.6 Comparability of onshore and offshore avoidance rates

The difficulty of recording collisions in the offshore environment has meant that estimates of within-windfarm avoidance rely on data collected from terrestrial windfarms. However, birds may respond differently to onshore and offshore turbines. For example, migrating geese have been found to consistently avoid entering offshore windfarms, demonstrating macro-avoidance, (Plonckzkier & Simms 2012) but may habituate to the presence of onshore turbines (Madsen & Boertmann 2008).

Understanding how avoidance behaviour differs between onshore and offshore environments requires an understanding of how flight behaviour differs between the two. Modern GPS tracking technologies have made such comparisons easier, and it appears that whilst lesser black-backed gulls may spend a similar proportion of their time in flight in both environments (Kolios 2009), there is a tendency to fly lower when offshore (Corman & Garthe 2014, Ross-Smith *et al.* in prep.). As this would result in fewer flights at risk height in the offshore than onshore environment, this would be accompanied by decrease in both the proportion of birds at risk height (and therefore the predicted collision rate) and the actual collision rate of the same proportion. Consequently the avoidance rate would be unchanged between the onshore and offshore environments. However, there remain a number of other possible differences between onshore and offshore flight behaviour. Gulls are capable of adjusting their flight mode in response to airflow patterns which differ between onshore and offshore environments, in order to minimize their energy expenditure (Shamoun-Baranes & van Loon 2006). In the onshore environment they can take advantage of thermals by soaring and wind blowing up slopes or other major topographical features resulting in slope lift soaring. Whereas in the offshore environment a boundary layer can be created as the wind blows over the surface of the sea resulting in differential air wind speeds which some seabirds including gulls can exploit for dynamic soaring (see Alexander 2004). It is unclear how these adjustments between soaring and flapping flight may influence collision risk, though changes in manouverability and flight speed may be important. At present, there are significant gaps in our understanding of how flight behaviour may differ between onshore and offshore environments, though recent technological advances may start to fill these gaps. However, at present, the data describing within-windfarm avoidance rates collected from onshore sites remains our best available evidence.

7 TOTAL AVOIDANCE RATES FOR PRIORITY SPECIES

In this section, we consider total avoidance rates for each of the five priority species – northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull and great black-backed gull.

7.1 Macro-response rates (section 5.1)

For gulls, the present evidence base is equivocal, with some studies suggesting evidence for attraction, others evidence for displacement, and others no significant response. Thus, for these species, the balance of evidence suggests a macro-response of 0 (i.e. no attraction to or avoidance of the windfarm) (Table 7.1).

Northern gannets typically show a strong macro-response to offshore windfarms. However, differences in survey methodologies make it difficult to arrive at realistic macro-response values by combining data from multiple sources. Based on currently available evidence, we believe that 0.64 to be a reasonable value for the macro-response rate (Table 7.1). However, it should be noted that this figure is based on data that are most representative of the non-breeding season.

7.2 Micro-response or meso-response rates (sections 5.2 and 5.3)

The review of existing evidence for avoidance rates in relation to offshore windfarms for the key species considered in this study indicated that insufficient data were available to generate separate micro-avoidance or meso-response rates for any of the species of interest.

7.3 Within-windfarm avoidance rates (section 5.4)

Within-windfarm avoidance rates, representing a combination of meso-responses and micro-avoidance may be derived by comparing observed collisions to those expected in the absence of avoidance (see equation 6 under section 1). Options 1 and 2 of the Band model are mathematically identical (both termed the basic Band model), with the proportion of birds at collision risk height estimated from modelled flight height distributions for option 2 and based on site-specific observational data using option 1. Therefore, it is necessary to use the same avoidance rates for both model options. As the rates derived using option 1 utilise site-specific data, rather than data derived from a generic curve (produced following the methodology of Johnston *et al.* 2013), we feel that these values are the most appropriate to recommend for use with the basic Band model. With respect to the extended Band model, the rate derived should be acknowledged as, potentially, being precautionary as, at several key sites, it is based on an underestimate of the proportion of birds flying at collision risk height (see Appendix 7). As a consequence, when calculating the avoidance rate by comparing the predicted and observed number of collisions, the resulting value is lower than would otherwise be expected. Therefore, where there is a significant difference between the observed proportion of birds at collision risk height and the proportion predicted to be at collision risk height from modelled distributions, the avoidance rates derived for use with the extended model are not considered appropriate as they will be based on an inaccurate assessment of the number of birds at risk of collision.

An alternative methodology with which to derive a within-windfarm avoidance rate for use with the extended Band model is described by in Annex 1 to this report. Following this methodology, the ratio between the number of collisions expected in the absence of avoidance derived using options 2 and 3 of the Band model is used to modify the avoidance rate derived using option 1 of the Band model. However, this requires knowledge of the flight height distribution (e.g. to 1m resolution) at the windfarm concerned – as opposed to the proportions of birds assigned to different flight height categories – in order to separate geometric avoidance (i.e. birds passing the rotor at lower altitudes where the probability of collision is lower) from behavioural avoidance. Whilst it is possible to use this methodology without knowledge of the flight height distribution at the windfarm in question, the result would be that the predicted collision rate using option 3 would be identical to that obtained using option 2. However, this methodology is likely to be of value in the future as data collection techniques improve and detailed flight height distributions are derived on a site-specific basis.

We were able to derive within-windfarm avoidance rates for herring gull and lesser black-backed gull (Table 7.1). Based on a sample of 526,048 predicted flights through windfarms, we derived an avoidance rate of 0.9959 (± 0.0006 SD) for herring gull based on the basic Band model and 0.9908 (± 0.0012 SD) using the extended Band model. For lesser black-backed gull, the derived avoidance rates were 0.9982 (± 0.0005 SD) and 0.9957 (± 0.0011 SD) respectively, based on a sample of 101,746 predicted flights through windfarms. However, the larger sample size and the fact that data originate from a greater number of sites (see Appendix 7) means that the avoidance rates derived for herring gull are more robust than those derived for lesser black-backed gull. We also derived within-windfarm avoidance rates for large gulls as a group. This group includes all birds positively identified as herring gull (this species accounting for 526,048 of the total of 639,560 flights through windfarms), lesser black-backed gull or great black-backed gull, but also those with uncertain species identification (10,638 predicted flights through windfarms), for example those identified as herring/lesser-black backed gull. For the large gulls group, we derived avoidance rates of 0.9956 (± 0.0004 SD) using the basic Band model and 0.9898 (± 0.0009 SD) using the extended Band model. A comparison of the observed and predicted proportions of birds at collision risk height (Appendix 7) shows that whilst there are some notable differences in these values, across most sites they are broadly consistent. For this reason, we feel that the avoidance rates derived using both the basic and extended Band models are appropriate to use.

We also derived within windfarm avoidance rates for small gulls (1,589,953 predicted flights through windfarms) based largely on data collected from common gull (746,668 predicted flights through windfarms) and black-headed gull (841,008 predicted flights through windfarms). For species within the small gulls group, we derived within-windfarm avoidance rates of 0.9921 (± 0.0015 SD) for use with the basic Band model and 0.9027 (± 0.0068 SD) for use with the extended Band model (Table 7.1). However, given significant differences between the proportion of birds observed and predicted to be at collision risk height at a number of key sites, we do not feel that it is appropriate to use the avoidance rate derived for use with the extended Band model for the small gulls grouping. These differences are likely to

arise from the fact that the data considered here originate from the terrestrial environment, often close to breeding colonies, whilst the modelled data were collected from the offshore environment.

Finally, we calculated a within-windfarm avoidance rate for all gulls as a group (2,567,124 predicted flights through windfarms). As with the large gull and small gull groups, this incorporated data for individuals with uncertain identification (350,338 predicted flights through windfarms), for example 'gull spp'. For all gulls, we derived an avoidance rate of 0.9893 (± 0.0007 SD) for use with the basic Band model and 0.9672 (± 0.0040 SD) for use with the extended Band model (Table 7.1). However, as with the small gulls group this includes data for which there were significant differences – due partly to the inclusion of unidentified gulls – between the observed and predicted proportions of birds at collision risk height. For this reason we do not feel that it is appropriate to use the avoidance rate derived for use with the extended Band model for the all gulls groupings.

Insufficient data were available to identify a reliable within-windfarm avoidance rate for northern gannet (Table 7.1).

It is important to note that where we report the standard deviation around the derived within windfarm avoidance rates, this relates variability between sites and not to uncertainty in the model input parameters. Estimating the contribution of the model input parameters to the uncertainty associated with the derived avoidance rates requires a more detailed understanding of the real range of values associated with each parameter than is available currently.

7.4 Total avoidance rates

Total avoidance rates are also provided in Table 7.1. Ideally, total avoidance rates should be calculated using equation 8 (section 3.1). For gulls, the balance of evidence suggests a macro-response of 0 (i.e. no consistent attraction to or avoidance of the windfarm). Consequently, the total avoidance rates for these species are equal to the within-windfarm avoidance rates.

As data describing macro-responses to the windfarm are limited, we are unable to estimate the variability around the macro-response rate. For this reason, whilst we are able to provide an estimate of variability around the within windfarm avoidance rates, we are unable to provide an estimate of variability of uncertainty around the total windfarm rates.

Table 7.1 Derived avoidance rates for priority species and current knowledge gaps based on the review of available data. Empty cells indicate a lack of robust and/or consistent data on which to base conclusions. Colour coding indicates confidence in presented values (based on sample size, representativity of data): green = highest, orange = intermediate, red = lowest (i.e. not suitable for use in CRM). Confidence in total avoidance rates reflects the lower of the confidence ratings given for macro-responses and within-windfarm avoidance rates.

Species/species groupings and sample size for within-windfarm avoidance rate given in parentheses*	Macro-response[1]	Meso-response[2]	Micro-avoidance[3]	Within-windfarm avoidance basic Band model[4]	Within-windfarm avoidance extended Band model[4]	Total avoidance basic Band model (1-total avoidance)	Total avoidance extended Band model (1-total avoidance)	Caveats
Black-legged kittiwake (0)	None							Whilst data were available for macro-response, no clear patterns were evident across studies. No data available for within-windfarm avoidance.
Lesser black-backed gull (101,746)	None			0.9982 (± 0.0005)	0.9957 (± 0.0011)	0.9982 (0.0018)	0.9957 (0.0043)	Whilst data were available for macro-response, no clear patterns were evident across studies. Within-windfarm rate based on data from only two sites.

Herring gull (526,048)	None			0.9959 (± 0.0006)	0.9908 (± 0.0012)	0.9959 (0.0041)	0.9908 (0.0092)	Whilst data were available for macro-response, no clear patterns were evident across studies. Within-windfarm rate based on a large sample size from seven different sites.
Great black-backed gull (1,128)	None							Whilst data were available for macro-response, no clear patterns were evident across studies. No within-windfarm avoidance rates estimated due to extremely small sample size.

Small gull spp (1,589,953) Comprising: black-headed gull (746,668), common gull (841,008), common/black-headed gull (2,090), little gull (188)	None			0.9921 (± 0.0015)	0.9027 (± 0.0068)	0.9921 (0.0079)	0.9027 (0.0973)	Whilst data were available for macro-response, no clear patterns were evident across studies. Within-windfarm avoidance rates based on large sample size from eight different sites. However, differences between observed and predicted proportions of birds at collision risk height mean it is not appropriate to use value derived for extended model.
Large gull spp (639,560) Comprising: lesser black-backed gull 101,746, herring gull 526,048, herring/Caspian gull 1,417, herring/lesser black-backed gull 8,345, herring/yellow-legged gull 876, great black-backed gull 1,128	None			0.9956 (± 0.0004)	0.9898 (± 0.0009)	0.9956 (0.0044)	0.9898 (0.0102)	Whilst data were available for macro-response, no clear patterns were evident across studies. Within-windfarm avoidance rates based on large sample size from seven different sites.

Gull spp (2,567,124) Comprising: black-headed gull 746,668, common gull 841,008, common/black-headed gull 2,090, little gull 188, lesser black-backed gull 101,746, herring gull 526,048, herring/Caspian gull 1,417, herring/lesser black-backed gull 8,345, herring/yellow-legged gull 876, great black-backed gull 1,128, gull spp. 337,610	None			0.9893 (± 0.0008)	0.9672 (± 0.0018)	0.9893 (0.0107)	0.9672 (0.0328)	Whilst data were available for macro-response, no clear patterns were evident across studies. Within-windfarm avoidance rates based on large sample size from nine different sites. However, differences between observed and predicted proportions of birds at collision risk height mean it is not appropriate to use value derived for extended model.

Northern gannet (0)	0.64							Macro-response rates for northern gannet indicated strong avoidance of windfarms. As data were available from a limited number of sites, the lowest reported value was felt to be most appropriate as a precautionary figure. Note the majority of data comes from the non-breeding season and it is unclear how applicable these findings may be to the breeding season. No data available for within-windfarm avoidance.

[1] See section 5.4; [2] See section 5.1; [3] See section 5.2; [4] see section 5.3.

7.5 Recommended avoidance rates

Please note that these recommendations apply to the five priority species only – northern gannet, black-legged kittiwake, lesser black-backed gull, herring gull and great black-backed gull – they are not intended to be applied to seabirds more generally.

Whilst we have estimated within-windfarm avoidance rates to four decimal places, current guidance from SNH is that expressing avoidance rates to more than three decimal places is unwarranted (SNH 2013). Given the inherent uncertainty in the data we feel that this is a sensible approach to apply to total avoidance rates. For this reason, we round within-windfarm avoidance rates down to three decimal places when deriving recommended total avoidance rates.

- A **macro-response rate of 0.64** is recommended for **northern gannet** (section 5.4). However, no data were available to derive a within-windfarm avoidance rate for this species (section 5.3). Given that there is consistent evidence for high macro-avoidance, and considering the at-sea ecology of northern gannet and gulls (section 6.3.5), we feel that there is no reason to suppose that the total avoidance rates for **northern gannet** should be less than those for all gulls (as opposed to large gulls). A **total avoidance rate** of **0.989** is thus recommended for use with the basic Band (2012) collision risk model. This would reflect a within windfarm avoidance rate of 0.9703. We acknowledge that this is precautionary, but in the absence of more species-specific data, we feel it is appropriate. However, given the evidence available at present, we are unable to recommend an avoidance rate for use with the extended Band (2012) collision risk model.

- No consistent evidence of macro-avoidance was found for **black-legged kittiwake** (section 5.4). It was not possible to derive species-specific within-windfarm avoidance rates for **black-legged kittiwake** (section 5.3)**.** However, as black-legged kittiwake have similar wing morphologies (wingspan, wing:body aspect ratio, wing area: Robinson 2005, Alerstam *et al.* 2007), flight speeds (Alerstam *et al*. 2007) and flight altitudes (Cook *et al.* 2012, Johnston *et al.* 2014b) to black-headed and common gulls, which contribute the majority of records for the small gulls group, the within-windfarm avoidance rates derived for the small gulls group were considered appropriate for this species. A **total avoidance rate** of **0.992** is thus recommended for the basic Band model. However, given the evidence available at present, we are unable to recommend an avoidance rate for use with the extended Band (2012) collision risk model (section 5.3).

- No consistent evidence of macro-avoidance was found for **lesser black-backed gull** (section 5.4). Whilst it was possible to derive species-specific within-windfarm avoidance rates for lesser black-backed gull, these were based on limited data and thus the within-windfarm avoidance rates for large gulls were considered more appropriate for use for this species (section 5.3). A **total avoidance rate** of **0.995** is thus recommended for use with the basic Band model and a **total avoidance rate** of **0.989** for use with the extended Band model.

- No consistent evidence of macro-avoidance was found for **herring gull** (section 5.4) and thus total avoidance rates reflect species-specific within-windfarm avoidance rates. A species-specific **total avoidance rate** of **0.995** is thus recommended for use with the basic Band model and a **total avoidance rate** of **0.990** for use with the extended Band model (section 5.3).

- No consistent evidence of macro-avoidance for **great black-backed gull** (section 5.4). It was not possible to derive species-specific within-windfarm avoidance rates for **great black-backed gull.** Given the taxonomic similarity between species within the large gulls group, the avoidance rates derived for use with this group were considered to be appropriate for great black-backed gull (section 5.3). A **total avoidance rate** of **0.995** is thus recommended for the basic Band model and a **total avoidance rate** of **0.989** for use with the extended Band model.

At present, the evidence available does not enable us to recommend a robust avoidance rate for northern gannet or black-legged kittiwake for use with Band model option 3. This does not imply that option 3 is not suitable for these species, and given the programmes of work currently underway in the offshore environment, it is envisaged that an appropriate rate will be derived in the near future. Note, while it is not possible to recommend a robust avoidance rate for use for these species at this time, this does not preclude presenting a no-avoidance collision estimate using option 3 alongside collision estimates derived using option 1 and/or option 2 (with or without using the avoidance rates recommended here) to inform on likely collision risk.

Table 7.2 Recommended total avoidance rates for use in the basic and extended Band models with each of the five priority species.

Species (rate used)	Basic Band model avoidance rate	Extended Band model avoidance rate
Northern gannet (all gull avoidance rate)	0.989	Not available
Black-legged kittiwake (small gull avoidance rate)	0.992	Not available
Lesser black-backed gull (large gull avoidance rate)	0.995	0.989
Herring gull (species-specific avoidance rate)	0.995	0.990
Great black-backed gull (large gull avoidance rate)	0.995	0.989

8 TRANSFERABILITY OF AVOIDANCE RATES BETWEEN MODELS

There are various collision risk models currently available within the scientific literature to estimate likely collision and mortality of birds due to windfarms (Band 2012; Desholm 2006; Eichorn *et al.* 2012; McAdam 2005; Smales *et al.* 2013; Tucker 1996; Holstrom 2011). The models vary in numerous ways including whether static components such as the tower are included in calculations, if oblique angles of attack are considered and whether single or multiple turbines are assessed, as well as how avoidance behaviour is incorporated. Although the Band model (Band 2012) is the most widely used collision risk model in the UK, it is not the only one available and therefore any developments in our understanding of avoidance behaviour should consider, where possible, these alternative models.

Although described in the literature, avian collision risk models are often not presented in enough detail to reproduce. The majority of models consider avoidance behaviour as an add-on to the process of estimating the probability of collision, separate from the calculation of collision probability for a single rotor transit. From the information available, however, it would seem that the definitions and avoidance rates presented in our report would generally be suitable for use within a range of collision risk models, not only Band (2012). Here we provide examples of how the definitions and rates may align with some of these alternative models.

Desholm (2006) developed a stochastic model analysis of avian collision which included variability in the input parameters and outputs of the model. Although it was a very specific example from an offshore windfarm in the Baltic Sea, the method could be used elsewhere. The definitions used in our project seem suitable for the model. The method considered the different stages at which birds may avoid a windfarm and uses values for the proportion of birds entering the windfarm (1 - macro-avoidance), the proportion within the horizontal/vertical reach of rotor blades (1 - meso-avoidance) and also the proportion trying to cross the area swept by the rotor blades without showing avoidance (1 - micro-avoidance).

Eichorn *et al.* (2012) developed an agent-based, spatially-explicit model of red kite foraging behaviour to assess collision risk related to wind turbines. The model is largely stochastic and combines a spatial model with a collision risk model. Although the study was specific to red kite, the methods could be used more widely. The model uses the method from Band (2007) for calculating probability of collision from a single rotor transit therefore it is likely that any definitions for avoidance behaviour provided by our study will be suitable. The model specifically includes the probability of a bird recognising the threat and actively avoiding, and this avoidance rate is taken from the literature. The value ranges from 0.98 – 0.995 and is therefore likely to be a value for overall avoidance, however the definitions within this study for meso- and micro-avoidance would seem to fit more appropriately because it is a single bird avoiding a single turbine within a 100 m x 100 m grid cell.

Smales *et al.* (2013) describe a collision risk model developed by Biosis Propriety Limited which has been widely used to assess wind energy developments in Australia since 2002. The model uses a deterministic approach and provides a predicted number of collisions between turbines and a local or migrating population of birds. The model uses flight activity data from the windfarm site and applies

avoidance rate to the typical number of turbines encountered per flight. Therefore the definitions and rates for within windfarm behaviour should be suitable in this context.

A note of caution when considering avoidance rates and their application within different collision risk models is that although not the intended purpose, avoidance rate may have become a sink for multiple sources of error and uncertainty within a model. Collision risk models rarely state the associated error along with collision estimates. In the process of apportioning overall avoidance into the different components of macro-, meso-, and micro-avoidance, this previous inclusion of model error may need to be considered, and may be model-specific.

9 CONCLUSIONS

We have derived within-windfarm avoidance rates for a variety of species for specific sites. In some cases, these differ from those presented elsewhere using, apparently, the same data (see Natural England/JNCC note). For this reason, we include an appendix (Appendix 7) detailing how each of our values has been derived. Note that the values in Appendix 7 are supplied for illustrative purposes only and that we would recommend the use of the total avoidance rates presented in Table 7.2. Given the variability in the values that have been presented for some datasets, we believe that this level of transparency is crucial to enable readers to come to an informed opinion as to what represents a robust avoidance rate. The derivation of the flux rate through the windfarm is particularly important, as it can have quite a strong influence on the predicted number of collisions, and therefore, the final avoidance rate.

Very little data were available describing separate meso-responses or micro-avoidance. There were limitations in the data from each of the studies we identified. However, observations of flight behaviour around individual turbines indicate that birds very rarely pass close to the rotor blades, suggesting that a significant proportion of avoidance behaviour is likely to occur at a meso-scale. We identified evidence from several sites to suggest that avoidance behaviour may be influenced by both the layout of the windfarm (e.g. the inter-turbine spacing) and the operational status of turbines. There is some limited evidence to suggest that overall avoidance rates may be lower during the breeding season than the non-breeding season, although significantly more data are required to confirm this hypothesis (see section 5.3.3.1).

The availability of suitable data has been a key problem throughout this review. In part, this relates to the difficulty in collecting collision data at sea, leading to gaps in data for key species such as northern gannet and black-legged kittiwake. It is to be hoped that the ongoing ORJIP work will help to address this issue. However, it also relates to the way in which data have been collected as part of post-construction monitoring at offshore windfarms. We identified extremely limited evidence for macro-response rates for our priority species. In many instances, this may be because when impacts which may contribute to macro-avoidance, such as displacement or barrier effects, are considered, the focal species are usually auks, divers and sea-ducks. As a consequence, the impacts on other species, such as northern gannet are less well understood.

Our review highlights that there are still significant data gaps in relation to avoidance rates for marine birds and offshore windfarms, particularly in relation to micro- and meso-responses, as opposed to the correction factors often used as avoidance rates at present. Despite this, we feel that our review represents a significant step forward. We are able to recommend for the first time within-windfarm avoidance rates for gulls using both the basic Band (2012) model (options 1 and 2) and extended Band (2012) model (option 3) based on significantly more data than has been used to make recommendations for geese and raptors in the past (e.g. Pendlebury 2006, Whitfield 2009). Significant data gaps still remain for within-windfarm avoidance behaviour in the northern gannet.

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References

Able, K.P. 1982. Skylight polarization patterns at dusk influence migratory orientation in birds. *Nature* **299,** 550-551.

Alerstam, T., Rosén, M., Bäckman, J., Ericson, P.G.P. & Hellgren, O. 2007. Flight Speeds among Bird Species: Allometric and Phylogenetic Effects. *PLoS Biol* **5,** e197. doi:10.1371/journal.pbio.0050197

Alexander, D.E. 2004. *Nature's flyers: Birds, Insects and the biomechanics of flight*. JHU Press, Baltimore.

Arnqvist, G. & Wooster, D. 1995. Meta-analysis: synthesizing research findings in ecology and evolution. *Trends in Ecology & Evolution* **10,** 236-240.

Band, W. 2000. *Guidance WINDFARMS AND BIRDS: calculating a theoretical collision risk assuming no avoidance action*. Guidance notes series 2000. Scottish Natural Heritage, Battleby.

Band, W., Madders, M., & Whitfield, D.P. 2007. Developing field and analytical methods to assess avian collision risk at wind farms. In: de Lucas, M., Janss, G.F.E. & Ferrer, M. (eds.) *Birds and Wind farms: Risk Assessment and Mitigation*, pp. 259-275. Quercus, Madrid

Band, W. 2012. *Using a collision risk to assess bird collision risk for offshore wind farms.* Report to SOSS. http://www.bto.org/sites/default/files/u28/downloads/Projects/Final_Report_SOSS02_Band1ModelGuidance.pdf.

Banks, A.N., Burton, N.H.K., Austin, G.E., Carter, N., Chamberlain, D.E., Holt, C., Rehfisch, M.M., Wakefield, E. & Gill, P. 2005. *The potential effects on birds of the Greater Gabbard Offshore Wind farm Report for February 2004 to March 2005*, BTO Research Report No. 419.

Barrios, L. & Rodríguez, A. 2004. Behavioural and environmental correlates of soaring-bird mortality at on-shore wind turbines. *Journal of Applied Ecology* **41,** 72-81.

Batschelet, E. 1976. *Introduction to Mathematics for Life Scientists*. Springer-Verlag, Berlin.

Beatrice Offshore Wind farm Ltd. 2012. *Beatrice Offshore Wind farm Environmental Statement*.

Bernardino, J., Bispo, R., Torres, P., Rebelo, R., Mascarenhas, M., & Costa, H. 2011. Enhancing carcass removal trials at three wind energy facilities in Portugal. *Wildlife Biology in Practice*, **7**, 1-14.

Bevanger, K. 1994. Bird interactions with utility structures: collision and electrocution, causes and mitigating measures. *Ibis* **136,** 412-425.

Blew, J., Hoffmann, M., Nehls, G. & Hennig, V. 2008. *Investigations of the bird collision risk and the response of harbour porpoises in the offshore wind farms Horns Rev, North Sea and Nysted, Baltic Sea, in Denmark. Part 1: Birds*. http://bioconsult-sh.de/pdf/Final%20Report%20Offshore%20Collision%20Risk%20Birds%20DK%202 0081219.pdf.

Bodey, T. W., Jessopp, M. J., Votier, S. C., Gerritsen, H. D., Cleasby, I. R., Hamer, K. C., Patrick, S. C., Wakefield, E. D. & Bearhop, S. 2014. Seabird movement reveals the ecological footprint of fishing vessels. *Current Biology*, **24,** R514-R515.

Brooke, M. D. L., Hanley, S., & Laughlin, S. B. 1999. The scaling of eye size with body mass in birds. *Proceedings of the Royal Society of London. Series B: Biological Sciences* **266,** 405-412.

Brown, W.K. & B.L. Hamilton. 2004. *Bird and bat monitoring at the McBride Lake Wind farm, Alberta, 2003-2004*. Unpublished report prepared for Vision Quest Windelectric, Calgary, AB.

Brown, W.K. & B.L. Hamilton. 2006. *Bird and bat interactions with wind turbines, Castle River Wind farm, Alberta, 2003-2004*. Unpublished report prepared for Vision Quest Windelectric, Calgary, AB.
Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L. & Thomas, L. 2001. *Introduction to Distance Sampling*. Oxford: Oxford University Press.

Buckland, S.T., Burt, M.L., Rexstad, E.A., Mellor, M., Williams, A.E. & Woodward, R. 2012. Aerial surveys of seabirds: the advent of digital methods. *Journal of Applied Ecology* **49,** 960-967.

Bundesamt fur Seeschiffart und Hydrographie (BSH). 2011. *Okologische Begleitforshung bei alha ventus erste Ergebnisse (Environmental research at alpha ventus – first results).* Contributions from the Event of 10 May 2010, Katholische Akademie Hamburg.

Burton, N.H.K., Thaxter, C.B., Cook, A.S.C.P., Austin, G.E., Humphreys, E.M., Johnston, A., Morrison, C.A., Wright, L.J. 2013. *Ornithology Technical Report for the Proposed Dogger Bank Creyke Beck Offshore Wind farm Projects*, BTO Research Report 630, Thetford.

Camphuysen, C.J., Calvo, B., Durinck, J., Ensor, K., Follestad, A., Furness, R. W., Garthe, S., Leaper, G., Skov, H., Tasker, M. L. & Winter, C. J. N., 1995. *Consumption of discards by seabirds in the North Sea. Final Report of EC DG XIV Research Contract BIOECO/93/10.* NIOZ-Report 1995–5, Netherlands Institute for Sea Research, Texel. 202 pp.

Camphuysen, C.J. & Garthe, S. 2004. Recording foraging seabirds at sea. Standardised recording and coding of foraging behaviour and multi-species foraging associations. *Atlantic Seabirds* **6,** 1-32.

Camphuysen, C.J., Fox, A.D., Leopold, M.F., & Petersen, I.K. 2004. *Towards standardised seabirds at sea census techniques in connection with environmental impact assessments for offshore wind farms in the UK A comparison of ship and aerial sampling methods for marine birds, and their applicability to offshore wind farm assessments.* Koninklijk Nederlands Instituut voor Onderzoek der Zee Report commissioned by COWRIE.

Centrica Energy. 2007. *Lincs Offshore Wind farm Environmental Statement*. Centrica Energy, Uxbridge.

Centrica Energy. 2009. *Race Bank Offshore Wind farm Environmental Statement.*

Centrica Energy. 2012. Irish Sea Zone: Zonal Appraisal and Planning (ZAP) Report. A strategic approach to the identification of Potential Areas for Development within the Irish Sea Zone.

Chamberlain, D., Freeman, S., Rehfisch, M., Fox, T. & Desholm, M. 2005. *Appraisal of Scottish Natural Heritage's wind farm collision risk model and its application.* BTO Research Report 401. British Trust for Ornithology, Thetford.

Chamberlain, D.E., Rehfisch, M.R., Fox. T., Desholm, M. & Anthony, S.J. 2006. The effect of avoidance rates on bird mortality predictions made by wind turbine collision risk models. *Ibis* **148** (Suppl. 1), 198-202.

Channel Energy Ltd. 2013. *Atlantic Array Offshore Wind farm Environmental Statement*. http://infrastructure.planningportal.gov.uk/wp-content/ipc/uploads/projects/EN010015/2.%20Post-Submission/Application%20Documents/Environmental%20Statement/6.1.10%20Chapter%2010%20Ornithology.pdf.

Christensen, T.K., Clausager, I. & Petersen, I.K. 2003. *Base-line investigations of birds in relation to an offshore wind farm at Horns Rev, and results from the year of construction.* Commissioned report to Tech-wise A/S. National Environmental Research Institute.

Christensen, T.K., Hounisen, J.P., Clausager, I. & Petersen, I.K. 2004*. Visual and radar observations of birds in relation to collision risk at the Horns Rev offshore wind farm. Annual status report 2003*. Commissioned report to Elsam Engineering A/S. National Environmental Research Institute.

Cochran, W.G. 1977. *Sampling Techniques*, 3rd Edition. John Wiley New York.

Collier, M.P., Dirksen, S. & Krijgsveld, K.L. 2011a. *A review of methods to monitor collisions or micro-avoidance of birds with offshore wind turbines: Part 1 Review. Strategic Ornithological Support Services Project SOSS-03A*, Bureau Waardenburg, Culemborg, the Netherlands.

Collier, M.P., Dirksen, S. & Krijgsveld, K.L. 2011b. *A review of methods to monitor collisions or micro-avoidance of birds with offshore wind turbines: Part 2 Feasibility*

study of systems to monitor collisions. Strategic Ornithological Support Services Project SOSS-03A, Bureau Waardenburg, Culemborg, the Netherlands.

Cook, A.S.C.P., Johnston, A., Wright, L.J. & Burton, N.H.K. 2012. A review of flight heights and avoidance rates of birds in relation to offshore wind farms. Strategic Ornithological Support Services Project SOSS-02. BTO Research Report Number 618. BTO, Thetford.

Cook, A.S.C.P., Ross-Smith, V.H., Roos, S., Burton, N.H.K., Beale, N., Coleman, C., Daniel, H., Fitzpatrick, S., Rankin, E., Norman, K. & Martin, G. 2011. *Identifying a range of options to prevent or reduce avian collision with offshore wind farms, using a UK-based case study*. BTO Research Report No. 580 to Defra. British Trust for Ornithology, Thetford.

Corman, A. M., & Garthe, S. 2014. What flight heights tell us about foraging and potential conflicts with wind farms: a case study in Lesser Black-backed Gulls (Larus fuscus). *Journal of Ornithology*, **155**, 1037-1043.

Crawford, R.I. 1981. Weather, migration and autumn bird kills at a north Florida TV tower. *Wilson Bulletin* **93,** 189-195.

Davies, I., Finlay, A., Cole, E. & Alcock, P. 2013. Avoidance behaviour at offshore wind farms: a new joint industry approach. *BOU Proceedings – Marine Renewables and Birds*. http://www.bou.org.uk/bouproc-net/marine-renewables/davies-et-al-b.pdf.

de Lucas, M., Ferrer, M. & Janss, G. F. E. 2012. Using wind tunnels to predict bird mortality in wind farms: The case of griffon vultures. *PLoS ONE* **7,** e48092. doi:10.1371/journal.pone.0048092.

Desholm, M. 2005. *TADS investigations of avian collision risk at Nysted offshore wind farm, autumn 2004*. NERI, Denmark

Desholm, M. 2006. *Wind farm related mortality among avian migrants – a remote sensing study and model analysis*. PhD University of Copenhagen.

Desholm, M., Fox, A.D., Beasley, P.D.L., Kahlert, J. 2006. Remote techniques for counting and estimating the number of bird-wind turbine collisions at sea: a review. *Ibis* **148,** 76-89.

Desholm, M. & Kahlert, J. 2005. Avian collision risk at an offshore wind farm. *Biology Letters* **1**: 296-298.

Devereux, C. L., Denny, M. J. H. & Whittingham, M. J. 2008. Minimal effects of wind turbines on the distribution of wintering farmland birds. *Journal of Applied Ecology* **45,** 1689-1694.

Dokter, A. M., Liechti, F., Stark, H., Delobbe, L., Tabary, P., & Holleman, I. 2011. Bird migration flight altitudes studied by a network of operational weather radars. *Journal of the Royal Society Interface*, **8**, 30-43.

DONG Energy. 2009. *Westernmost Rough Offshore Windfarm*, DONG Energy, Copenhagen

DONG Energy. 2013. *Walney Extension Offshore Wind Farm Environmental Statement*. http://infrastructure.planningportal.gov.uk/wp-content/ipc/uploads/projects/EN010027/2.%20Post-Submission/Application%20Documents/Environmental%20Statement/10.2.25%20ES%20Annex%20B.7.A%20Ornithology%20Technical%20Report.pdf.

DONG Energy. 2013. *Burbo Bank Extension Offshore Wind Farm Environmental Statement*, http://infrastructure.planningportal.gov.uk/wp-content/ipc/uploads/projects/EN010026/2.%20Post-Submission/Application%20Documents/Environmental%20Statement/5.1.2.15%20Offshore%20Ornithology.pdf.

Douglas, D.J.T., Follestad, A., Langston, R.H.W. & Pearce-Higgins, J.W. 2012. Modelled sensitivity of avian collision rate at wind turbines varies with number of hours of flight activity input data. *Ibis* **154,** 858-861.

Dulac, P. 2008. *Evaluation de l'impact du parc eolian de Bouin (Vendee) sur l'avifaune et les chauves-souris. Bilan de 5 anness de suivi*. Ligue pour la Protection des Oiseaux, Nantes.

East Anglia Wind. 2012. *East Anglia One Offshore Wind farm* http://infrastructure.planningportal.gov.uk/wp-content/ipc/uploads/projects/EN010025/2.%20Post-Submission/Application%20Documents/Environmental%20Statement/7.3.7%20Volume%202%20Chapter%2012%20Ornithology%20Marine%20and%20Coastal.pdf.

ECON. 2009. *Ornithological assessment of the Dudgeon Offshore Wind farm: Technical Report*, ECON Ecology, Norwich.

EDF Energy (Northern Offshore Wind) Ltd. 2004. *Teesside Offshore Wind farm Environmental Statement.* EDF Energy – NOWL, Sunderland http://www.redcar-cleveland.gov.uk/Maps/R_2008_0526_FFM/Cover%20and%20Contents.pdf.

Eichhorn, M., Johst, K., Seppelt, R. & Drechsler, M. 2012. Model-based estimation of collision risks of predatory birds with wind turbines. *Ecology & Society* **17**: 1. http://dx.doi.org/10.5751/ES-04594-170201.

EON. 2012. *Rampion Offshore Wind farm Environmental Statement.* http://infrastructure.planningportal.gov.uk/wp-content/ipc/uploads/projects/EN010032/2.%20Post-Submission/Application%20Documents/Environmental%20Statement/6.1.11%20Marine%20Ornithology.pdf.

Everaert, J. 2003. Windturbines en vogels in Vlaanderen: voorlopige onderzoeksresultaten en aanbevelingen. *Natuur Oriolus* **69,** 145-155.

Everaert, J. 2008. *Effecten van windturbines op de fauna in Vlaanderen. Onderzoeks resultaten, discussie en aanbevelingen*. Rapporten van het Instituut voor Natuur- en Bosonderzoek, Brussels. Rapport INBO.R.2008.44.

Everaert, J. 2014. Collision risk and micro-avoidance rates of birds with wind turbines in Flanders. *Bird Study* **61**, 220-230.

Everaert, J., Devos, K. & Kuijken, E. 2002. *Windturbines en vogels in Vlaanderen: Voorlopige onderzoeksreultaten en buitenlandse bevindingen* [Wind turbines and birds in Flanders (Belgium): Preliminary study results in a European context]. Instituut voor Natuurbehoud, Report R. 2002.03, Brussels.

Everaert, J. & Kuijken, E. 2007. *Wind turbines and birds in Flanders (Belgium). Preliminary summary of the mortality research results*. Research Institute for Nature and Forest, Brussels.

Everaert, J. & Stienen, E.W.M. 2006. Impact of wind turbines on birds in Zeebrugge (Belgium). Significant effect on breeding tern colony due to collisions. *Biodiversity Conservation* **16,** 3345-3359.

Everaert, J. & Stienen, E.W.M. 2007. Impact of wind turbines on birds in Zeebrugge (Belgium) Significant effect on breeding tern colony due to collisions. *Biodiversity and Conservation* **16,** 3345-3359.

Fernley, J., Lowther, S. & Whitfield, P. 2006. *A review of goose collisions at operating wind farms and estimation of the goose avoidance rate.* Natural Research Ltd.

Furness, R.W. Wade, H.M., Masden, E.A. 2013. Assessing vulnerability of marine bird populations to offshore wind farms. *Journal of Environmental Management* **119,** 56-66

Garthe, S. & Hüppop, O. 1994. Distribution of ship-following seabirds and their utilization of discards in the North Sea in summer. *Marine Ecology Progress Series*, **106**, 1-9

Garthe, S. & Hüppop, O. 2004. Scaling possible adverse effects of marine wind farms on seabirds: developing and applying a vulnerability index. *Journal of Applied Ecology* **41,** 724-734.

Garvin, J.C., Jennelle, C.S., Drake, D. & Grodsky, S.M. 2011. Response of raptors to a windfarm. *Journal of Applied Ecology*, **48,** 199-209.

Gauthreaux, S.A., Jr. 1991. The flight behavior of migrating birds in changing wind fields: Radar and visual analyses. *American Zoologist* **31,** 187-204.

GoBe Consultants Ltd. 2012. *Gunfleet Sands Offshore Wind farm I & II – Post Construction Year 2 – Marine Licence Environmental Monitoring Report.* Prepared for DONG Energy.

Global Renewable Energy Partners. 2002. *Kentish Flats Offshore Wind farm*. Global Renewable Energy Partners, Egham

Grünkorn, T. , Diederichs, A., Poszig, D., Diederichs, B. & Nehls, G. 2009. Wie viele Vögel kollidieren mit Windenergieanlagen? *Natur und Landschaft* **7,** 309-314.

Gurevitch, J. & Hedges, L.V.1999. Statistical issues in ecological meta-analyses. *Ecology* **80,** 1142-1149.

Hamer, K.C., Humphreys, E.M., Magalhaes, M.C., Garthe, S., Hennicke, J., Peters, G., Grémillet, D., & Wanless, S. 2009. Fine-scale foraging behaviour of a medium-ranging marine predator. *Journal of Animal Ecology* **78,** 880-889.

Hamer, K.C., Monaghan, P., Uttley, J.D., Walton, P. & Burns, M.D. 1993. The influence of food supply on the breeding ecology of kittiwakes *Rissa tridactyla* in Shetland. *Ibis* **135,** 255–263

Hartman, J,C., Krijgsveld. K.L., Poot, M.J.M., Fijn. R.C., Leopold, M.F., and Dirksen, S. 2012. *Effects of birds of Offshore Wind farm Egmond aan Zee (OWEZ)- An overview and intergration of insights obtained*. Bureau Waardenburg BV.

Henderson, I.G., Langston, R.H.W. & Clark, N.A. 1996. The response of Common Terns *Sterna hirundo* to power lines: an assessment of risk in relation to breeding commitment, age and wind speed. *Biological Conservation* **77,** 185-192.

Holmstrom, L. , Hamer, T. E., Colclazier, E., Denis, N., Ruché, D. & Verschuyl, J. 2011. Assessing avian-turbine collision risk: an approach angle dependent model. *Wind Engineering* **35,** 289-312.

Howe, R.W., Evans, W. & Wolf, A.T. 2002. *Effects of wind turbines on birds and bats in northeastern Wisconsin*. Report to Wisconsin Public Service Corporation and Madison Gas and Electric Company.

Howell, J.A. 1997. Avian Mortality at Rotor Swept Area Equivalents, Altamont Pass and Montezuma Hills, California. *Transactions of the Western Section of the Wildlife Society* **33,** 24-29.

Hull Innogy. 2002. *North Hoyle Offshore Wind farm Environmental Statement.*

IECS. 2007 *Collision Risk Assessment: Humber Gateway Offshore Wind farm*, IECS.

Inger, R., Attrill, M.J., Bearhop, S., Broderick, A.C., Grecian, W.J., Hodgson, D.J., Mills, C., Sheehan, E., Votier, S.C., Witt, M.J. & Godley, B.J. 2009. Marine renewable energy: potential benefits to biodiversity? An urgent call for research. *Journal of Applied Ecology* **46,** 1145-1153.

Jackson, D. & Whitfield, P. 2011. *Guidance on survey and monitoring in relation to marine renewables deployments in Scotland, volume 4. birds*. Technical report, Scottish Natural Heritage and Marine Scotland.

Janoska, F. 2012. Investigations of Bird Collisions in 2 Wind farms. *International Scientific Conference on Sustainable Development & Ecological Footprint*, Sopron, Hungary, March 26-27 2012.

Johnson, G.D., Erickson, W.P., Strickland, M.D., Shepherd, M.F., Shepherd, D.A. 2000. *Avian Monitoring Studies at The Buffalo Ridge, Minnesota Wind Resource Area: Results of a 4-year study.* Western EcoSystems Technology Inc., Wyoming.

Johnston, A., Cook, A.S.C.P., Wright, L.J., Humphreys, E.M. & Burton, N.H.K. 2014a. Modelling flight heights of marine birds to more accurately assess collision risk with offshore wind turbines. *Journal of Applied Ecology* **51,** 31-41.

Johnston, A., Cook, A.S.C.P., Wright, L.J., Humphreys, E.M. & Burton, N.H.K. 2014b. Corrigendum. *Journal of Applied Ecology* doi: 10.1111/1365-2664.12260

Kahlert, J., Petersen, I. K., Fox, A. D., Desholm, M. & Clausager, I. 2004. *Investigations of birds during construction and operation of Nysted offshore wind farm at Rødsand*. Annual status report 2003. National Environmental Research Institute (NERI) Kalø, Denmark.

Kolios, S. 2009. *Quantification of the movement characteristics of Lesser Black-backed Gull (Larus fuscus), using high resolution GPS measurements.* MSc. Thesis, University of Amsterdam

Komdeur, J., Bertelsen, J. & Cracknell, G. (eds) 1992. *Manual for aeroplane and ship surveys of waterfowl and seabirds*. IWRB Special Publ. No. 19, National Environmental Research Institute Kalø.

Krijgsveld, K.L., Akershoek, K., Schenk, F., Dijk, F. & Dirksen, S. 2009. Collision risk of birds with modern large wind turbines. *Ardea* 97: 357-366.

Krijgsveld, K.L., Fijn, R.C., Japink, M., van Horssen, P.W., Heunks, C., Collier, M.P., Poot, M.J.M., Beuker, D. & Dirksen, S. 2011. *Effect Studies Offshore Wind farm Egmond aan Zee. Final report on fluxes, flight altitudes and behaviour of flying bird.* Bureau Waardenburg report 10-219, NZW-ReportR_231_T1_flu&flight. Bureau Waardenburg, Culemborg, Netherlands.

Kubetzki, U., & Garthe, S. 2003. Distribution, diet and habitat selection by four sympatrically breeding gull species in the south-eastern North Sea. *Marine Biology*, **143,** 199-207.

Larkin, G.P. & Thompson, D. 1980. Flight speeds of birds observed with radar: evidence for two phases of migratory flight. *Behavioral Ecology & Sociobiology* **7**: 301-317.

Lawrence, E.S., Painter, S. & Little, B. 2007. Responses of birds to the wind farm at Blyth Harbour, Northumberland, UK. In: de Lucas, M., Janss, G.F.E., Ferrer, M. 2007. *Birds and Wind farms Risk Assessment and Mitigation*, Quercus, Madrid.

Leopold, M.F., Camphuysen, C.J., van Lieshout, S.M.J., ter Braak, C.J.F. & Dijkman, E.M. 2004. *Baseline studies North Sea wind farms: Lot 5 marine birds in and around the future site Nearshore Windfarm (NSW)*. Alterra-rapport 1047.

Leopold, M.F., Dijkman, E.M. & Teal, L. 2011. *Local birds in and around the Offshore Wind farm Egmond aan Zee (OWEZ) (T-0 & T-1, 2002-2010)*. NoordzeeWind report OWEZ_R_221_T1_20110915_localbirds_final. Imares / NoordzeeWind, Wageningen / IJmuiden.

Lincs Wind farm Ltd. 2010. *LID6 Environmental Statement*, Lincs Wind farm Ltd.

Lindeboom, H.J., Kouwenhoven, H.J., Bergman, M J.N., Bouma, S., Brasseur, S. Daan, R., Fijn, R.C., de Haan, D., Dirksen, S., van Hal, R., Hille Ris Lambers, R., ter Hofstede, R., Krijgsveld, K.L., Leopold, M. & Scheidat, M. 2011. Short-term ecological effects of an offshore windfarm in the Dutch coastal zone; a compilation. *Environmental Research Letters* **6**: dx.doi.org/10.1088/1748-9326/6/3/035101.

Maar, M., Bolding, K., Petersen, J.K., Hansen, J.L.S. & Timmermann, K. 2009. Local effects of blue mussels around turbine foundations in an ecosystem model of Nysted off-shore wind farm, Denmark. *Journal of Sea Research* **62,** 159-174.

MacArthur Green. 2012. *Note to support the use of a 99% avoidance rate for the Beatrice Offshore Wind farm collision risk modelling*, MacArthur Green Ltd., Glasgow.

Mackenzie, M.L, Scott-Hayward, L.A.S., Oedekoven, C.S., Skov, H., Humphreys, E. & Rexstad E. 2013. *Statistical modelling of seabird and cetacean data: guidance document*. University of St. Andrews contract for Marine Scotland.

MacLean, I.M.D., Wright, L.J., Showler, D.A. & Rehfisch, M.M. 2009. *A review of assessment methodologies for offshore wind farms*. Report commissioned by COWRIE Ltd.

Maclean, I.M.D., Rehfisch, M.M., Skov, H. & Thaxter, C.B. 2013. Evaluating the statistical power of detecting changes in the abundance of seabirds at sea. *Ibis* **155,** 113-126.

Madsen, J., & Boertmann, D. 2008. Animal behavioral adaptation to changing landscapes: spring-staging geese habituate to wind farms. *Landscape ecology* **23**, 1007-1011.

Mainstream Renewable Power. 2012. Neart na Gaoithe Environmental Statement.

Martin, G. R. 2010. Bird collisions: a visual or perceptual problem? *BOU Proceedings – Climate Change and Birds.* http://www.bou.org.uk/bouproc-net/ccb/martin.pdf.

Martin, G.R. 2011. Understanding bird collisions with man-made objects: a sensory ecology approach. *Ibis* **153,** 239-254.

Martin, G.R. & Shaw, J.M. 2010. Bird collisions with power lines: Failing to see the way ahead? *Biological Conservation* **143,** 2695-2702.

Masden, E. A., Fox, A. D., Furness, R. W., Bullman, R. & Haydon, D. T. 2010. Cumulative impact assessments and bird/wind farm interactions: Developing a conceptual framework. *Environmental Impact Assessment Review* **30**, 1-7.

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.

Masden, E. A., McCluskie, A., Owen, E., Langston, R. H. W. 2014. Renewable energy developments in an uncertain world: The case of offshore wind and birds in the UK. *Marine Policy* **51**, 169-172.

Mateos, M., Arroyo, G. M., Rodríguez, A., Cuenca, D. and de la Cruz, A. 2010. Calibration of visually estimated distances to migrating seabirds with radar measurements. *Journal of Field Ornithology*, **81**, 302-309.

May, R., Hoel, P.L., Langston, R., Dahl, E.L., Bevanger, K., Reitan, O., Nygård, T., Peder-sen, H.C., Røskaft, E. & Stokke, B.G. 2010. *Collision risk in white-tailed eagles. Modelling collision risk using vantage point observations in Smøla wind-power plant.* – NINA Report 639. 25 pp.

May, R., Nygard, T., Lie Dahl, E., Bevanger, K. 2013. Habitat utilization in white-tailed eagles (*Haliaeetus albicilla*) and the displacement impact of the Smola wind-power plant. *Wildlife Society Bulletin* **37**, 75-83.

McAdam, B.J., 2005. *A Monte-Carlo model for bird / wind turbine collisions*. MSc Ecology. University of Aberdeen.

MMO. 2014. *Review of post-consent offshore wind farm monitoring data associated with licence conditions*. A report produced for the Marine Management Organisation, pp 194. MMO Project No: 1031. ISBN: 978-1-909452-24-4.

Mendel, B., Kotzerka, J., Sommerfeld, J., Schwemmer, H., Sonntag, N. & Garthe, S. 2014. *Effects of the alpha ventus offshore test site on distribution patterns, behaviour and flight heights of seabirds*. *In Ecological Research at the Offshore Windfarm Alpha Ventus: Challenges, Results and Perspectives.* Editors Federal Maritime and Hydrographic Agency, Federal Ministry for the Environment, Nature Conservation and Nuclear Safety. Springer Spektrum.

Meyers, J. & Meneveau, C. 2012. Optimal turbine spacing in fully developed wind farm boundary layers. *Wind Energy* **15**, 305-317.

Moray Offshore Renewables Ltd (MORL). 2012. *Environmental Statement: Telford, Stevenson, MacColl Wind farms and Associated Transmission Infrastructure*. http://www.morayoffshorerenewables.com/Document-Library.aspx?path=environmental+statement.

Morecambe Wind Ltd. 2006. *West of Duddon Sands Offshore Wind farm Environmental Statement*. Morecambe Wind Ltd., Helsby.

Munsters, C.J.M., Noordervliet, M.A.W. & Ter Keurs, W.J. 1996. Bird casualties caused by a wind energy project in an estuary. *Bird Study* **43,** 124-126.

Natural Power Consultants. 2012. *Analysis of Marine Environmental Monitoring Plan Data from the Robin Rigg Offshore Wind farm, Scotland (Operational Year 1).* Natural Power, Castle Douglas.

Natural Power. 2013. *Responses to the Examining Authority's Second Written Questions – Question 22 Appendix 9 – Moray Offshore Wind farm. Avoidance Rate Study Telford, Stevenson, MacColl Windfarms and Associated Transmission Infrastructure. Ornithology Avoidance Rate Calculations.* Report written for Moray Offshore Renewables Ltd (MORL).

Natural Power. 2014. *Analysis of Marine Ecology Monitoring Plan Data from the Robin Rigg Offshore Windfarm, Scotland (Post-construction Year 3). Draft Technical Report*. E.ON Climate & Renewables.

NIRAS. 2011. *Gunfleet Sands 2 Offshore Wind farms. Year 1 Post-construction Ornithological monitoring*. NIRAS, Cambridge

Norman, J.F., Crabtree, C.E., Clayton, A.M. & Norman, H.F. 2005. The perception of distances and spatial relationships in natural outdoor environments. *Perception* **34**, 1315-1324.

Orloff, S. & Flannery, A. 1992. *Wind Turbine Effects on Avian Activity, Habitat Use, and Mortality in Altamont Pass and Solano County Wind Resource Areas.* Prepared by BioSystems Analysis, Inc., Tiburon, CA, for the California Energy Commission, Sacramento.

Painter, S., Little, B. & Lawrence, S. 1999. *Continuation of birds studies at Blyth Harbour Wind farm and the implications for offshore wind farms*. ETSU Report.

Pearce-Higgins, J.W., Leigh, S., Douse, A. & Langston, R.H.W. 2012. Greater impacts of windfarms on bird populations during construction than subsequent operation: results of a multi-site and multi-species analysis. *Journal of Applied Ecology* **49,** 386-394.

Pendlebury, C. 2006. *An appraisal of 'A review of goose collisions at operating wind farms and estimation of the goose avoidance rate' by Fernley, J., Lowther, S. and Whitfield, P.* BTO Research Report No. 455, Thetford.

Pennycuick, C.J. 1997. Actual and "optimum" flight speeds: field data reassessed. *Journal of Experimental Biology* **200,** 2355-2361.

Percival, S.M., 1998. *Birds and wind turbines*: *managing potential planning issues.* Proceedings of the 20th British Wind Energy Association Conference, 1998. pp. 345-350.

Percival, S.M. 2001. *Assessment of the effects of offshore wind farms on birds.* Consultancy Report to UK Department of Technology and Industry. Ecology Consulting. University of Sunderland, Sunderland.

Percival, S. 2012. *Hellrigg Wind farm: Goose Refuge Monitoring Report Winter 2011-12*, Ecology Consulting, Durham

Percival, S. 2013. *Hellrigg Wind farm: Goose Refuge Monitoring Report Winter 2012-13*, Ecology Consulting, Durham

Pérez Lapeña, B., Wijnberg, K.M., Hulscher, S.J.M.H. & Stein, A. 2010. Environmental impact assessment of offshore wind farms: a simulation-based approach. *Journal of Applied Ecology* **47**: 1110-1118.

Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M. & Fox, A.D. 2006. *Final results of bird studies at the offshore wind farms at Nysted and Horns Rev, Denmark.* National Environmental Research Institute, Denmark.

Petterson, J. 2005. *The impact of offshore wind farms on bird life in Southern Kalmar Sound, Sweden: A final report based on studies 1999-2003.* Swedish Energy Agency.

Pettex, E., Bonadonna, F., Enstipp, M., Siorat, F. & Grémillet, D. 2010. Northern gannets anticipate the spatio-temporal occurrence of their prey. *Journal of Experimental Biology* **213,** 2365-2371.

Pinder, J. M. 1989. Kittiwakes Rissa tridactyla inland in Yorkshire 1981–1985. *Ringing & Migration*, **10**, 113-118.

Plonczkier, P. & Simms, I.C. 2012. Radar monitoring of migrating pink-footed geese: behavioural responses to offshore wind farm development. *Journal of Applied Ecology* **49,** 1187-1194.

Powell, L. A. 2007. Approximating variance of demographic parameters using the delta method: A reference for avian biologists. *Condor* **109**, 949-954.

Robinson, R.A. (2005) *BirdFacts: profiles of birds occurring in Britain & Ireland*. BTO Research Report 407. BTO, Thetford (http://www.bto.org/birdfacts, accessed on 12/06/2014)

Rothery, P., Newton, I. & Little, B. 2009. Observations of seabirds at offshore wind turbines near Blyth in northeast England. *Bird Study* **56,** 1-14.

Richardson, W.J. 1990. Timing of bird migration in relation to weather: updated review. In: *Bird Migration: the Physiology and Ecophysiology* (ed. Gwinner, E.), pp. 78-101. Springer-Verlag, Berlin.

RPS. 2005. *London Array Environmental Statement*, London Array Ltd., Coventry.

RPS. 2008. *Gunfleet Sands Monitoring Report*, RPS, London.

RPS. 2011. *Galloper Wind farm Project Environmental Statement – Technical Appendices 2: Appendix 4: Greater Gabbard post-construction vantage point surveys*, RPS, Glasgow.

RPS 2013. *Inch Cape Offshore Wind farm Environmental Statement*, RPS, Edinburgh.

RWE Npower Renewables. 2005. *Gwynt Y Mor Offshore Wind farm Environmental Statement*.

RWE Npower Renewables. 2011. *Galloper Wind farm Project Environmental Statement*, Galloper Wind farm Ltd.

Rexstad, E. 2011. *DSM: density surface modelling of distance sampling data*. R package version 1.0.5.

Safi, K., Kranstauber, B., Weinzierl, R., Griffin, L., Rees, E., Cabot, D., Cruz, S., Proaño, C., Takekawa, J. Y., Waldenström, J., Bengtsson, D., Kays, R., Wikelski, M. & Bohrer, G. 2013. Flying with the wind: scale dependency of speed and direction measurements in modelling wind support in avian flight. *Movement Ecology* **1**: 4 doi:10.1186/2051-3933-1-4.

Scira Offshore Energy Ltd. 2006. *Sheringham Shoal Offshore Wind farm Environmental Statement*.

Seascape Energy Ltd. 2002. *Burbo Bank Offshore Wind farm Environmental Statement*. Seascape Energy Ltd., Lancashire, http://www.burbobank.co.uk/en/environment/environmental-statement

Scottish Natural Heritage (SNH). 2010. *Use of avoidance rates in the SNH wind farm collision risk model. SNH Avoidance Rate Information & Guidance Note*. Scottish Natural Heritage, Inverness, UK. http://www.snh.gov.uk/docs/B721137.pdf.

Scottish Natural Heritage (SNH). 2013. *Avoidance Rates for wintering species of geese in Scotland at onshore wind farms*. Scottish Natural Heritage, Inverness, UK. http://www.snh.gov.uk/docs/A916616.pdf .

Schwemmer, P., & Garthe, S. 2005. At-sea distribution and behaviour of a surface-feeding seabird, the lesser black-backed gull Larus fuscus, and its association with different prey. *Marine Ecology Progress Series* **285,** 245-258.

Shamoun-Baranes, J. & van Loon, E. 2006. Energetic influence on gull flight strategy selection. *Journal of Experimental Biology* **209,** 3489-3498.

Shamoun-Baranes, J., van Gasteren, H., van Belle, J. van Loon, E., Bouten, W. & Buurma, L. 2006. A comparative analysis of the influence of weather on the flight altitudes of birds. *Bulletin of the American Meteorological Society* **87,** 47-61.

Skov, H., & Durinck, J. (2001). Seabird attraction to fishing vessels is a local process. *Marine Ecology Progress Series*, *214*, 289-298.

Skov H., Leonhard, S.B., Heinänen, S., Zydelis, R., Jensen, N.E., Durinck, J., Johansen, T.W., Jensen, B.P., Hansen, B.L., Piper, W. & Grøn, P.N. 2012. *Horns Rev 2 Monitoring 2010-2012. Migrating Birds*. Orbicon, DHI, Marine Observers and Biola. Report commissioned by DONG Energy.

Smales, I., Muir, S., Meredith, C., Baird, R., 2013. A Description of the Biosis Model to Assess Risk of Bird Collisions with Wind Turbines. *Wildlife Society Bulletin* **37,** 59-65.

Smallwood, K.S. & Thelander, C. 2008. Bird mortality in the Altamont Pass wind resource area, California. *Journal of Wildlife Management* **72,** 215-223.

Smartwind. 2013. *Hornsea Offshore Wind farm Environmental Statement*, Smartwind Ltd., London http://infrastructure.planningportal.gov.uk/wp-content/ipc/uploads/projects/EN010033/2.%20Post-Submission/Application%20Documents/Environmental%20Statement/7.5.5.1%20Ornithology%20Technical%20Report.pdf.

Spear, L.B. & Ainley D.G. 1997. Flight speed of seabirds in relation to wind speed and direction. *Ibis* **139,** 234-251.

Stephens, D.W. & Krebs, J. R. 1986. *Foraging theory*. Princeton University Press.

Stewart, G. 2010. Meta-analysis in applied ecology. *Biology Letters* **6,** 78-81.

Stewart-Oaten, A., Murdoch, W.M. & Parker, K.R. 1986. Environmental impact assessment: 'pseudoreplication' in time? *Ecology* **67,** 929-940.

Stienen, E. W., Van Beers, P. W., Brenninkmeijer, A., Habraken, J. M., Raaijmakers, M. H., & Van Tienen, P. G. 2000. Reflections of a specialist: patterns in food provisioning and foraging conditions in Sandwich Terns Sterna sandvicensis. *Ardea*, **88**, 33-49

Still, D., Little, B., Lawrence, S. 1996. *The effect of wind turbines on the bird population at Blyth Harbour*. ETSU Report

Talisman Energy (UK) Ltd. 2006. *Beatrice Wind farm Demonstrator Project Environmental Statement*, Talisman Energy (UK) Ltd., Aberdeen

Tasker, M.L., Jones, P.H., Dixon, T.J. & Blake, B.F. 1984. Counting seabirds at sea from ships: a review of methods employed and a suggestion for a standardized approach. *Auk* **101,** 567-577.

Technip UK Ltd. 2012. *Aberdeen Offshore Wind farm Ornithological Baseline and Impact Assessment Addendum.* http://www.vattenfall.co.uk/en/file/AOWFL_ES_AddendumJune_12__Revised_Ornithological_Baseline___Impact_Assessment.pdf_21927083.pdf.

Thanet Offshore Wind farm Ltd. 2005. *Thanet Offshore Wind farm Environmental Statement*. Thanet Offshore Wind farm Ltd., Warwickshire.

Thelander, C.G., Smallwood, K.S. & Rugge, L. 2003. *Bird risk behaviours and fatalities at the Altamont Pass Wind Resource Area*. National Renewable Energy Laboratory, Colorado.

Trinder, M. (2012) *Note to support the use of a 99% avoidance rate for the Beatrice Offshore Wind farm collision risk modelling*, MacArthur Green Ltd., Glasgow

Triton Knoll Offshore Wind farm Ltd. 2011. *Triton Knoll Environmental Statement*. Triton Knoll Offshore Wind farm Ltd., Swindon.

Tucker, V.A. 1996. A mathematical model of bird collisions with wind turbine rotors. *Journal of Solar Energy Engineering* 253-262.

Tucker, V.A. & Schmidt-Koenig, K. 1971. Flight speeds of birds in relation to energetics and wind directions. *Auk* **88,** 97-107.

Vanermen, N., Stienen, E.W.M., Courtens, W., Onkelinx, T., Van de walle, M. & Verstraete, H. 2013. *Bird monitoring at offshore wind farms in the Belgian part of the North Sea - Assessing seabird displacement effects*. Rapporten van het Instituut voor Natuur- en Bosonderzoek 2013 (INBO.R.2013.755887). Instituut voor Natuur- en Bosonderzoek, Brussel.

Vanermen, N., Stienen, E.W.M., Onkelinx, T., Courtens, W., Van de walle, M., Verschelde, P. & Verstraete, H. 2012. Seabirds & Offshore Wind Farms: Monitoring Results 2011. pp. 85-109. In: Degraer, S., Brabant, R. & Rumes, B., (Eds.) Offshore wind farms in the Belgian part of The North Sea: Heading for an understanding of environmental impacts. Royal Belgian Institute of Natural Sciences, Management Unit of the North Sea Mathematical Models, Marine Ecosystem Management Unit.

Votier, S.C., Bicknell, A., Cox, S.L., Scales, K.L., Patrick, S.C. 2013. A bird's eye view of discard reforms: bird-borne cameras reveal seabird/fishery interactions. *PLoS One* **8,** e57376.

Wakeling, J. M., & Hodgson, J. 1992. Optimisation of the Flight Speed of the Little, Common And Sandwich Tern. *Journal of experimental biology*, **169**, 261-266.

Whitfield, D.P. (2008). *Review of evidence presented to support conclusions drawn during offshore wind farm application process: Appropriate Assessment for Sheringham Shoal proposed wind farm.* Consultants' Report by Natural Research Ltd., Banchory, Aberdeen to Marine and Fisheries Agency

Whitfield, D.P. 2009. *Collision avoidance of golden eagles at wind farms under the 'Band' collision risk model.* Report from Natural Research to Scottish Natural Heritage, Banchory, UK.

Whitfield, D.P. & Madders, M. 2006a. *A review of the impacts of wind farms on hen harriers* Circus cyaneus *and an estimation of collision avoidance rates*. *Unpublished report, Natural Research Ltd, Banchory, Aberdeenshire, Scotland*.

Wild Frontier Ecology. 2013. *Kessingland Wind farm Annual Post-construction Monitoring Report Year 2.* Wild Frontier Ecology, Norfolk.

Williams, T.C., Williams, J.M. & Klonowski, T.J. 1974. *Flight patterns of birds studied with an Ornithar*. In: Proc. A Conf. on the Biol. Aspects of the Bird/Aircraft Collision Problem. S. Gauthreaux (ed.), p. 477-490. Washington, DC, U.S.A.F. Office of Scientific Research.

Wilson, J. C. & Elliot, M. 2009. The habitat-creation potential of offshore wind farms. *Wind Energy* **12,** 203-212.

Winkelman, J.E. 1990. *Nocturnal collision risks for and behaviour of birds approaching a rotor in operation in the experimental wind park near Oosterbierum, Friesland, The Netherlands; English summary.* Rijksinstituut voor Natuurbeheer, Arnheim. RINRapport 90/17.

Winkelman, J.E. 1992. The impact of the Sep Wind Park near Oosterbierum (Fr.). *The Netherlands, on birds, 1: Collision victims*. RIN Rep. 92/2. Rijksinstituut voor Natuurbeheer, Arnhem, The Netherlands (Dutch, English summary).

Zehnder, S., Åkesson, S., Liechti, F. & Bruderer, B. 2001. Nocturnal autumn bird migration at Falsterbo, South Sweden. *Journal of Avian Biology* **32,** 239-248.

Zielinski, P., Bela, G. & Marchlewski, A. 2008. *Report on monitoring of the wind farm impact on birds in the vicinity of Gniezdzewo (gmina Puck, pomorskie voivodeship).*

Zielinski, P., Bela, G. & Marchlewski, A. 2010. *Report on monitoring of the wind farm impact on birds in the vicinity of Gniezdzewo (gmina Puck, pomorskie voivodeship).*

Zielinski, P., Bela, G. & Marchlewski, A. 2011. *Report on monitoring of the wind farm near Gniezdzewo impact on birds (gmina Puck, pomorskie voivodeship).*

Zielinski, P., Bela, G. & Marchlewski, A. 2012. *Report on monitoring of the wind farm near Gniezdzewo impact on birds (gmina Puck, pomorskie voivodeship).*

APPENDIX 1 Evidence review macro-response – barrier effect studies

A1.1 Egmond aan Zee

Location / habitat

Marine, 10-18km offshore

Turbine / array specification

Turbine array consists of 36 Vestas V90 3 MW turbines covering an area of 27 km^2. Distances between turbines are 650 m within rows and 1000 m between rows. Turbine specifications given as hub height 70 m; rotor diameter 90 m; rotor altitude min 25 m (above mean sea level) and max rotor altitude 115 m (above mean sea level).

Case study number 1

Krijgsveld, K.L., Fijn, R.C., Japink, M., van Horssen, P.W., Heunks, C., Collier, M., Poot, M.J.M., Beuker, D. and Dirksen, S. 2011. *Effect studies offshore windfarm Egmond aan Zee: Final report on fluxes, flight altitudes and behaviour of flying birds*. Bureau Waardenburg Report No. 10 - 219.

Lindeboom, H.J., Kouwenhoven, H.J., Bergman, M.J.N., Bouma, S., Brasseur, S., Daan, R., Fijn, R.C., de Haan, D., Dirksen, S., van Hal, R., Hille Ris Lambers, R., ter Hofstede, R., Krijgsveld, K.L., Leopold, M. & Scheidat, M. 2011. Short-term ecological effects of an offshore windfarm in the Dutch coastal zone; a compilation. *Environmental Research Letters* **6,** doi:10.1088/1748-9326/6/3/035101.

Methods

Krijgsveld *et al.* (2011) focussed on the disturbance of flight paths otherwise referred to as barrier effects. Whereas what was termed as the disturbance of locally resting and/or feeding birds were covered by another project (Leopold *et al.* 2011) as birds recorded on the water. Lindeboom *et al.* (2011) reported the impacts of the windfarm on a range of taxonomic groups but with respect to birds focussed on barrier effects, displacement effects and attraction. As the results presented in Lindeboom *et al.* (2011) were based on the preliminary results of Krijgsveld *et al.* (2011), cited as Krijgsveld *et al.* (2010), this paper is not considered further here.

Data collection was carried out during the post-construction period only.

Radar: Horizontal radar was used to record flight paths, with the radar located on a meteorological mast 500 m from the nearest turbine at the south western side of the windfarm). The radar was set to scan up to distances of 5.6 km from the meteorological mast (although it was calculated that gulls could be detected up to shorter distances of 4.5 km). There was no coverage from the angles of 155° to 220° relative to the mast however).

The radar signal was processed and recorded by Merlin (DeTect Inc). Flight paths of birds or groups of birds were visualised in QuantumGIS and grid cells (750 m x 750 m) were set up in order to analyse both the numbers of tracks and flight directions. In order to mitigate for reduced detection of tracks, due to the presence of turbines and decreasing detection rates with increasing distance from the radar, correction factors were applied to the numbers of tracks recorded inside the windfarm.

Visual and auditory observations: Panorama scans from the meteorological mast consisting of hourly 360° scans to record all birds flying within sight of the observation platform. This information was then used to calibrate the radar counts and provided information on species composition, density, flight altitude and flight direction. Additional information was collected at night and included moon watches, call registration by ear, and call registration by an automated bird call recording system. In addition, the opportunistic recording of flight paths of individual birds or bird groups (picked up either visually using a binoculars or a telescope) or on the radar) was carried out.

Study period

Radar: Continuous recording through the period of April/May 2007 to 31 May 2010. Flight path data was obtained for 817 days (out of a possible 918 due to factors such as high winds).

Visual observations: A total of 405 panoramic scans were carried out over 53 days (dawn to dusk) spread throughout the period of Feb 2006 to Dec 2009 and six nights (dusk to dawn) during spring and autumn migration (October 2008 to April 2009). Opportunistic observations of flight paths were carried between and during panoramic scans (n = 666 flight paths of 85 species were recorded with great cormorant (n = 82) and northern gannet (81) being the most commonly observed).

Species

Local seabirds (gull spp, northern gannet, scoter spp, and auks spp); migrating seabirds (diver spp and scoter spp) and migrating non-marine birds (thrushes and geese spp).

Conditions data collected under

Radar: all conditions.

Visual observations: recording carried out in generally dry, relatively calm conditions (all but day had one Beaufort scale of less than 5) and with a range of visibility conditions (0 - 50 km).

Results

Macro-responses (which were regarded by this study as being due to barrier effects), referred to in the report as macro-avoidance rates, were quantified by two methods[2]:

i. Panoramic scans were used to derive the proportion of birds within, at the edge and outside the windfarm. Using the combined values of the first two groupings, it was possible calculate the % of birds that passed through the windfarm[3]. The resulting values were corrected for relative surface area for within and outside the windfarm and then used to derive macro-avoidance rates[4] for northern gannet = 0.64 (n = 282 birds[5]), sea ducks/scoters spp = 0.71 (n = 123 birds), diver spp = 0.68[6] and alcid spp = 0.68[3]. Sample sizes were too small for other species/species groups for values to be derived and, hence, values have to be derived by other means;

ii. Flight path data collected by radar showed that the number of all birds that flew within the windfarm was on average 72% of the numbers outside the windfarm. This was proposed to equate to an average macro-avoidance rate of 0.28 of birds in relation to the windfarm, and when broken down according to time of year, the values ranged from an average of 0.18 (in winter) and 0.34 (in autumn)[7]. For gull spp and great cormorant, the average avoidance rate in winter of 0.18 was used, as the species composition was heavily dominated by those birds at that time of year (as shown by the visual observations). The overall average avoidance value of 0.28 was assumed for grebe spp, tubenoses spp, skua spp, and tern spp (in the absence of other available data or rationale). It was also shown using radar that the percentage of birds flying in the windfarm was significantly higher during the day compared to night (when data from spring was excluded) and these differences were greatest during summer and winter. Hence avoidance was argued to be higher at night.

Results of the opportunistic recording of flight paths indicated deflection rates of 89% for northern gannet and 40% for gulls spp based on sample sizes of 38 and 78 birds respectively[8]. These values were not considered by the authors to provide evidence for macro avoidance (Karen Krijgsveld *pers. comm.*) however.

There was inherent variation in flight direction as recorded by radar with higher variability recorded winter and summer (probably due to the inclusion of locally foraging birds which are less likely to have a consistent flight trajectory than birds migrating through the area) and during the day. Nevertheless, adjustment of flight paths occurred at 750 - 1,500 m from the windfarm when there was a pronounced

[2] Table 15.1- Krijgsveld *et al.* 2011.
[3] Table 9.3 - Krijgsveld *et al.* 2011.
[4] Macro-avoidance = 100-((x/50)*100). Where x = % of birds that passed through the windfarm and 50 is the correction factor for surface area. Karen Krijgsveld *pers. comm.* Values of x for northern gannet and common scoter were 18 and 14 respectively (sum of the relative abundance inside and at the edge of the windfarm – see Table 9.3).
[5] Taken from Figure 9.25.
[6] Based on the average of northern gannet (0.64) and scoter spp (0.71) which was justified on the grounds of their avoidance behaviour being similar (based on their flight paths).
[7] Figure 9.15 - Krijgsveld *et al.* 2011.
[8] Table 9.6 - Krijgsveld *et al.* 2011

change in flight direction. This was largely based on plots of the mean ± standard errors of flight direction in relation to distance according to season and time of day[9]. The reported changes at 750-1500 m appear to occur before and after midnight in the spring and at dusk during autumn. There were also changes in flight direction at distances further away from the windfarm but these are not highlighted – notably in spring, for most times of day, at distances between 4,500 and 5,250 m.

Numbers of birds were also shown to be highest at 750 - 1500 m, which was taken as evidence of flying birds building up as they were deflected away from the windfarm (also confirmed by visual observations of birds). Moreover, the number of tracks for all seasons in the grid cells circa 750 m from the windfarm was also shown to be significantly higher than the number of tracks for the grids cells containing the adjacent single row of turbines[10].

Assessment of methodology

The values of macro-avoidance derived from the panoramic scans were species specific and were collected in a systematic manner. As for all visual observations, data collection was mostly restricted to days of reasonable visibility and calm conditions.

Macro-avoidance rates (barrier effects) derived using radar were based on mean values across all species and should be interpreted very carefully since there is likely to be variability in response rates between species. Hence this should be borne in mind when citing values derived for gull spp, grebe spp, tubenoses spp, skua spp, and tern spp. It is also unclear whether the actual numbers reported will consist of solely individual birds or whether flocks of birds may have been inadvertently included. Hence as for most radar studies, the avoidance rates cannot be necessarily assumed to correspond to those of individual birds. It is also worth bearing in mind, that the way these data have been collected (comparison of number of flight paths inside and outside the windfarm) could also be potentially considered to be evidence of displacement.

It is also problematic to overlay the arbitrarily selected boundary of 500 m buffer surrounding the outermost turbines used to delineate inside (micro and meso) and outside (macro) the windfarm avoidance (section 3.5) with the grid cell system of 750 km^2 used to analyse the number of tracks.

The grid cell system also does not correspond exactly to the boundaries of the windfarm and hence some cells will overlay areas inside and outside the windfarm which could be an issue for the values cited for % of tracks inside and outside the tracks.

A1.2 Horns Rev

Location / habitat

[9] Figure 9.28- Krijgsveld *et al.* 2011.
[10] Generalised Linear Model (t_{2228} =3.4, $p < 0.001$) - Krijgsveld *et al.* 2011.

Horns Rev 1: Marine, 14 km offshore.
Horns Rev 2: Marine, 30 km offshore.

Turbine /array specification

Horns Rev 1: Turbine array consists of 80 2.0 MW Vesta turbines. Distance between turbines – north to south (560 m) and east to west (560 m). Turbine specifications given as: hub height 70 m; rotor blade length 40 m (diameter 80 m); and total height 110 m. Height of the lowest tip of rotor blade.

Horns Rev 2: Turbine array consists of 91 turbines. Distance between turbines – north to south (560 m) and east to west (560 m). Turbine specifications given as: hub height 68 m; rotor diameter 93 m; and total height 114.5 m. Height of the lowest tip of rotor blade 21.5 m.

Case study number 1

Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M. & Fox, A.D. 2006. *Final results of bird studies at the offshore windfarms at Nysted and Horns Rev, Denmark*. Report commissioned by DONG Energy and Vattenfall A/S. National Environmental Research Institute.

Methods

This report focussed on barrier effects, displacement effects, physical changes to the habitat and collision risk. Work was carried out at the Horns Rev 1 and Nysted offshore windfarms but there were differences in methodology and timing of data collection in relation to the development phase – data collection was carried out during the post-construction period only at Horns Rev 1.

Radar observations*:* Recordings by radar occurred in a circular area of radius ca. 11 km (no coverage in the north east quadrant with the exception of late November 2005). The radar was located on a transformer station located less than 0.6 km from the windfarm. Migration mapped by tracing course of flocks onto a transparency and subsequently digitised. As fewer tracks were recorded both within and beyond the windfarm, due to presence of the turbines and the increasing distance from the radar, densities of tracks were not used to quantitatively to look at barrier effects.

All tracks (n = 468 north of the windfarm and n = 342 east of the windfarm) which were deemed to have a theoretical chance of entering the windfarm were selected using the criteria that they were orientated towards the windfarm at distances between 1.5 and 2 km from the windfarm and had lengths of tracks greater than 2 km.

In order to look at the lateral (horizontal) change in migration route in response (where avoidance occurs) to the windfarm, two sets of transects lines were set up. The first were located east of the windfarm running parallel to the direction of the rows of turbines (from north to south) and were set up at intervals of 50, 100, 150, 200, 250, 300, 400, 500, 1000, 2000, 2500, 3000, 3500 and 4000 m (max. range set

by limits of the radar). The second were set up north of the windfarm at 50, 100, 200, 300, 400, 500, 1000, 1500, 2000, 2,500, 3000 m and then at intervals of 1000 m until 7000 m. The orientation of all bird tracks that intersected two adjacent transects were calculated for all of the transects running east and north of the windfarm.

Visual observations: four transects from the transformer station set up, one of which passed diagonally through the windfarm.

Study period

Radar observations: A total of 17 survey periods (shortest = 5 h 30 min, longest = 39 h 30 min) were carried out covering the periods of August to November 2003; March to May 2004; August to September 2004; March to May 2005; and August to November 2005. Total of 243 h 45 min of observations.

Visual observations: 19 surveys (shortest = 7 h 0 min, longest = 29 h 30 min) were carried out covering the periods of April to May 2003; August- November 2003; March to May2004; August to September 2004; March to May 2005; and August to November 2005. Total of 403 h 18 min of observations.

Species

Staging and migrating birds. Based on visual observations of birds during transect counts, likely to consist primarily of diving ducks (by an order of magnitude higher than any other group and consisting almost exclusively of common scoter), gulls (herring gull, little gull, greater back-backed gull and black-legged kittiwake and terns (Sandwich tern and common/Arctic tern)[11].

Conditions data collected under

During day and night, weather conditions not presented.

Results

The annual percentage of bird tracks (based on the years 2003, 2004 and 2005) entering the windfarms from either the northern or the eastern side of the windfarm ranged from 13.6 % (2005, north of windfarm) and 29.3% (2004, east of the windfarm[12]). The number of tracks that these percentages are based upon are relatively small however (ranging from 12 to 39 tracks). These values appear to provide the origins of cited macro-avoidance rates of 0.71 and 0.86. Spring and autumn periods were not differentiated between as it was argued that bird behaviour would be similar regardless of the time of season.

The mean orientation of tracks of migrating birds, as calculated for all intervals between transects, was used as the response variable to look at the lateral deflection of south bound tracks for birds north (n = 2108) and east of the windfarm

[11] Table 48 - Peterson *et al.* 2006.
[12] Table 55 - Peterson *et al.* 2006.

(n = 1168). For birds north of the windfarm during southbound bird migration, analyses[13] showed that distance to windfarm, wind direction (crosswinds), time of day and the interaction between distance and time of day were significant. Plots of the mean flight orientation with distance to windfarm in relation to time of day wind direction showed that deflections were most pronounced at distances of less than 400 m from the windfarm and that changes at larger distances (<2 km) were more obvious during the daytime compared to the night time period[14]. For birds east of the windfarm analyses[15] found that distance had a significant effect on the orientation of the birds (wind direction, time of day and the interaction between distance to windfarm and wind direction were also significant. Plots of the mean flight orientation with distance to windfarm in relation to time of day wind direction showed that deflections were most pronounced at distances of less than 500 m from the windfarm. Changes in orientation occurred up to 4 km from the windfarm during south bound migrations notably during the day in westerly winds[16].

Assessment of methodology

The derived macro-avoidance rates (based on barrier effects) are a mean value for all birds which occurred during the study and according to visual observations consisted mainly of common scoter. Therefore, these reported avoidance rates may have limited applicability to the less commonly recorded gulls spp and tern spp. In addition these avoidance rates are based on relatively small sample of tracks. Moreover, tracks do not differentiate between individuals or flocks, therefore the reported macro-avoidance rates do not respond to the level of individual birds.

Case study number 2

Blew, J., Hoffman, M., Nehls, G. & Hennig, V. 2008. *Investigations of the bird collision risk and the responses of harbour porpoises in the offshore windfarms Horns Rev, North Sea, and Nysted, Baltic Sea, in Denmark. Part 1: Birds*. Report from the University of Hamburg and BioConsult SH, 145pp.

Methods

The report focussed on the collision risk to migrating birds at Horns Rev 1 and Nysted offshore windfarms and the same methodology was used at both sites.

Blew *et al.* (2008) proposed that avoidance occurred at the three broad scales of: (1) large scale avoidance >2000 m; (2) medium to small scale avoidance 1000 m to 150 m and either horizontally or vertically as measured directly (reactions) or indirectly (comparison of numbers or flight altitudes); (3) last second avoidance. Thus, the

[13] ANOVA analyses: distance F_{14}=18.93, $p < 0.0001$; wind direction F_1=57.49, $p < 0.0001$; time of day F_1=95.33, $p < 0.0001$; and distance*time of day $F_{14} = 3.27$, $p < 0.0001$)- Peterson *et al.* 2006.
[14] Figure 170 - Peterson *et al.* 2006.
[15] ANOVA analyses: distance F_{14}=25.38, $p < 0.0001$; wind direction F_1=13.37, $p = 0.0003$; time of day F_1=132.67, $p < 0.0001$; and distance*wind direction $F_{14} = 2.79$, $p = 0.0004$) - Peterson *et al.* 2006.
[16] Figure 172 - Peterson *et al.* 2006.

second category, which was the focus of this report, overlaps with the definitions in section 3 of this report of both macro- and meso-avoidance.

Data collection was carried out during the post-construction period only.

***Radar observations**:* Horizontal radar (Bridgemaster E-series and Pathfinder) was deployed from ships with a range of anchoring sites (three, four and four at the eastern, southern and western edges of the windfarm respectively) at distances of 150 to 300 m to the windfarm. Screenshots were captured using a digital camera for the horizontal radar and the angle of tracks and their length were also registered. The range of the radar was set to 1.5 nautical miles. No manual tracking of signals on the horizontal radar was carried out which meant that changes in flight trajectories for individual tracks could not be looked at.

Radar tracks were categorised according to their direction in relation to the first row of the windfarm; flying towards (± 45° either side of perpendicular to the windfarm; flying away; and flying parallel (more or less).

In order to look at lateral avoidance, four intervals ranging from 0-500 m, 500-1,000 m, 1,000-1,500 m and 1,500-2,000 m in relation to the ship and the relative orientation of tracks were recorded in the range of ± 90° with 0° being perpendicular to the windfarm. Due to sample size issues (insufficient number of tracks), it was not possible to report results for Horns Rev, however.

Visual observations: Visual observations were carried out along a 2 km transect which ran perpendicular to the outer edge of the windfarm, with the ship located halfway along it length. On the windfarm side of the transect, the gap between the edge of windfarm as defined by the row of the outer turbines (approximately 300 m from the ship) to 700 m inside the windfarm (or 1,000 m from the ship) was regarded as being inside the windfarm. On the corresponding non-windfarm side, the transect which was between 300–1,000 m from the ship was regarded as being outside the windfarm (in relation to the windfarm this represents a distance of between 600 and 1,300 m). Collectively these were termed as Class A, whereas the transect up to 300 m either side of the ship was Class B (excluding birds within 30 m either side of the ship which were disregarded). Visual observations of flying birds (optics only used for identification purposes) were carried out every half hour for observation periods of 15 minutes from sunrise to sunset. Distance, flight direction and altitude were recorded (classes were largely defined by the upper and lower limits of the rotor blade: 0-5 m: 5-30 m; 30-100 m; >110 m). The results of this work are not considered further here.

Visual observations were carried out for 219.5 and 238.5 h in 2005 and 2006 respectively.

Study period

March to May to coincide with spring migration (27.5 observation days in 2005 and 2006) and September to November to cover autumn migration (39 observation days in 2005 and 2006).

Radar appeared to have been run continuously.

Species

Seaducks, geese, gulls and terns and wide range of songbird species.
Transect counts showed that gulls (many of which were unidentified to the species level) were the most common group recorded in both spring and autumn (with little gull notably more common in the former time of year). Common scoter were also common but more so in spring.

Conditions data collected under

Horizontal radar observations were limited to calm sea state conditions (wind speed < 2 ms^{-1}) and generally dry weather.

Visual observations were stopped when visibility <1 km but visual and acoustic observations were possible for all observation days

Results

During the day, the overall number of tracks flying parallel to the windfarm was higher (n = 1,045) compared to flying away from (n = 486) or towards (n = 386) the windfarm. This pattern was less pronounced at night with the number of birds parallel to the windfarm (n = 253) being only marginally higher compared to flying away from the windfarm (n = 206) but were higher than towards the windfarm (n = 101).

Although the visual observations were designed primarily to look at the differences in flight height distribution, they were able to provide supporting evidence for macro avoidance occurring. For northern gannet, out of 66 gannets recorded only 2 flew within the windfarm. For both little gull and all gull spp (excluding little gull), significantly less birds were present inside the windfarm.

Assessment of methodology

Results from the observations from horizontal radar were limited as only 5% (9% for Nysted) of the observation time yielded screenshots which could be used and these were biased to daytime periods. There was also the additional problem that detection within the windfarm was considerably lower compared to outside due to the presence of the wind turbines (tracks were observed to disappear and reappear when entering and leaving the windfarm).

There were several limitations with working on a ship compared to from land or a fixed platform, including rough sea conditions, which would likely hamper data collection. There were also issues associated with the tidal cycles (particularly at Horns Rev, less so at Nysted) and strong winds which could result in the ship turning and this affected the radar data collected. Another potentially confounding factor is that the ship could also act as an attractant to some species of seabirds (e.g. gull spp) or potentially act as a disturbance to others (e.g. diver spp and duck spp).

In terms of demonstrating macro-avoidance, horizontal radar was unable to provide quantitative evidence. Avoidance appeared to be implied by the percentage of birds flying parallel being higher than those values reported for birds flying towards and away from the windfarm and this pattern was more pronounced during the day when the windfarm was more visible. The significance of birds tracks running parallel to as opposed to being orientated towards or away from the windfarm was not explained, however, and there was a lack of pre-construction information to make comparisons with. There was also insufficient data to look at potential changes in the orientation of tracks (but enough data was available for Nysted – see section 5.4.4). Similarly the visual observations did not provide quantitative evidence of macro-avoidance rates.

Case study number 3

Skov H., Leonhard, S.B., Heinänen, S., Zydelis, R., Jensen, N.E., Durinck, J., Johansen, T.W., Jensen, B.P., Hansen, B.L., Piper, W. & Grøn, P.N. 2012. *Horns Rev 2 Monitoring 2010-2012. Migrating Birds*. Orbicon, DHI, Marine Observers and Biola. Report commissioned by DONG Energy.

Methods

This report focussed on migrating birds in relation to Horns Rev 1 and 2.

Radar observations: Horizontal radar was used from observation stations located to the north east of Horns Rev 1 (assumed to be the same as used in previous studies at Horns Rev 1, 560 m distance to the windfarm) and to the east of Horns Rev 2 (no distance provided but estimated to be less than 2 km away). Radar range was set at 6.0 km and covered a circular area. Additional information on species identification was possible by use of "a real-time tracking" procedure whereby tracks of individual birds or tracks could be followed on background images to produce videos. Videos were produced using a frame grabber connected to the radar and tailor made software provided the video as a back ground image on the PC screen. Whilst one observer followed the trace on the screen, a second attempted to locate the target in the field using a binocular or telescope to provide names, number of birds and altitude. Identification on tracks was not always possible during busy periods. Track densities were estimated for a 100 m^2 grid system within the radius of the radar.

Laser range finders: Laser range finders (Vectronix 21 Aero) were also used from the observation stations used to collect species-specific data up to distances of 2-3 km for large bird species (depending on the field of view and flight mode of the bird). Positions and altitudes of birds were logged automatically via GPS recorded at intervals of 10-15 sec. Data from the laser range finders were used to supplement data collected by the radar. Calibrations in order to correct the readings provided by the GPS were necessary due to interference by the observation tower.

Track data for range finders and radar were also integrated with weather data including wind direction, wind speed, air pressure, clearness, humidity, total precipitation and air temperature. In addition, the relative flight direction of the bird in relation to wind direction was also calculated.

Generalised Additive Models (GAMs) with a Tweedie distribution were used to look at track densities derived by radar for all bird tracks and common scoter tracks in relation to distance to the radar and distance to the windfarm. Generalized Additive Mixed Models (GAMMs) with a correlation structure (to deal with spatial and temporal autocorrelation) were used to look at the flight altitude in relation to weather variables and distance to the nearest wind turbine. However, this information could not be used to quantify an avoidance rate.

Study period

Data collection carried out during spring and autumn from September 2010 to May 2012. No further details given.

Radar observations: 15 min per h during daylight.

Laser range finders: operated permanently with observation periods of a minimum of 15 min per h.

Species

All spring and autumn migrants (seabirds, water birds, ducks and passerines).

Conditions data collected under

Not specified.

Results

Tracks recorded by both horizontal radar and the laser range finders were mapped for a range of species/groups in order to visualise movement patterns. It was proposed that diver spp (small sample size), northern gannet and common scoter tended to migrate along corridors along the periphery of the windfarms, although looking at the maps provided it is clear that northern gannet[17] and common scoter[18] did occur within the windfarms, notably Horns Rev 2. This was thought to be a result of the bathymetry as common scoters seemed to associate with waters less than 10 m in depth.

At Horns Rev 2 both distance to radar and distance to the windfarm were significant predictors of the densities for all birds tracks combined [19] and common scoter tracks. Response curves[20] produced by the models were similar for both analyses, which was unsurprising given the relative proportion of all tracks that were from common scoter. A peak in the density of birds occurred at around 1,500-2,500 m from the windfarm and was argued to provide evidence for a barrier effect due to birds

[17] Figure 5-14- Skov *et al*. 2012.
[18] Figure 5-15- Skov *et al*. 2012.
[19] GAM; Distance to radar $F=321.5$, $p < 0.01$ and distance to windfarm $F=286.4$, $p < 0.01$. Overall deviance explained 18.6% - Skov *et al*. 2012.
[20] Figure 5-23 - Skov *et a*l. 2012.

altering their flight path. Similarly at Horns Rev 1, both distance to radar and distance to the windfarm were significant predictors for all bird tracks and common scoter tracks. In terms of the response curves, distance to windfarm the peak for all birds was between 2,000-3,000 m, whereas for common scoter it was around 1,000-2,000 m[21].

Assessment of methodology

From the results provided it is not possible to quantify an overall macro-avoidance rate although this study did provide information on the distances to which barrier effects were observed.

A1.3 Nysted offshore Windfarm

Location / habitat

Marine, offshore 10 km.

Turbine /array specification

Turbine array consists of 72 2.3 MW Bonus turbines covering 24 km^2. Distance between turbines – north to south (480 m) and east to west (850 m). Turbine specifications given as: hub height 69 m; rotor blade length 41 m; total height 110 m. Clearance above water is 28 m.

Case study number 1

Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M. & Fox, A.D. 2006. *Final results of bird studies at the offshore windfarms at Nysted and Horns Rev, Denmark*. Commissioned by DONG Energy and Vattenfall A/S. National Environmental Research Institute.

Desholm, M. & Kahlert, J. 2005. Avian collision risk at an offshore windfarm. *Biology Letters* 1: 296-298[22].

Methods

Peterson *et al.* (2006) focussed on barrier effects, displacement effects, physical changes to the habitat and collision risk. Work was carried out at Horns Rev and Nysted offshore windfarm but there were differences in methodology and timing of data collection. Study at Nysted covered the three phases of: baseline (1999-2002); during construction (2002-2003) and post-construction (2003-2005). Desholm and Kahlert (2005) reported the results from the barrier effects and collision risk work only.

[21] Figure 5-26 -- Skov *et al*. 2012.
[22] Assumed to be derived from the same data as Peterson *et al.* 2006.

Radar observations: Recordings by radar (Furuno FR125) were carried out from an observation tower, 5 km north-east of the windfarm area. The range was approximately 11 km and covered a circular area of 388 km^2. Migration was mapped by tracing the course of flocks onto a transparency and subsequently digitised. Only tracks longer than 5 km were included in the analyses.

The lateral response to the windfarms was investigated by setting a number of transects: the eastern gate (located along the full length the most eastern edge of the windfarm); the northern gate (located along the full length the most northern edge of the windfarm) and the buoy transect (running from north to south from the observation tower to a buoy, 6.9 km in length). During autumn migration, tracks of flocks of birds travelling in a westerly direction which crossed the buoy transect were selected to see if they crossed the eastern gate (in order to derive the percentage of birds which did so). In contrast, during spring migration the flight behaviour of birds was studied after they passed the windfarm and so is not considered further here. The total numbers of flocks of birds crossing the eastern and northern gate were also counted. In addition, migration intensities were compared for an area within the windfarm with an adjacent area outwith the windfarm (both less than 11 km^2 in area). Each area was subdivided into squares of 0.1 km^2 and within each cell, the lengths of radar tracks (bird flocks) were expressed as the total sum of track meters (the track density). In order to derive the change due to the windfarm, proportional differences in the bird densities within and inside the windfarm from the baseline data (pre-construction) were used to correct the data collected post-construction to derive avoidance rates.

In order to determine the response distance (where avoidance occurs) to the windfarm, transect lines to the east of the windfarm were set up which ran parallel to the direction of the rows of turbines (from north to south). These were spaced at intervals of 100, 200, 300, 400, 500 m and then at intervals of 500 m to 4,000 m and after which there were a further two transects at 5,000 and 6,000 m. The mean ± s.d. migration course of tracks were calculated for each transect (based on the gap between the transect itself and the 100 m interval to the west).

Visual observations*:* Abundance, phenology, diurnal pattern and flock sizes of species were recorded along the buoy transect. Count data was then converted into number of birds per 15 mins for all westerly bound birds in autumn and easterly bound birds in spring (although again the latter represents the number of birds after passing through the windfarm).

Study period

Radar observations: spring (easterly-orientated migration) and autumn (westerly-orientated migration) periods covered. Total number of hours or breakdown by season not reported.

Visual observations*:* During the main survey periods of 14 March to 19 April and 30 August to 12 November from 1999 - 2005, observations were carried out two days per week covering day and night time periods. A total of 259 h and 579 h observations gathered for the spring and autumn periods.

Species

Staging and migrating birds but common eider and geese spp most commonly recorded.

Conditions data collected under

Not specifically described but very little data of conditions under poor visibility (<1 km).

Results

The probability of birds crossing the windfarm was analysed using a logistic regression model and included the following explanatory terms and first order interactions (phase of development; distance to the observation tower when crossing the buoy transect), time of day, direction of winds (all of which were found to have significant effects). It was shown through comparison of data from the baseline and operation phases that 0.78 of all birds[23], which consisted mostly of common eider, avoided entering the windfarm post-construction during autumn migration. This was based on 40% of flocks entering the eastern edge of the windfarm during the baseline period compared to 9% during operation[24]. This was suggested to equate to 8 out of 10 flocks crossing the eastern gate during the baseline study then avoiding the windfarm during the post-construction phase. It was also shown that during the post-construction phase, the numbers of flocks crossing the eastern gate were higher at night than during the day (Desholm and Kahlert 2005 cited values of 13.5 % and 4.5 % respectively).

More specifically there was notable inter-annual variation in macro-avoidance rates for autumn migrating birds, again mostly common eider, ranging from 0.63 and 0.83[25] in the use of the windfarm post-construction compared to the baseline. These rates were derived from figures of 0.08-0.09 of flocks passing the eastern side of the windfarm compared to 0.24-0.48 passing the eastern gate of the windfarm during the pre-construction period[26].

There was a difference in migration intensity during the baseline period as the track densities in the eastern windfarm were 60% of the reference area which suggested a problem with detection rate. Nevertheless a significant reduction in track densities was reported for the post-construction period but there was acknowledgement that a reduction could be partially explained by problems of what is termed a shadow effect to do with individual turbines.

The standard deviation of the orientation was used to determine the lateral deflection as means of quantifying response distance to the windfarm (citing Kahlert *et al.*

[23] Figure 121. Calculated as 1-(0.09/0.40) - Peterson *et al.* 2006.
[24] Desholm and Kahlert (2005) reported the proportion of flocks entering the windfarm decreased from 40.4% during pre-construction to 8.9% during initial operation. Data collection methods were not extensively described - Peterson *et al.* 2006.
[25] Calculated as 1-0.08/0.48 and 1-0.09/0.24 - Peterson *et al.* 2006.
[26] Figure 122 - Peterson *et al.* 2006.

(2005) as justification for this approach). Analyses of data collected during the autumn migration, showed a significant interaction between the phase of development and distance to the windfarm (other terms were also significant but not discussed here due to lack of information presented which can be evaluated with respect to providing evidence for the response distance)[27]. Plots of the means of annual standard deviation values showed that there was little change in orientation for distances between 100 m and 5 km from the windfarm during the baseline period[28]. However, during the operation period, the orientation of tracks steadily changed over the distances 5 to 1 km away from the windfarm (orientation of birds at 3 km from the windfarm were significantly different to the baseline period) and the greatest deflection occurred between 500 m and 100 m (note that the way the transects were set up, there was a gap between 500 m and 1 km). A tendency was also reported for the first deflection to be recorded at greater distances during the day compared to the night time period (based on the multiple use of pair-wise t tests across each distance interval)[29].

Assessment of methodology

As there was a before and after comparison carried out at Nysted this was argued to provide greater confidence (compared to Horns Rev) that any changes were as a direct result of the windfarm presence.

The response distance was only possible for birds entering the windfarm during autumn (the area used during spring migration was beyond the edge of the radar range and hence the derived figures are based on autumn migration only. Moreover, tracks do not differentiate between individuals or flocks, therefore the reported macro-avoidance values do not respond to the level of individual birds.

[27] Table 41- Peterson *et al.* (2006).
[28] Figure 119 - Peterson *et al.* (2006).
[29] Table 42 - Peterson *et al.* (2006).

Case study number 2

Blew, J., Hoffman, M., Nehls, G. & Hennig, V. 2008. *Investigations of the bird collision risk and the responses of harbour porpoises in the offshore windfarms Horns Rev, North Sea, and Nysted, Baltic Sea, in Denmark. Part 1: Birds*. Report from the University of Hamburg and BioConsult SH, 145pp.

Methods

Methods used were exactly the same as used for Horns Rev (Appendix 1, section A1.2)

Study period

March to May to coincide with spring migration (44 ship days in 2005 and 2006) and September to November to cover autumn migration (51.5 ship days in 2005 and 2006).

Radar appeared to have been run continuously.

Species

Wide range of non-pelagic waterbirds with high numbers of common eider as well as raptors and songbirds. Transect counts showed that in spring, the common eider was by far the most common bird recorded and in autumn it was the great cormorant.

Conditions data collected under

Horizontal radar observations were limited to calm sea state conditions (wind speed < 2 ms^{-1}) and generally dry weather. Weather and sea state conditions tended to be better than those experienced at Horns Rev where fewer observation days were possible.

Visual observations were stopped when visibility <1 km.

Results

Radar tracks were categorised according to their direction in relation to the first row of the windfarm: flying towards (± 45° either side of perpendicular to the windfarm; flying away; and flying parallel (more or less). Initially tracks were presented regardless of their location (and therefore distance) in relation to the windfarm (but included tracks within the boundary of the outer row of the windfarm). During the day the overall number of tracks flying parallel to the windfarm was higher (n = 2,274) compared to towards (n = 1,725) or away (n = 563) from the windfarm. This pattern was not evident at night when the numbers flying towards (n = 968) and parallel (n = 804) were more similar but still much higher than flying away (n = 216).

In terms of determining whether horizontal avoidance occurred, the mean (and standard deviations) of angles of the approaching tracks were presented for the four

500 m width distance bands, for all anchor points east and west of Nysted offshore windfarm. It was reported that the angles did not increase (as would be predicted if horizontal avoidance occurred) or differ with decreasing distance to the windfarm (no statistical analyses were carried out).

Although the visual observations were designed primarily to look at the differences in flight height distribution, they were able to provide supporting evidence for macro avoidance occurring. For all gull spp significantly less birds were present inside the windfarm. No results for northern gannet were provided.

Assessment of methodology

See Appendix 1, section A1.2 for a discussion regarding the work carried out on radar and visual observations at Horns Rev where the same approach was used. With respect to looking for evidence of horizontal avoidance this study was unable to show evidence for a change in flight orientation. It was unclear though whether this was due to relatively wide bands being used (500 m in width) as other studies have used smaller intervals of 100 m at distances less than 1,000 m from the windfarm.

APPENDIX 2 Evidence review macro-response – displacement and attraction studies

A2.1 Egmond aan Zee

Leopold, M.F., Dijkman, E.M. & Teal, L. 2011. *Local Birds in and around the Offshore Windfarm Egmond aan Zee (OWEZ) (T-0 & T-1, 2002-2010)*. Texel, The Netherlands: Wageningen IMARES.

Leopold M.F., Camphuysen C.J., van Lieshout S.M.J., ter Braak C.J.F. & Dijkman E.M. 2004. *Baseline studies North Sea windfarms: Lot 5 marine birds in and around the future site Nearshore Windfarm (NSW)*. Alterra-rapport 1047.

Lindeboom, H.J., Kouwenhoven, H.J., Bergman, M.J.N., Bouma, S., Brasseur, S., Daan, R., Fijn, R.C., de Haan, D., Dirksen, S., van Hal, R., Hille Ris Lambers, R., ter Hofstede, R., Krijgsveld, K.L., Leopold, M. & Scheidat, M. 2011. Short-term ecological effects of an offshore windfarm in the Dutch coastal zone; a compilation. *Environmental Research Letters* **6**. doi:10.1088/1748-9326/6/3/035101.

Location/habitat

Marine 10-18 km offshore.

Turbine /array specification

Hub height 70 m and a rotor diameter 90 m (rotor altitude min 25 m, max rotor altitude 115 above mean sea level). Turbine array consists of 36 Vestas V90 3 MW turbines covering an area 27 km^2. Distance within turbines is 650 m within rows and 1000 m between rows.

Methods

The focus of Leopold *et al.* (2004) and (2011) was to look at avoidance and attraction by birds to the windfarm at Egmond aan Zee for what were termed local birds (although the survey work did cover the Princess Amalia windfarm site, results specific to this windfarm site were not presented). Survey periods covered the pre-construction and post-construction phases of the development. Lindeboom *et al.* (2011) reported the impacts of the windfarm on a range of taxonomic groups but with respect to birds presented less detail than the above reports and therefore is not considered further here.

The study area was approximately 725 km^2 (22 x 33 km). It was selected on the basis that it would include an adjacent offshore windfarm, Princess Amalia, and an anchorage area, where ships wait before entering the nearby major port. Ten transect lines were selected running east to west at distances of 2.47 km apart (with eight additional transect lines added in 2008 running north east to south west). The aim was to cover each transect twice (this was possible until the additional transect lines were added) and the transect lines were sailed in the same order each survey period. Successfully completed surveys ranged between 4-8 days in duration.

Ship based strip census surveys based on the methods adopted in the baseline studies in 2002-2004 (described in Leopold *et al.* 2004) which were originally derived from Tasker *et al.* (1984); Komdeur *et al.* (1992) and Camphuysen and Garthe (2004). All swimming birds were assigned to distance bands: AB (0-100 m); C (100-200 m) and; D (200-300 m) and all observations were assigned to five minute intervals. Flying birds were recorded using the snap shot methodology at intervals of 1 min.

Although BACI design was originally set to look at bird responses to the windfarm, there was considerable annual variation in seabird presence which hampered the ability to look for any differences between pre-construction and post-construction. Therefore the results focussed on comparisons within surveys (e.g. species-specific monthly counts). Presence/absence data were used as the response in Generalised Additive Mixed Models, which took into account temporal auto-correlation, for all individual species/month combinations there were sufficient data for. Otherwise a more simple General Additive Model was used or, in some cases, statistical models could not be run (birds were counted less than 10 occasions). Therefore, the number of surveys that were available for further analyses varied according to species and were a reflection of the relative abundance of birds each month. Presence /absence data were argued to be more appropriate as they were less affected by the large numbers of zero counts or the few counts with very large numbers of birds recorded. These models took into account the distance to coast, the northing value and the presence of impact area as factors (Egmond aan Zee, Princess Amalia and the anchorage area were considered individually within these models). The model output was then used to predict and subsequently map the probability of birds occurring across the survey area.

Within surveys, there was the possibility of four outcomes: attraction (probability of finding birds inside the windfarm was significantly higher than expected on the basis of the general distribution pattern); avoidance (probability of finding birds inside the windfarm was significantly lower than expected); indifference (probability of finding birds within the perimeter was not impacted by the windfarm and insufficient data.

Study period

Baseline/pre-construction surveys: T-0 = September and October 2002; April, May, June, August and November 2003; February 2004 (described in Leopold *et al.* 2004).

Post-construction surveys: T-1a = April, June, August, September, November (incomplete) 2007 and January 2008 (May was not repeated); T-1b = April, June, August, September (incomplete), November 2008; January, 2010; T-1c = April, June, August, October (September not possible) November 2009 and; January and February 2010.

Species

Local seabirds as defined as those which reside for some time in the study area. Species accounts were presented for: diver spp, great crested grebe, northern fulmar, northern gannet, great cormorant, common scoter, little gull, black-headed

gull, common gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake, Sandwich tern, common/arctic tern, common guillemot and razorbill).

Conditions data collected under

Generally aimed to survey in conditions with a Beaufort scale of less than 6 Bft but there were a number of transects that were carried out in higher winds of 6-7 Bft (when light conditions permitted).

Results

Northern gannet: Northern gannet tended to occur on all sides around Egmond aan Zee windfarm but rarely within the perimeter of the windfarm[30]. Observations recorded that those few birds that did enter only went one turbine deep. Where presence/absence analyses were possible for the post-construction period (n = 10 surveys), it was shown that the presence of the species was significantly negatively related to the Egmond aan Zee windfarm for only two surveys. Anecdotally it was reported that gannets never entered Princess Amalia Windfarm (which has a higher turbine density[31]). Also highlighted was the lack of searching feeding, resting in the windfarms during the surveys.

Lesser black-backed gull: It was evident that lesser black-backed gulls were often seen within perimeters of windfarm[32]. These birds tended to be either resting on the water or foundation structures or feeding at the tidal wakes around the monopiles. Presence/absence analyses for the post-construction period (n = 12 surveys), found that the presence of the species was negatively related to the Egmond aan Zee windfarm for only one survey (the rest were also negative but insignificant). This was counter to what would have been predicted as large fishing vessels only operated outside the windfarm which should have in effect reduced the numbers of birds inside the windfarm (resulting in an apparent avoidance). Most observations of lesser black-backed birds were anecdotally reported to be associated with, looking out for or resting in the wake of active fishing vessels.

Herring gull: Birds did occur in the windfarm area but overall fewer birds were recorded in the offshore environment compared to other gulls (notably in August where herring gulls remain mostly near shore). Like lesser black-backed gulls they were often associated with fishing vessels. Presence/absence analyses for the post-construction period (n = 14 surveys), found that the presence of the species was negatively related to Egmond aan Zee windfarm for eight surveys although this effect was only significant in three cases. Herring gull distribution patterns were thought to be likely to be attributable to overall latitudinal variation, as evidenced by the strong effect of distance to coast in the models (significant p values for six surveys).

[30] Figure 31 - Leopold *et al.* 2011.
[31] 60 2 MW turbines which are evenly spaced (550 m apart) in area of 14 km^2 - Leopold *et al.* 2011.
[32] Figure 34 - Leopold *et al.* 2011.

Great black backed gull: Birds were reported as occurring in the windfarm area[33]. Presence/absence analyses for the post-construction period (n = 18 surveys), found that the presence of the species was positively related to the Egmond aan Zee windfarm in five cases, four significantly, although this effect was only apparent at low densities. There were also two surveys in which significant effects were reported. As reported for lesser black-backed gull, birds did tend to feed around fishing vessels but not in the same high numbers.

Black-legged kittiwake: birds were recorded within the windfarm and in general numbers declined with decreasing distance to shore (apart from in November and one January). Presence/absence analyses for the post-construction period (n = 5 surveys), found that the presence of the species was positively related to the Egmond aan Zee windfarm in three cases, one significantly.

Assessment of methodology

Overall, there was lack of consistent evidence for either displacement or attraction for any of the species. This could have been partly due to the importance of factors operating at the larger scale of study area. For the larger gull species, there was a strong association with fishing vessels in the study area. Since fishing was no longer permitted in the windfarm areas, this could have confounded any results reported to do with possible attraction or avoidance of windfarms. There was also evidence that distance to coast was an important factor in determining the overall distribution patterns of herring gulls.

There were potential issues relating to the choice of statistical approach. As comparisons of pre-construction and post-construction data was deemed not to be possible, multiple tests for individual surveys were carried out which may have led to the possibility of a Type 2 error (increased chances of reporting a false significant result). Also the numbers of observations were low for northern gannet and gull spp and consequently the modelling power was very low (Lindeboom *et al.* 2011). Moreover, the model outputs were in the form of p values and model co-efficients which could not be converted into avoidance rates without further details being presented (even if consistent effects had been observed). Therefore, from the results provided, it is not possible to derive displacement/attraction rates or thus macro-response rates for the study species.

A2.2 Robin Rigg

Natural Power. 2014. Analysis of Marine Ecology Monitoring Plan Data from the Robin Rigg Offshore Windfarm, Scotland (Post-construction Year 3). Draft Technical Report. E.ON Climate & Renewables.

Location/habitat

Marine, offshore < 11 km

[33] Figure 38 - Leopold *et al.* 2011.

Turbine /array specification

Turbine array consists of 60 3.0 MW Vestas turbines which are positioned approximately 500 m apart. Turbine specifications are given as turbine towers 80 m high and a rotor blade length of 44 m.

Methods

The purpose of this report was to look at: displacement of key species; changes in patterns of abundance and distribution; compare observed patterns with predicted impacts/sensitivities from the EIA process.

Data collection was carried out during the pre-, during and post-construction periods.

Boat based surveys based on standard European Seabirds-At-Sea (ESAS) survey methods were carried our (e.g. prior to the publication of Camphuysen *et al.* 2004) as used in the baseline period. In order to ensure comparability between the different phases of the development, methods were kept the same throughout. Additional survey work has been carried out from year 3 of the post-construction period which corresponds to current best practice. The main difference between the two approaches is that for flying birds the former records flying birds using transect methodology whereas the latter uses the snap shot methodology currently regarded as best practice. A total of 10 parallel transects running in a south west to north east direction of 18 km in length and spaced 2 km apart.

For the purpose of analyses, each survey was divided into individual blocks of 600 m^2 (corresponding to the 300 m either side of the transect line as both sides of the boat are surveyed). In terms of the data, there was a cleaning process applied. Uneven sampling effort across the different phases of the development (some months were surveyed twice) was identified as an issue and therefore a single survey at random was selected. The study area was also cropped to remove an area in the northeast where shallow waters sometimes prevented access and two transects in the southeast were removed due to under surveying during the pre- and during construction phases. There was also a gap during the construction period where there was no building activity (January and July 2008) and these were also excluded from the analysis.

Birds on the water and birds in flight were analysed separately. Datasets that had fewer than 300 non-zero observations were not considered. Raw observations were mapped and summary statistics for the three development phases were calculated in order to provide an initial indication of any change. These included: mean number of sightings (groups of animals), mean number of individuals per segment and mean number of individuals per segment per month. These are not discussed here however and the results of models output are focussed upon.

Distance Sampling techniques were not used to correct the survey counts and a correction factor derived using the detection function was applied instead. Generalised Additive mixed effects mixture modelling carried out within a Bayesian framework were applied in order to deal with zero inflation (high number of zeros). Transect and survey were incorporated as random effects in order to deal with

spatial and temporal autocorrelation. Covariates used in the models were latitude, longitude month (or season) and time of day.

Outputs of the models were used to produce density surface maps of the predicted distribution during the three different phases of the development. Abundance and density estimates for each species within the windfarm and the study area were produced for each phase. In order to look at avoidance, model outputs were used to predict the number of animals within the windfarm and for buffers 0.5, 1, 1.5 and 2 km of the three different windfarm phases. Model outputs were presented only for the comparisons of pre-construction to construction and pre-construction to post-construction (but it was not clear which of the spatial scales they related to).

Study period

Baseline surveys: monthly basis between May 2001 and April 2002. Further pre-construction surveys April and May 2003 and then on a monthly basis between January 2004 and September 2004 (excluding April and June) with further work in July 2007. Construction surveys: monthly basis between January 2008 and February 2010 (excluding November 2009). Post-construction: monthly surveys from March 2010 to February 2013 – scheduled to continue until 2015.

Species

Data were collected for a wide range of species (e.g. seabirds, seaducks, waders, passerines). Species accounts were only presented for the following key species: scaup, common scoter, red-throated diver, Manx shearwater, northern gannet, great cormorant, black-legged kittiwake, herring gull, great black-backed gull, common guillemot, and razorbill.

Conditions data collected under

Not specified but ESAS provide guidance regarding suitability of conditions.

Results

Northern gannet: Modelling of the numbers of northern gannet on the water was not possible as there were too few sightings. The predicted numbers of northern gannet in flight across the three different phases of the development were found not to be significantly different. There appeared to have also been relatively little change in the predicted densities for the windfarm site, windfarm plus buffers (at any of the scales) or even at the level of the study area[34]. Although northern gannet was recorded throughout the study area, densities of the gannets were reported as being generally low[35].

Black-legged kittiwake: The predicted numbers of black-legged kittiwake on the water across the three different phases of the development were found not to be

[34] Figure 3.55 – Natural Power 2014.
[35] Figures 3.56-3.61- Natural Power 2014.

significantly different. There appeared to have also been relatively little change in the predicted densities for the windfarm site, windfarm plus buffers (at any of the scales) or even at the level of the study area[36]. A similar result was found for black-legged kittiwakes in flight[37].

Herring gull: Modelling of the numbers of herring gull on the water was not possible as there were too few sightings. The predicted number of herring gull in flight across the three different phases of the development were found to be significantly different with the numbers within the windfarm decreasing over the development (pre-construction to construction p = 0.0021, parameter estimate -0.750 and pre-construction to post-construction p = 0.0013, parameter estimate - 0.841).

Great black-backed gull: Modelling of the numbers of herring gull on the water was not possible as there were too few sightings. The predicted number of herring gull in flight were found to significantly differ from pre-construction to construction (p = 0.0166, parameter estimate -1.133) but not from pre-construction to post construction (p = 0.7854).

Assessment of methodology

There were insufficient data to allow modelling of the observations of birds on the water for northern gannet, herring gull, and great black-backed gull. For birds in flight, there was evidence for a significant decrease for herring gull both during the construction and post-construction periods whereas this decrease was only noted during construction for great black-backed gull. Northern gannet and black-legged kittiwake did not appear to respond to the presence of the windfarm. From the results provided, it was not possible to derive macro-response rates since it was not clear what models have been fitted and it was not apparent whether the changes were due to the presence of the windfarm or as result of changes at the scale of the overall study site. It is acknowledged though that despite this being year 3 of the post construction, it is not the final report and any reported results should be considered as preliminary findings.

A2.3 Blighbank

Vanermen, N., Stienen, E.W.M., Courtens, W., Onkelinx, T., Van de walle, M. & Verstraete, H. 2013. *Bird monitoring at offshore windfarms in the Belgian part of the North Sea - Assessing seabird displacement effects*. Rapporten van het Instituut voor Natuur- en Bosonderzoek 2013 (INBO.R.2013.755887). Instituut voor Natuur- en Bosonderzoek, Brussel.

Location/habitat

Marine, 42 km offshore

Turbine /array specification

[36] Figure 3.82 - Natural Power 2014.
[37] Figure 3.83 - Natural Power 2014.

55 turbines. Additional information was not presented.

Methods

This report looked at Blighbank and Thorntonbank windfarms (but also referred to the more recent development of Lodewijckbak) in what is termed the windfarm concession zone located in the north eastern edge of the Belgian Part of the North Sea (BPNS). Surveys at both windfarms are still on going.

Data collection was carried out during the pre-construction, during and post-construction periods.

A BACI approach was adopted in order to monitor sea bird displacement. A control area of comparable size was selected on the basis of having similar attributes in terms of number of birds, environmental conditions and having sufficient historic data. A buffer zone of 3 km was applied to the boundary of the windfarm (and the control area), in order to reflect the distance to which the effects of the windfarm could be an issue for birds.

Boat transects were carried out on a monthly basis (citing Tasker *et al.* 1984) from 2008. The time interval used in this survey for recording was 10 minutes (a number of other windfarm surveys use 1 min). Although only transect routes used post 2012 were shown[38], despite some apparent minor shifts in the location the overall configuration was considered to be the same over the whole monitoring period (Nicolas Vanermen *pers. comm.*). An overview was provided of all the ESAS counts carried out by INBO during the period of 1992-2012 based on location of counts, this could not be used to look at survey effort which varied over the study period[39]. Count effort for Blighbank[40] (as shown by the number of surveys) indicated overall higher effort in the pre-construction period (but this included data possibly dating back to 1992). There was also marked monthly variation in effort in the preconstruction phase with peaks in February/August for the pre-construction period and in March/December for the post construction period.

Although distance sampling was used to correct count data to estimate the total numbers of birds within the BPNS (based on Buckland *et al*. 2001), it was not applied for modelling of the windfarm data (this was on the grounds that the correction factor used for both control and the windfarm area was likely to be the same Nicolas Vanermen *pers. comm.*). In order to analyse the count data, generalised linear models were used, with a negative binomial distribution assumed in order to cope with over dispersion. Modelling was carried out using area (the reference area or the impact area) and month (as a as a continuous variable in order to model seasonality) included as explanatory terms in what was termed the reference model (based on data collected prior to April 2008). The best model was then selected using a backward approach using a Wald test and looking at the resulting AIC values. The impact model was a simple extension of the count

[38] Figure 27 - Vanermen *et al*. 2013.
[39] Figure 2 - Vanermen *et al.* 2013.
[40] Figure 29 - Vanermen *et al*. 2013.

component of the reference model with before and after being added as factor variables to the model. Although not carried out in this report, the natural exponent of the model coefficients can be used to derive the factorial change (and hence the overall percentage change in numbers from pre to post construction – see Table A6.1).

Species' preference for the windfarm area was calculated using Jacob's Selectivity Index (calculated using the proportion of birds that occur inside the entire windfarm concession zone compared to the total numbers within the BPNS and the proportion of the surface area of the concession zone to the total area of the BPNS) whereby values of -1 represent total avoidance and + 1 is total preference (attraction). However this data was only carried out for the baseline data and hence are not considered further here.

The impact of the windfarm was considered separately for the post-construction phase at the scale of the windfarm, the windfarm and buffer, and the buffer without the windfarm[41]. Displacement-related coefficients and their respective p values were reported.

Study period

The baseline period (reference period) referred to data pre-September 2009. The construction period ran from September 2009 to August 2010, and the post-construction period was from September 2010 onwards. Data collected during the initial construction period were not used in subsequent assessment due to access issues over this period. Results are presented for up until December 2012.

Species

Northern fulmar, northern gannet, great skua, little gull, common gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake, common guillemot, and razorbill.

Conditions data collected under

Not specified in the report. Conditions were, however, mostly favourable - boat surveys are cancelled when wave heights > 1.8 m, and in poor visibility (Nicolas Vanermen *pers. comm.*).

Results

Northern gannet: Model coefficients were significant for the scale of the windfarm and buffer and buffer without the windfarm (see Table A6.1). Therefore there were highly significant decreases in numbers of northern gannet in the windfarm and the buffer of 3 km at all three spatial scales considered.

[41] Table 18 - Vanermen *et al.* 2013.

Lesser black-backed gull: Model coefficients were significant for the windfarm and buffer, and buffer without the windfarm, and were only just not significant for just the windfarm. Therefore there was a significant increase in numbers of lesser black-backed gull in the windfarm and the buffer of 3 km relative to the pre-construction period.

Herring gull: The model coefficient was only significant at the scale of the windfarm, indicating an increase in numbers in the windfarm area relative to the pre-construction period.

Great black-backed gull: The model coefficients were not significant, indicating no changes in numbers of the species relative to the pre-construction period.

Black-legged kittiwake: The model coefficients were not significant, indicating no changes in numbers of the species relative to the pre-construction period.

Table A2.1 Model outputs of Negative binomial modelling converted into factorial changes

Species	Scale	Model coefficient	P value	Factorial Change*	Overall change as a proportion
Northern gannet	Windfarm	-1.83	0.000	0.16	0.84
	Windfarm plus buffer	-1.52	0.000	0.22	0.78
	Buffer	-1.32	0.003	0.27	0.73
Lesser black-backed gull	Windfarm	1.57	0.059	4.81	-3.81
	Windfarm plus buffer	2.39	0.004	10.91	-9.91
	Buffer	2.37	0.006	10.70	-9.70
Herring gull	Windfarm	3.97	0.000	52.98	-51.98
	Windfarm plus buffer	1.26	0.111	-	-
	Buffer	0.83	0.269	-	-
Great black-backed gull	Windfarm	1.08	0.127	-	-
	Windfarm plus buffer	0.47	0.447	-	-
	Buffer	0.54	0.428	-	-
Black-legged kittiwake	Windfarm	0.25	0.605	-	-
	Windfarm plus buffer	0.50	0.264	-	-
	Buffer	0.77	0.092	-	-

*natural exponent of the model co-efficient.

Assessment of methodology

The results of this report should be considered as being preliminary since further data was collected for 2013. Nevertheless, northern gannet was shown to decrease in response to the presence of windfarm by a value of 0.84. This value could be taken as being indicative of macro-avoidance. Whereas both lesser black-backed gull and herring gull shown quite marked attraction to the windfarm. Great black-backed gull and black-legged kittiwake showed no overall response to the windfarm. From the results provided it was not possible to look at seasonal variation in displacement or attraction.

Sampling effort was biased towards the pre-construction phase and was characterised by variable effort on a monthly basis. Spatial coverage over the whole study period is likely to have been fairly consistent however. The data presented in this report is based on a BACI approach and potentially has limited value in looking at changes in the wider area but long term monitoring in the BPNS has continued throughout the study period and hence there is scope to include this at a later stage if required.

A2.4 Thorntonbank

Vanermen, N., Stienen, E.W.M., Courtens, W., Onkelinx, T., Van de walle, M. & Verstraete, H. 2013. *Bird monitoring at offshore windfarms in the Belgian part of the North Sea - Assessing seabird displacement effects*. Rapporten van het Instituut voor Natuur - en Bosonderzoek 2013 (INBO.R.2013.755887). Instituut voor Natuur- en Bosonderzoek, Brussel.

Location/habitat

Marine, 27 km offshore.

Turbine /array specification

Initially six turbines, final array to consist of 54 turbines.

Methods

See Appendix 2, section A2.3 for overall approach.

The impact of the windfarm was considered separately for the two different operation phases: phase 1 (turbine array consisting of six turbines) and; phase 2 (second construction period). Models were run at the scale of the windfarm and buffer only[42].

Power analyses were also carried out for the reference data collected in the Thorntonbank study area in order to determine the power required to detect change in numbers of birds (25, 50 and 75% decrease) and the length of the monitoring period required.

Study period

Monthly surveys were started in 2005 (although additional data were available from 1993 based on surveys that have been carried out of the whole region of the BPNS but coverage was uneven spatially and temporally). The baseline period (reference period) referred to data pre-April 2008. The construction period ran from April 2008 to May 2009, and the post-construction period (called here the impact period) was from June 2009 to April 2011. Thereafter there was another period of construction from May 2011 that was ongoing at the time of the report.

Species

Northern fulmar, northern gannet, great skua, little gull, common gull, lesser black-backed gull, herring gull, great black-backed gull, black-legged kittiwake, Sandwich tern, common tern, common guillemot, razorbill.

Conditions data collected under

Not specified but ESAS provide guidance regarding suitability of conditions.

[42] Table 15 - Vanermen *et al.* 2013.

Results

Northern gannet: For both phase 1 and phase 2, the model coefficients were not significant, indicating no changes in numbers of the species relative to the pre-construction period.

Lesser black-backed gull: For phase 1, the model co-efficient was not significant. For phase 2, a significant model co-efficient of 2.13 was reported ($p = 0.052$) for the scale of the windfarm, indicating a decrease inside the windfarm (but this effect was not found for the other models at the scales of the windfarm plus buffer, and buffer without the windfarm).

Herring gull: For both phase 1 and phase 2, the model coefficients were not significant, indicating no changes in numbers of the species relative to the pre-construction period.

Great black-backed gull: For phase 1, the model co-efficient was reported as 1.5 and was found to be significant ($p = 0.024$) for the windfarm plus buffer indicating an attraction to the windfarm. Whereas for phase 2, the model coefficients were not significant, indicating no change in numbers of the species relative to the pre-construction period.

Black-legged kittiwake: For both phase 1 and phase 2, the model coefficients were not significant, indicating no changes in numbers of the species relative to the pre-construction period.

Assessment of methodology

The results of this study were derived from when the windfarm only consisted of 6 turbines (phase 1) or during the next phase of construction of a further 48 turbines (phase 2). Hence the years covered by this report do not include the post-construction phase of a fully post-construction windfarm. Hence the results are not considered further here as part of this review.

A2.5 Nysted

Petersen, I.K., Christensen, T.K., Kahlert, J., Desholm, M. & Fox, A.D. 2006. *Final results of bird studies at the offshore windfarms at Nysted and Horns Rev, Denmark*. Commissioned by DONG Energy and Vattenfall A/S. National Environmental Research Institute.

Location/habitat

See under section 5.4.1.3 under barrier effects.

Turbine /array specification

See under section 5.4.1.3 under barrier effects.

Methods

Aerial transect surveys were carried out using methodology described in Kahlert *et al.* 2004 (which prior to the publication of Camphuysen *et al.* 2004 was commonly cited by other studies as the standard methodology). A total of 26 parallel transects running north to south separated by distances of 2 km were carried out covering an area of 1,700 km^2. The area was extended by four additional transect lines in 2002 to increase the area to 1,846 km^2.

Jacob's selectivity indexes (D) were used in order to look at displacement and attraction. This approach essentially determines bird preferences for the windfarm area and a buffer zone (2 and 4 km) where birds could still be impacted, in relation to their preference to the whole study area. Values fell between -1 (displacement) and +1 (attraction). Bird encounters (for both individuals and groups here termed as clusters) rather than estimates of bird densities were used. Bird preferences were then compared by looking at the pre- and post-construction D values, based on a simple comparison of number rather by formal statistical analyses, in order to describe the change in bird utilisation of the windfarm.

Bird encounter rate (number of birds reported per km of survey route per observer) was used as a proxy of density in order to calculate mean densities in the windfarm area and in the buffer zone. Comparisons of the mean densities pre- and post-construction were carried out using Student's t-test with corrections for unequal variance. Sufficient data (with respect to the five priority species) was available for comparisons for herring gull at Nysted in January and Horns Rev in March.

Study period

Pre-construction period = August 1999 to August 2002 (n = 21 surveys); construction period = January 2003 to August 2003 (n = 3); post-construction period = January 2003 (sic) to November 2005 (n= 8). The timing of the actual surveys (e.g. by month were not reported). Only the pre-construction and post-construction surveys were used. There was a lack of autumn surveys for the post-construction phase and therefore only winter and spring surveys were available.

Species

Diver spp, great cormorant, long-tailed duck, common eider, common scoter, red-breasted merganser, herring gull and great black-backed gull.

Conditions data collected under

Not specified.

Results

Herring gull: Comparisons of pre- and post-construction selectivity indices for numbers clusters of birds showed no change[43]. Whereas selectivity indices for

[43] Table 27 and 29 - Peterson *et al*. 2006.

numbers individuals showed a tendency towards decreased selectivity (e.g. less birds were using the area) for the windfarm as well as both buffer zones[44]. There was no significant difference between bird encounter rate between the pre- and post-construction phases in the windfarm area or the 4 km zone but a significant difference was found for the 2 km buffer. The report concluded there was no evidence for either attraction or avoidance.

Great black-backed gull: outputs of the models were all found to be insignificant apart for the selectivity indices for individual birds post-construction and hence are not reported further here as they have no meaningful comparison for pre-construction.

Assessment of methodology

Overall there was little evidence that herring gull showed any response to the presence of the windfarm.

There are a number of potential limitations of the approach used. There may be issues to do temporal coverage – from the information provided, it was difficult to be able to evaluate how sampling effort varied over the different phases of the development. Also whilst the Jacob's selectivity indices may provide an indication of the likely direction of response, these cannot be directly translated into displacement rates. Also the comparison of pre- and post-construction bird encounter rate had limited value since they provided no indication of changes in distribution that may have occurred at a wider scale (and therefore nothing to do with the presence of the windfarm).

[44] Table 28 and 30 - Peterson *et al*. 2006.

A2.6 Horns Rev

Location/habitat

See under Appendix 1, section A1.2.

Turbine /array specification

See under section Appendix 1, section A1.2.

Methods

Aerial surveys: Aerial transect surveys were carried out using methodology described in Kahlert *et al.* (2004) which prior to the publication of Camphuysen *et al.* (2004) was commonly cited by other studies as the standard methodology. A total of 26 parallel transects separated by distances of 2 km were carried out covering an area of 1,350 km^2.

Study period

Pre-construction period = August 1999 to January 2002 (n = 16 surveys); construction period = March 2002 to August 2002 (n = 3); post-construction period January 2003 to November 2005 (n= 15). The timing of the actual surveys (e.g. by month were not reported). Only the pre-construction and post-construction surveys were used.

Species

Diver spp, northern gannet, common eider, common scoter, little gull, Arctic/common tern and guillemot.

Conditions data collected under

Not specified.

Results

Northern gannet: There were no observations of northern gannet inside the windfarm pre- or post-construction. Comparisons of pre- and post-construction selectivity indices for the buffer zones indicated increased avoidance at the 2 and 4 km zone. Insufficient numbers of birds were recorded in order to be able look at encounter rates and limited further interpretation of what the likely overall response of northern gannet to the windfarm.

Herring gull: Comparisons of pre- and post-construction selectivity indices for clusters and individuals of birds indicated a reduced avoidance of the windfarm area. The bird encounter rate revealed no significant difference between the pre- and post-construction period. It was concluded that despite an increased preference being found during construction (citing Christensen *et al.* 2003), attraction was not observed post-construction.

Black-legged kittiwake: Model outputs were not significant for numbers of clusters of birds post-construction and for both pre- and post-construction for numbers of individual birds. Hence the results are not reported here.

Assessment of methodology

See Appendix 2, section A2.5.

A2.7 Alpha ventus demonstration site

Bundesamt fur Seeschiffart und Hydrographie, BSH 2011. Okologische Begleitforshung bei alha ventus erste Ergebnisse (Environmental research at alpha ventus – first results). Contributions from the Event of 10 May 2010, Katholische Akademie Hamburg.

Mendel, B., Kotzerka, J., Sommerfeld, J., Schwemmer, H., Sonntag, N. & Garthe, S. 2014. Effects of the Alpha Ventus offshore test site on distribution patterns, behaviour and flight heights of seabirds. *In* Ecological Research at the Offshore Windfarm Alpha Ventus: Challenges, Results and Perspectives. Editors Federal Maritime and Hydrographic Agency, Federal Ministry for the Environment, Nature Conservation and Nuclear Safety. Springer Spektrum.

All the post-consent monitoring reports from this OWF demonstration site are written in German (Stefan Garthe *pers. comm.*). The first reference reviewed is a report (BSH 2014) which has a full English translation. The second reference (Mendel *et al.* 2014) is a book chapter and is written in English. Neither reference can be considered to be fully comprehensive in the level of detail provided but given the importance of this OWF site this information should be included. The information which is cited below is largely taken from Mendel *et al.* (2014).

Location/habitat

45 km offshore

Turbine /array specification

Twelve turbines. Two designs (jacket foundation and tripod steel foundations) – no further information provided.

Methods

Two study areas were selected: the key study area, the size of which was in excess of 30 times the size of the windfarm itself and; a reference site which appeared to be nearly twice the size of the study area. Boat based surveys were carried out according to standard European Seabirds-At-Sea (ESAS) survey methods. Aerial-based methods were based on methods described in Pihl and Frikke (2002), Noer *et al.* (2000) and Diederichs *et al.* (2002) (full citations are given in Mendel *et al.* 2014). As well as data from the EIA studies, additional data from eight multiple-day ship-based surveys and 21 aerial surveys carried out in both study areas were

available. No further information was provided, however (e.g. on the timing of the surveys in relation to season).

In order to carry out analyses of the changes in distribution patterns for pre-and post-construction data, data were collated into grid cells of 1 km^2 and only data from the key study area were used. A total of six species or species groups were looked at (divers, northern gannet, lesser-black backed gull, little gull, black-legged kittiwake and common guillemot) and only the most important period/s for each of these were focussed upon. Data were also collated over large time periods (usually seasons).

Changes in abundance were looked at using the pre- and post-construction data and only two species were considered (lesser-black backed gull and common guillemot). Generalised Linear Mixed Models of the abundances of birds at different distances in relation to the windfarm (0-2 km, 2-6 km and 6-10 km) were tested in three different models using a Poisson error distribution.

The percentage of birds recorded in each behavioural category was calculated for the key study areas and the reference area for lesser black-backed gull only.

Study period

Data from 2000-2008 were regarded as pre-construction (construction started in September 2008) and data from 2010-2012 represented the post-construction period.

Species

Northern gannet, northern fulmar, black scoter, skua spp, gull spp, and auks spp. Key species: Red-throated diver, black-throated diver, lesser black-backed gull, black-legged kittiwake, little gull, common guillemot and razorbill.

Conditions data collected under

Data collected according to ESAS methods (sea state < 5Bft).

Results

Changes in distribution

The statistical significance of the following results was not provided and interpretation of results was largely based on maps representing densities of birds for the 1 km^2 grid cell system of the key study area. Overall lower abundances were reported post-construction for six of the species/groups but only the relevant species are reported further here.

Northern gannet: the impact of the windfarm was hard to qualify due to the very low numbers recorded within the key study area. This species was reported to have occurred on seven occasions (nine individuals) within the windfarm area during the pre-construction period and none were observed post-construction. Data were taken from March to September and hence represented the breeding season.

Lesser-black-backed gull: a ‘clear decrease’ was reported to have occurred from the pre- to the post-construction period. Although low to medium densities were reported post construction within the windfarm area, the highest densities were found a few kilometres away from the windfarm site (previously some of the highest were found within the perimeter of the windfarm area during pre-construction). Data were taken from May to July and hence represented the breeding season.

Black-legged kittiwake: a ‘remarkable decline’ occurred post-construction not only within the perimeter of the windfarm but at the scale of the whole key study sites. Numbers recorded overall were very low however (e.g. highest number of birds recorded per km^2 was 5). Data were taken from November to April and hence represented the non-breeding season.

Changes in abundance

Lesser-black backed gull: Statistically significantly lower abundances were reported for the 0-2 km, 2-6 km and 6-10 km distance class and the models suggested that the disturbance effect was strongest within 2 km of the windfarm,

Assessment of methodology

Based on the information provided, it is not possible to carry out a proper assessment of the methodology used. The overall abundance of northern gannet was very low and therefore this study cannot be cited as evidence of the windfarm having an impact on their distribution. There is some evidence to suggest that displacement may be occurring for lesser black-backed gull and black-legged kittiwake based on the maps of the distribution of bird densities for pre- and post-construction, but there was a lack of statistical analyses. However a statistically significant reduction in the abundance of lesser black-backed gulls was reported for all the three distances classes from the windfarm.

APPENDIX 3 EVIDENCE REVIEW HORIZONTAL MESO-RESPONSE

A3.1 De Put, Nieuwkapelle

Everaert, J. 2008. *Effecten van windturbines op de fauna in Vlaanderen Onderzoeks resultaten, discussie en aanbevelingen*. INBO, Brussels

Methods

Baseline data describing bird movements within the area, prior to turbine construction, were collected on six days between December 2004 and February 2005 at periods of dawn and dusk. Following turbine construction, additional data were collected on six days between December 2005 and March 2006, again at dawn and dusk. Changes in the number of birds flying within 100 m and 300 m of each turbine pre- and post-construction were then modelled using a factorial ANOVA.

Seasons / time of day

Data were collected over the winter at dawn and dusk.

Species

Black-headed and common gulls.

Conditions data collected under

Not specified.

Location / habitat

Terrestrial site in Belgium.

Turbine / array specification

A two turbine array. Each turbine has a mast height of 75 m and a rotor diameter of 48 m.

Results

No significant differences were recorded in the number of black-headed or common gulls passing within 300 m or 100 m of the turbines between the pre- and post-construction periods.

Assessment of methodology

A key flaw in this study is the lack of a control site with which to compare differences in movement pre- and post-construction. A consequence of this is that it is not possible to determine whether the lack of significant changes reflects the local population remaining relatively stable or whether the overall proportion, but not numbers, of a variable local population passing the turbines has changed.

A3.2 Egmond aan Zee

Krijgsveld, K.L., Fijn, R.C., Japink, M., van Horssen, P.W., Heunks, C., Collier, M.P., Poot, M.J.M., Beuker, D. & Dirksen, S. 2011. *Effect studies Offshore Wind farm Egmond aan Zee: Final report on fluxes, flight altitudes and behaviour of flying birds*. Bureau Waardenburg

Methods

Radar Observations

Between July 2009 and March 2010, the flight paths of birds within the windfarm were recorded using a horizontal radar with range of 0.75 nautical miles. The study area included six turbines and it was possible to collect data on 235 out of the 239 days during the study period, although it was necessary to filter out data on an additional 59 days due to the incidence of 'clutter'. Data were then analysed using a t-test to assess whether birds were distributed evenly within the windfarm by comparing the number of birds passing within 50 m of a turbine to the number of birds elsewhere.

Seasons / time of day

Data were collected during daylight on eight occasions between July and December.

Species

Not stated

Conditions data collected under

All conditions.

Location / habitat

Marine 10 km offshore.

Turbine / array specification

Egmond aan Zee Offshore Windfarm covers an area of 27 km^2 and contains 36 turbines. Each turbine has a hub height of 70 m and rotor diameters of 90 m. Turbines are arranged in four rows, with 650 m between turbines in each row and 1 km between rows. The study of horizontal meso-responses covered six turbines at the edge of the windfarm.

Results

There was a statistically significant difference in the numbers of birds flying within 50 m of the turbines in comparison to the proportion of birds elsewhere in the study area. Over the course of the study period, this reflected a horizontal meso-response

rate of 0.34 (i.e. the number of birds within 50 m of a turbine was 66% of that elsewhere within the windfarm).

Assessment of methodology

Data used in this study have been collected using radar, meaning near-continuous data collection was possible. In order to detect finer scale movements of birds in relation to the windfarm, the resolution of the radar was reduced to cover a distance of 0.75 nautical miles. As a consequence, it was possible to detect movements of birds that were as close as 1 m to turbines. However, a key limitation of the data is that it is not possible to relate echoes to individual species, or to determine whether a single echo reflects an individual birds, or a flock. An additional limitation is that birds at low altitudes may have been obscured by high waves, which they exploit in order to minimise energy expenditure.

A3.3 Horns Rev I and II

Skov, H., Leonhard, S.B., Heinanen, S., Zydelis, R., Jensen, N.E., Durinck, J., Johansen, T.W., Jensen, B.P., Hansen, B.L., Piper, W., Grøn, P.N. 2012. *Horns Rev 2 Monitoring 2010-2012. Migrating Birds.* Orbicon, DHI, Marine Observers and Biola. Report commissioned by DONG Energy

Methods

Radar Monitoring

Between September 2010 and May 2012 Bird movements were recorded using horizontal radar at stations within the Horns Rev I and Horns Rev II offshore windfarms. All movements within 6 km of the radar were recorded. Two observers were used during the data collection. The first observer followed the tracks and recorded information within a database. The second observer attempted to locate each of the tracked objects in the field using binoculars or a telescope and relayed information on the species identification, number and altitude to the first observer.

Seasons / time of day

Data were collected during the spring and autumn migration periods during the hours of daylight.

Species

Northern gannet (442 birds), common scoter (2,374 birds), large gulls (408 birds), terns (617 birds).

Conditions data collected under

Data were generally collected during relatively calm conditions (little wind or rain and good visibility).

Location / habitat

Horns Rev I is located 17.9 km from the Danish coast and Horns Rev II is located 31.7 km from the Danish coast.

Turbine / array specification

Horns Rev I is an array of 80 turbines, each with a hub height of 70 m and a rotor diameter of 80 m. Horns Rev II is an array of 91 turbines, each with a hub height of 68 m and a rotor diameter of 93 m.

Results

The study estimated the mean, minimum and maximum distances from turbines recorded by each species. On average, northern gannets were recorded passing within 1,119 m of turbines (range 0-2,840 m), common scoter were recorded passing within 921 m of turbines (range 0-4,302 m), large gulls were recorded passing within 783 m of turbines (range 50-2,252 m) and terns were recorded passing within 840 m of turbines (range 0-2,355 m). In practice, without knowing the shapes of these distributions, it is hard to use this information to estimate the magnitude or direction of horizontal meso-responses to the turbines. In practice, the mean distance to turbines is likely to be strongly influenced by the body size of the species concerned, or by their tendency towards flocking behaviour, both of which are likely to increase their detection at greater distances. However, of the 408 large gulls tracked, none passed within 50 m of the turbines, suggesting a strong, negative meso-response to the turbines occurring at a distance of at least 50 m.

Assessment of methodology

The way data are presented make it difficult to disentangle meso-responses to the turbines. In particular, biases may exist relating to the detectability of different species, which may make the estimates of mean distance to turbines unreliable. Of the information presented, the minimum distance to turbines for large gulls is of value in estimating a meso-response rate.

A3.4 Hungary

Janoska, F. 2012. Investigations of Bird Collisions in 2 Wind farms. *International Scientific Conference on Sustainable Development & Ecological Footprint*, Sopron, Hungary, March 26-27 2012

Methods

Between November 2010 and November 2011, two Hungarian windfarms were visited every two weeks. During visits, the altitude and flight direction of birds were noted.

Seasons / time of day

Data were collected throughout the year.

Species

Yellow-legged gull

Conditions data collected under

No Details given.

Location / habitat

Two terrestrial sites in Hungary.

Turbine / array specification

No details given.

Results

Of the yellow-legged gulls recorded, only 2.5% (23/917) were recorded flying within 75 m of turbines, reflecting a meso-response of 0.975, and only 0.6% (6/917) were recorded flying within 25 m of turbines, reflecting a meso-response of 0.994.

Assessment of methodology

Very little detail is given describing the methodology used. As a consequence, these data must be interpreted with extreme caution. In particular, it is unclear to what extent data reflect avoidance, and to what extent they more generally reflect the flight paths taken by birds passing through the area.

APPENDIX 4 EVIDENCE REVIEW VERTICAL MESO-RESPONSE

A4.1 Barrow Offshore Windfarm

Barrow Offshore Wind Limited. *Post Construction Ornithological Monitoring – Third Year Report and Overall Conclusions*.

Methods

Boat-based estimation of flight heights.

Following the construction of Barrow Offshore windfarm, boat-based surveys were carried out during the breeding season and autumn migration in 2006, 2008, 2009 and 2010. In total 12 surveys, each lasting a single day were carried out, of which 8 were during the breeding season (May to August) and 4 during autumn migration (September to November). Boat survey data were collected within the windfarm according to standard protocols (Camphuysen *et al.* 2004) and flying birds were assigned to height bands of <5 m, 5- 15 m, 15-100 m and >100 m. Birds at risk of collision were assumed to be all those flying >15 m. The proportion of birds observed flying at heights presenting a risk of collision were then summarised across all surveys. Pre-construction proportions at collision risk height within the windfarm were compared to post-construction proportions at collision risk height, although no detailed analyses were undertaken.

Seasons / time of day

Data were collected during the breeding season and autumn migration periods.

Species

Auk spp. (238 recorded in 2010), common guillemot (2,002 recorded in 2010), razorbill (691 recorded in 2010), great cormorant (5 recorded in 2010), red-throated diver (2 recorded in 2010), black-headed gull (6 recorded in 2010), common gull (5 recorded in 2010), great black-backed gull (23 recorded in 2010), herring gull (142 recorded in 2010), black-legged kittiwake (132 recorded in 2010), lesser black-backed gull (425 recorded in 2010), gull spp. (51 recorded in 2010), Arctic skua (2 recorded in 2010), northern gannet (53 recorded in 2010), Manx shearwater (12 recorded in 2010), Sandwich tern (30 recorded in 2010), common scoter (10 recorded in 2010),

Conditions data collected under

No details given.

Location / habitat

Marine 7 km Offshore

Turbine / array specification

An array of 30 turbines covering an area of 10 km^2 and arranged in four rows of seven or eight turbines each. The rows are separated by a distance of 750 m and within the rows, each turbine is separated by a distance of 500 m. Each turbine has a hub height of 75 m above sea-level and a rotor diameter of 90 m.

Results

Several species were not present in sufficient numbers to allow a reliable estimate of the changing proportion of birds flying at a height placing them at risk of collision. Of those that were, common guillemot, great black-backed gull, herring gull, lesser black-backed gull and Sandwich tern all showed a decline in the proportion of birds flying at risk height, with meso-responses of 1, 0.29, 0.65, 0.28 and 0.55 respectively. However, other species (or groups) showed an increase in the proportion of birds flying at risk height including black-legged kittiwake, unidentified gulls and northern gannet, with meso-responses of -0.41, -0.85 and -0.59 respectively, reflecting an apparent attraction to the rotor-swept area of the turbines.

Assessment of methodology

Boat-based data collection was robust, following standard methodologies (Camphuysen *et al.* 2004). However, in assessing the vertical response to turbines there is a key flaw in the available data. In order to compare flight height data to that collected pre-construction, the same flight height bands were used in both study periods, and it was assumed that all birds flying at a height of more than 15 m above sea-level were potentially at risk of collision. However, as the rotor-swept area covers an area from 30 m to 120 m above sea-level, this may lead to a significant over-estimate of the actual number of birds flying at collision risk height. As a result, the meso-response rates of birds within the windfarm may be underestimated. An additional, arguably less serious, flaw in the data collection is that estimates of the birds at collision risk height refer to flocks, rather than individuals. Flock size is likely to show significant variation, making it difficult to infer what the proportional changes mean in relation to actual numbers of birds.

A4.2 Blyth Offshore Windfarm

Rothery, P., Newton, I., Little B. (2009) Observations of seabirds at offshore wind turbines near Blyth in northeast England. *Bird Study* **56,** 1-14

Methods

Shore based observations were undertaken between 18 April 1998 and 30 August 2003 covering the pre-construction, construction and post-construction periods of Blyth Offshore Windfarm. Observations were carried out at pre-determined times, at least twice a month. All passing birds were recorded, and it was stated that all birds were visible at a range of 1 km, although the turbines are only likely to comprise a small part of the total observation area. All birds were assigned to one of four height categories – 0-9.1 m, 9.1-26.4 m, 26.4-92.4 m and >92.4 m. A total of 70.3 hours of monitoring were available for the pre-construction period and 351.6 hours for the post-construction period, although no analyses were undertaken to assess the significance of any changes in flight height.

Seasons / time of day

Data were collected throughout the year and during daylight hours.

Species

Northern gannet (432 birds post-construction), great cormorant (352 birds post-construction), common scoter (341 birds post-construction), common eider (1,034 birds post-construction), black-headed gull (978 birds post-construction), herring gull (1,408 birds post-construction), great black-backed gull (564 birds post-construction), black-legged kittiwake (1,350 birds post-construction), Sandwich tern (2,135 birds post-construction).

Conditions data collected under

Data collected under all conditions in which visibility was at least 1 km.

Location / habitat

A shallow spit, approximately 1 km from shore.

Turbine / array specification

Two turbines spaced 200 m apart with a hub height of 59.4 m above mean sea-level and a rotor diameter of 66 m.

Results

For each species, the change in the proportion of birds flying at altitudes greater than 9.1 m above mean sea-level pre and post-construction are available. For most species, a greater proportion of birds fly above 9.1 m post-construction than pre-construction. The increase in the proportion of gulls flying above 9.1 m varied from 114-238% during the summer and 267-2,900% in the winter. Similarly during the summer, the proportion of gannets flying above 9.1 m increased by 2,800%.

Assessment of methodology

Despite the authors' assurances, it is unlikely that all birds were detected over the full range of the observation area. In particular, birds at lower altitudes may be obscured by waves, or be less visible against the sea surface. As a result, the proportion of birds at lower altitudes may have been under-estimated. In addition, the change in observation platform between pre- and post-construction periods is likely to have afforded an improved view of the observation area. These factors mean that pre- and post-construction comparisons of the estimates of birds at different altitudes may not be reliable. In addition, the presence of the turbines offering a fixed structure with which to assess birds' flight heights against, is likely to have improved the accuracy of estimates of flight heights made post-construction. Finally, by limiting the comparison to birds above 9.1 m, well below the rotor sweep of the turbines, the proportion of birds at risk is likely to be vastly over-estimated.

A4.3 Egmond aan Zee

Krijgsveld, K.L., Fijn, R.C., Japink, M., van Horssen, P.W., Heunks, C., Collier, M.P., Poot, M.J.M., Beuker, D., Dirksen, S. 2011. *Effect studies offshore wind farm Egmond aan Zee. Final report on fluxes, flight altitudes and behaviour of flying birds.* Bureau Waardenburg

Methods

Visual observations

Between spring 2007 and December 2009, 405 panorama scans were carried out from a met mast on the edge of the Egmond aan Zee Windfarm. Scans were undertaken once an hour during daylight covering a 360˚ angle around the windfarm with a pair of 10 x 42 binoculars fixed on a tripod. During each observation period, two scans were undertaken, the first to capture birds close to the sea surface and the second to capture birds at greater altitudes. The height of birds was estimated using trigonometry to combine the distance and angle between the bird and observer. Birds could be viewed to a distance of up to 3 km, although imperfect detection is likely to be an issue at these distances. The area covered by each panorama scan is approximately 50% within the windfarm and 50% outside, allowing for simple comparisons to be made of birds inside and outside of the windfarm, although differences were not assessed statistically.

Seasons / time of day

Data were collected during daylight, throughout the year. There was increased effort during the spring and autumn migration periods.

Species

Northern gannet, great cormorant, black-legged kittiwake, black-headed gull, common gull, lesser black-backed gull, herring gull, great black-backed gull, Sandwich tern, small gull spp., large gull spp., gull spp.

Conditions data collected under

Data collected under all conditions.

Location / habitat

Marine 10 km offshore.

Turbine / array specification

Egmond aan Zee Offshore Windfarm covers an area of 27 km^2 and contains 36 turbines. Each turbine has a hub height of 70 m and rotor diameters of 90 m. Turbines are arranged in four rows, with 650 m between turbines in each row and 1 km between rows.

Results

Species varied in their vertical responses to wind turbines. Of the 13 species or groups considered, the proportion flying at rotor height was lower inside the windfarm than outside for kittiwake, black-headed gull, northern gannet, great black-backed gull, Sandwich tern and unidentified gull species (no numbers were presented). Large gulls appeared to show little, or no vertical response to the turbines, with roughly the same proportion flying at rotor height inside as outside. In contrast, the proportions of great cormorants, common gulls, little gulls and other small gulls flying at rotor height showed a noticeable increase inside the windfarm.

Assessment of methodology

Data are presented as the proportions of birds at rotor height both within and outside the windfarm. Without any details on the number of birds involved, it is difficult to determine the strength of these data, and the subsequent findings. Of particular concern is the way in which data for unidentified gulls have been presented and the apparent inconsistency in the results for each category which show roughly the same proportion of unidentified large gulls at rotor height inside as outside the windfarm, more small gulls at rotor height inside than outside the windfarm, but unidentified gulls assigned to neither category significantly less likely to be at rotor height within the windfarm. Without more details of the species likely to be covered by each category, and their abundance within the study area, it is difficult to assign levels of confidence to the results presented.

A4.4 Gunfleet Sands I and II

RPS. 2008. *Gunfleet Sands Monitoring Report*, RPS, London

NIRAS. 2011. *Gunfleet Sands 2 Offshore Wind farms. Year 1 Post-construction Ornithological monitoring*. NIRAS, Cambridge

GoBe Consultants Ltd. 2012. *Gunfleet Sands Offshore Wind farm I & II – Post Construction Year 2 – Marine Licence Environmental Monitoring Report.* Prepared for DONG Energy.

Methods

Boat surveys

Pre- and post-construction monitoring data were collected as part of boat surveys following standardised methodologies (Camphuysen *et al.* 2004). Flying birds were assigned to one of the following flight height bands <5 m, 5- 15 m, 15 -150 m. Pre-construction surveys were carried out between October 2007 and March 2008. Post-construction surveys were carried out between October 2010 and March 2011 and between October 2011 and March 2012. However, differences were not assessed statistically.

Seasons / time of day

Data were collected over winter, during periods of daylight.

Species

Red-throated diver, black-headed gull, common gull, great black-backed gull, gull spp., herring gull, black-legged kittiwake, lesser black-backed gull

Conditions data collected under

No details given.

Location / habitat

Gunfleet Sands I & II offshore windfarms, approximately 7 km from the coast.

Turbine / array specification

Gunfleet Sands I and II contain 48 turbines between them, each with a hub height of 75 m and a rotor diameter of 107 m. The projects cover a total area of 16 km^2.

Results

The proportion of red-throated divers flying at collision risk height declined following the construction of the windfarm, by 39% in winter 2010/11 and by 96% in winter 2011/12. In contrast, the proportion of great black-backed gulls at rotor height showed an increase following construction, by 75% in winter 2010/11 and 53% in winter 2011/12. The proportion of herring gulls at rotor height showed little change between pre-construction years and either post-construction survey. Results for other species were less consistent. For example common gulls showed an increase in the proportion at rotor height in 2010/11 compared to pre-construction data, but a decrease in 2011/12.

Assessment of methodology

Data were collected following a relatively robust methodology and the height bands used were a reasonable match for the dimensions of the rotor swept area of each turbine meaning the proportions of birds at risk height are less likely to be significantly over-estimated. However, the limited duration of pre- and post-construction surveys, reflected in the quantity of data available, means that there may only be limited power to detect significant changes in species flight heights.

A4.5 Nysted/Horns Rev

Blew, J., Hoffman, M., Nehls, G., Hennig, V. 2009. *Investigations of the bird collision risk and the responses of harbour porpoises in the offshore windfarms Horns Rev, North Sea, and Nysted, Baltic Sea, in Denmark*.

Methods

X-Band Radar

The spring and autumn migration periods were monitored at Horns Rev and Nysted in 2005 and 2006 using x-band radar mounted on vessels anchored in each windfarm. In total, across both windfarms 71.5 days of monitoring were carried out during the spring and 93.5 days during the autumn. Data were captured up to a height of 1,500 m and movements were examined in two height bands <200 m and 200-500 m. All birds tracked for > 100 m and showing a change in movement of >20 m were considered to have changed altitude.

Seasons / time of day

Data were collected throughout spring and autumn in 2005 and 2006.

Species

Having used radar, it was not possible to determine the species captured by the radar.

Conditions data collected under

It was not possible to collect data during periods of strong wind or heavy rain. However, all other conditions were covered.

Location / habitat

Horns Rev 17.9 km from the Danish North Sea Coast.
Nysted 10.8 km from the Danish Baltic Sea Coast.

Turbine / array specification

Horns Rev is an array of 80 turbines covering an area of 21 km^2. Each turbine has a hub height of 70 m and a rotor diameter of 80 m.

Nysted is an array of 72 turbines covering an area of 26 km^2. Each turbine has a hub height of 69 m and a rotor diameter of 82 m.

Results

Across both windfarms, and within the 0-200 m observation band, 4.8% of birds flying towards the windfarm were shown descending by more than 20 m and 13.4% were shown ascending by more than 20 m during the day time. At night time, the values were 2.9% and 13.6% respectively. However, these proportions did not differ significantly from the observations within the 200-500 m band, suggesting that the change in flight heights did not differ from what may be expected to occur by chance and are therefore unlikely to reflect avoidance behaviour.

Assessment of methodology
The rotor-swept area of each turbine covers altitudes from 20-110 m. Consequently, as data were relatively coarse and restricted to all flights within a band of 0-200 m, it

may not have been possible to detect responses to turbines. In addition, having used radar, any responses to turbines that had been recorded could not have been identified to species level.

A4.6 Robin Rigg

Natural Power Consultants. 2013. *Analysis of Marine Environmental Monitoring Plan Data from the Robin Rigg Offshore Wind farm, Scotland (Operational Year 3).* Natural Power, Castle Douglas.

Methods

Boat-based surveys

Pre- and post-construction boat surveys were carried out within the windfarm following standard methodologies (Camphuysen *et al.* 2004). Birds in flight were assigned to bands of 0-5 m, 6-25 m, 26-34 m, 35-125 m, 126-200 m and >200 m. Surveys were carried out on a bi-monthly basis during pre-construction monitoring (2001-2007), and on a monthly basis during post-construction monitoring (2010-2011). Where sufficient data were available, differences in the proportions of birds flying at rotor height were assessed using a chi-squared test.

Seasons / time of day

Surveys were carried out throughout the year, during daylight.

Species

Common scoter, red-throated diver, diver spp., Manx shearwater, northern gannet, great cormorant, black-legged kittiwake, herring gull, great black-backed gull, gull spp, common guillemot, razorbill, auk spp.

Conditions data collected under

No details given.

Location / habitat

Robin Rigg Offshore Windfarm, 11 km from shore.

Turbine / array specification

Robin Rigg is an array of 60 turbines, each with a hub height of 80 m and a rotor diameter of 88 m. The turbines are spaced at intervals of approximately 500 m.

Results

There were no significant differences in the proportions of birds flying at rotor height during pre- and post-construction surveys for common scoter and red-throated diver. However, the proportion of northern gannet, great cormorant, black-legged kittiwake

and large gull species flying at rotor height within the windfarm all increased between pre- and post-construction. However, the low power of the data was noted raising concerns over the validity of the results.

Assessment of methodology

Flight height data were not collected following the standard ESAS methodology and concerns are raised that this is likely to lead to a double counting of individuals, meaning estimates of changes in the proportion of birds at collision risk height may not be reliable.

APPENDIX 5 EVIDENCE REVIEW MICRO-AVOIDANCE

A5.1 Egmond aan Zee

Krijgsveld, K.L., Fijn, R.C., Japink, M., van Horssen, P.W., Heunks, C., Collier, M.P., Poot, M.J.M., Beuker, D., Dirksen, S. 2011. *Effect studies offshore wind farm Egmond aan Zee. Final report on fluxes, flight altitudes and behaviour of flying birds.* Bureau Waardenburg

Methods

Between July and December 2009, the flight paths of birds around six turbines were observed visually. These flight paths were then related to short range radar tracks in order to estimate the altitude and distance to nearest turbine. As a result, a dataset containing high resolution observations of bird behaviour around turbines was created. Birds were assigned to 5 m horizontal distance bands beginning at the rotor hub. All birds flying between 20 and 120 m above sea-level (reflecting the rotor-swept area of each turbine) were considered to be at risk of collision and the number of birds within each 5 m band was compared to the number of birds that would have been expected if they had been distributed evenly. To assess the level of last-second avoidance action taken, the number of birds within the 45-50 m band (just outside the rotor-sweep) was compared to the number of birds recorded between 0 and 45 m from the rotor hub.

Seasons / time of day

Data were collected during daylight on eight occasions between July and December.

Species

Seabirds, waterbirds and other migrants.

Conditions data collected under

All conditions.

Location / habitat

Marine, 10 km offshore.

Turbine / array specification

Egmond aan Zee Offshore Windfarm covers an area of 27 km^2 and contains 36 turbines. Each turbine has a hub height of 70 m and rotor diameters of 90 m. Turbines are arranged in four rows, with 650 m between turbines in each row and 1 km between rows. The study of micro-avoidance covered six turbines at the edge of the windfarm.

Results

Whilst 1,610 birds in 409 groups were recorded over the course of the study, only 115 in 52 groups were recorded passing within 50 m of the turbines. Of these, only 36 birds were recorded between 20 and 120 m, at heights placing them at risk of collision. Of the 36 birds passing within 50 m of the turbine and at rotor height, it is reported that 0.926 did not fly within the rotor swept window of the turbine (i.e. 2-3 birds). This would reflect a micro-avoidance rate of 0.926.

Assessment of methodology

The described methodology of combining visual and radar observations to record the tracks of birds approaching turbines is robust. This makes it possible to relate tracks to individual species and to determine how close each individual, or flock, gets to a turbine. Focussing on the area 50 m either side of the rotor hub and comparing the proportion in the 45-50 m band to the proportion in the 0-45 m band data is likely to capture the type of last-minute action covered by micro-avoidance.

However, only limited weight can be given to the data presented here. Observations were recorded on only four days, during which only 36 birds were recorded passing within 50 m of the turbine, the distance presented to represent micro-avoidance. This figure may be substantially inflated as it includes a single observation of a flock of 28 skylark.

A5.2 Greater Gabbard

RPS. 2011. *Galloper Wind farm Project Environmental Statement – Technical Appendices 2: Appendix 4: Greater Gabbard post-construction vantage point surveys*, RPS, Glasgow

Methods

Visual Observations

Two surveyors collected data from 180˚ arcs to the port and starboard sides of a stationary vessel within Greater Gabbard Offshore Windfarm. Each arc had a radius of 2 km and all birds entering each arc were recorded during snapshot counts taken every 15 seconds. The location of the boat and the viewing area, which covered a total of 15.9 km^2, included seven operational turbines and a total of 36 hours of data were collected during the survey. The flight paths of each bird within the viewing area were noted, as was the proportion of time each bird spent at different heights.

Seasons / time of day

Data were collected between 1st June 2011 and 28th July 2011, with each survey lasting four hours.

Species

Northern gannet (0.14 birds/hr), Arctic skua (0.03 birds/hr), lesser black-backed gull (3.69 birds/hr), herring gull (0.11 birds/hr), black-legged kittiwake (1.28 birds/hr).

Conditions data collected under

Conditions were limited to sea-states one and two, to ensure the vessel remained as a stable observation platform.

Location / habitat

Greater Gabbard, UK (offshore).

Turbine / array specification

The survey monitored seven operational turbines, each with a hub height of 77.5 m and a rotor diameter of 107 m.

Results

Over the course of the study period, 190 flights through the area were recorded. Of these, the vast majority did not pass close to the turbines. Given the proportion of the total study area occupied by turbines, this is unsurprising. As a consequence, only a single evasive manoeuvre, involving a kittiwake, was recorded.

Assessment of methodology

The length of the observation periods carried out during this study were extremely limited, so it is difficult to make an accurate assessment of how widespread different avoidance actions are. In addition, records of avoidance action have been made in a subjective fashion, both in relation to assessing the number of birds on a collision course for the turbines, and in assessing the actions recorded. For these reasons, it is not possible to quantify the micro-avoidance behaviour reported in this study.

A5.3 Kessingland Windfarm

Wild Frontier Ecology. 2013. *Kessingland Windfarm Annual Post-construction Monitoring Report Year 2.* Wild Frontier Ecology, Norfolk

Methods

Bird activity was monitored within the windfarm through nine two-hour vantage point surveys at each turbine carried out between November 2012 and March 2013. In total 36 hours of survey effort was completed throughout the study period. The response of birds whose flight paths were likely to overlap with turbines was noted.

Seasons / time of day

Late morning – early afternoon during winter.

Species

Black-headed gull (97 birds/hr), common gull (31.4 birds/hr), lesser black-backed gull (11 birds/hr),

herring gull (56.72 birds/hr), great black-backed gull (0.28 birds/hr).

Conditions data collected under

No details given.

Location / habitat

Kessingland, Suffolk, UK (terrestrial).

Turbine / array specification

Two turbines with hub heights of 80 m and rotor diameters of 92 m. Distance between turbines within each row is not described.

Results

All birds recorded as being on a collision course with the turbines were observed to take evasive action to avoid collision. Typically this action occurred at a distance of 0-50 m from the turbine. Over the course of the study period, five black-headed gulls, two lesser black-backed gulls and a herring gull were recorded taking evasive action. In three instances this involved a change in altitude to fly below the rotor blades, whilst in other instances it involved a change to flight direction. In the case of the two lesser black-backed gulls, both were observed to take last minute evasive action at just five metres from the blades.

Assessment of methodology

The length of the observation periods carried out during this study were extremely limited, so it is difficult to make an accurate assessment of how widespread different avoidance actions are. In addition, records of avoidance action have been made in a subjective fashion, both in relation to assessing the number of birds on a collision course for the turbines, and in assessing the actions recorded and the distances at which they occur. For these reasons, it is not possible to quantify the micro-avoidance behaviour reported in this study.

A5.4 Nysted

Desholm, M. 2005. *TADS investigations of avian collision risk at Nysted offshore wind farm, autumn 2004*. NERI, Denmark

Petersen, I.K., Christensen, T.K., Kahlert, Desholm, M., Fox, A.D. 2006 *Final results of bird studies at the offshore wind farms at Nysted and Horns Rev, Denmark*, NERI, Denmark

Methods

Using a Thermal Animal Detection System (TADS) all bird movements past a single turbine during spring and autumn 2004 and spring and autumn 2005 were recorded. Birds were detected at distances of up to 120 m.

Seasons / time of day

Data were collected throughout both day and night in the spring and autumn.

Species

Mostly migrant passerines and waterbirds.

Conditions data collected under

All conditions.

Location / habitat

Located approximately 11 km offshore in the Danish part of the Baltic Sea.

Turbine / array specification

An array of 72 turbines arranged in eight rows of nine turbines each. Turbines have a hub height of 69 m and a rotor diameter of 92 m.

Results

In over 123 days of continuous monitoring, cameras captured 5,507 video sequences of which only 14 were found to include birds. Of these, none revealed birds passing close to the turbine.

Assessment of methodology

The methodology is robust with sufficient capability to record all birds passing the turbine over the study period. However, the low frequency with which birds were recorded passing close to the turbine suggests that the data are unlikely to have sufficient power to detect avoidance activity.

APPENDIX 6 EVIDENCE REVIEW WITHIN-WINDFARM AVOIDANCE

A6.1 Avonmouth Docks

The Landmark Practice. 2013. *Birds and Wind Turbines At Avonmouth Docks. Year 5 Monitoring Report for Ecotricity.* The Landmark Practice, Bristol

Methods

Monitoring was undertaken at the Avonmouth Docks windfarm between October and March in the winters of 2007/08, 2008/09, 2009/10, 2011/12. Three vantage point surveys, each lasting three hours, were carried out in each month to record bird activity at the site. Flight altitude was estimated in five bands 0-20 m, 20-40 m, 40-80 m, 80-160 m and >160 m.

During the visits for each vantage point survey, a search with a radius of 60 m around each turbine was carried out for corpses. Additional surveys were carried out following periods of severe weather. In total 343 checks were carried out around the base of each turbine in the post-construction period.

Seasons / time of day

Vantage point surveys were carried out between October and March, and timed so that periods of rising, falling and high tide were covered each month.

Species

Black-headed gull (4.4 birds/hr 2007/08, 7.1 birds/hr 2008/09, 2.9 birds/hr 2009/10, 12.8 birds/hr 2011/12), herring gull (6.8 birds/hr 2007/08, 13 birds/hr 2008/09, 18.8 birds/hr 2009/10, 38.2 birds/hr 2011/12)

Conditions data collected under

Not stated.

Location / habitat

Avonmouth Docks, coastal.

Turbine / array specification

A line of 3, 2 MW Enercon E82 turbines, with a hub height of 79m and a rotor diameter of 83 m.

Results

A single black-headed gull was identified as a probable collision victim in the winter of 2007/08. An average of 4.4 black-headed gulls were recorded passing through the site over the study period, suggesting a total flux rate of 10,530 birds, of which 57 were predicted to collide based on option 1 of the Band model, 2 were predicted to

collide based on option 2 of the Band model and 1 was predicted to collide based on option 3 of the Band model. This reflects avoidance rates of 0.9826 using option 1 of the Band model, 0.5152 using option 2 of the Band model and -0.0005 using option 3 of the Band model.

Assessment of methodology

The corpse search methodology is likely to provide an accurate estimate of collision numbers as previous studies have shown that the majority of corpses are recovered within 40 m of a turbine base (Orloff & Flannery 1992, Munster *et al.* 1996, Howell 1997). Furthermore, corpses were examined to confirm collision as cause of death. No corrections were carried out to account for searcher efficiency or predator activity. However, given the habitat surrounding the turbines and the frequency of searches through the study period, it is unlikely corpses would have been missed. Bird activity surveys were carried out throughout the study period and are therefore likely to give a realistic impression of bird activity in the area.

As the bird activity surveys were carried out concurrently with the corpse searches and covered the same area, these data were combined with data from other sites to estimate representative avoidance rates.

A6.2 Altamont Pass

Thelander, C.G., Smallwood, K.S. & Rugge, L. 2003. *Bird risk behaviours and fatalities at the Altamont Pass Wind Resource Area*. National Renewable Energy Laboratory, Colorado.

Methods

Visual observations and fatality searches.

Circular areas with a 50 m radius around the base of 685 wind turbines were searched for corpses every five to six weeks between 1998 and 2000. These searches were combined with 1,958 30 minute point counts carried out in 20 study plots on 303 different days between 1998 and 2000.

Seasons / time of day

Counts carried out throughout the year and between 0700 h and dusk.

Species

Gulls (0.48 birds/hour).

Conditions data collected under

All conditions unless wind or rain resulted in visibility dropping to <60 m.

Location / habitat

Altamont Pass, California, U.S.A. (terrestrial).

Turbine / array specification

685 turbines arranged in 109 rows across an area of 50 km^2. Turbine hub heights ranged from 14 m-30 m, with rotor diameters of 17-23 m. Distance between turbines within each row is not described.

Results

At this site, a total of five gulls, of unknown species, were recovered following collision with turbines. Across the study plots as a whole, the average rate at which gulls passed through the windfarm was 0.48 birds per hour, reflecting a total of 7,428 gull movements within the area over the two year study period. Site specific flight height data were not available, so it was not possible to calculate an avoidance rate based on option 1 of the Band model. Assuming no avoidance behaviour, and a bird with the characteristics of a herring gull, the total number of collisions expected would have been 296 per annum under option 2 of the Band model and 295 under option 3 of the Band model. The collision rate of five birds over the study therefore indicates a within-windfarm avoidance rate of 0.9831 using option 2 and 0.9831 using option 3. The similarity between these values reflects the relatively small size of the turbines installed at the site, in particular the rotor-swept area, diameters of 17-23 m are significantly smaller than many of the turbines installed at offshore sites.

Assessment of methodology

The corpse search methodology is likely to provide an accurate estimate of collision numbers as previous studies have shown that the majority of corpses are recovered within 40 m of a turbine base (Orloff & Flannery 1992, Munster *et al.* 1996, Howell 1997). Furthermore, corpses were examined to confirm collision as cause of death. Correction factors were applied to account for carcass removal by scavengers, but not to correct for searcher efficiency. However, the limited size of the search area and terrain made it unlikely that any corpses would have been undetected.

To minimise the effects of observer bias in point counts, paired observations were carried out during the early part of the study period so that different observers calibrated their perceptions of altitude, distance and behaviour with one another. However, no correction was applied for the detection distance of different species. This is a concern given that study plots were up to 4 km^2, meaning that the total number of birds present within the study areas may have been an underestimate and that, therefore, the final, derived avoidance rate would also have been an underestimate.

However, as it has been necessary to extrapolate bird activity data across the site, this has not been combined with data from other sites to identify representative avoidance rates.

A6.3 Blyth Harbour

Visual observations and fatality searches.

Lawrence, E.S., Painter, S. & Little, B. 2007. Responses of birds to the wind farm at Blyth Harbour, Northumberland, UK. In de Lucas, M., Janss, G.F.E., Ferrer, M. 2007. *Birds and Wind farms Risk Assessment and Mitigation*, Quercus, Madrid.

Newton, I. & Little, B. 2009. Assessment of wind farm and other bird casualties from carcasses found on a Northumbrian beach over an 11-year period. *Bird Study*, **56,** 158-167.

Methods

Once a week over an 11 year period, a 4.7 km stretch of beach near Blyth in Northumberland was searched for corpses. Depending on the condition of the birds, an attempt was made to assign a cause of death to each carcass, and those with symptoms thought to be typical of collision with a wind turbine – head or one or both wings missing, broken bones blood in body cavity and a ruptured liver – were identified. The total number of carcasses found was then corrected to account for those lost to scavengers, those not washed up on the beach and those not found during searches.

Between October 1996 and August 1998, 31 three hour-long periods of observation were made of flight activity perpendicular to the turbine row and in the vicinity of five of the nine turbines. Observations were made from a point on the shore opposite the turbines, at a distance of approximately 80 m. In total 93 hours of observational data were collected.

Seasons / time of day

Fatality data were collected throughout the year. Bird activity data were also collected throughout the year, between the hours of 0800 and 1500 h, with observation periods split equally between the morning and afternoon.

Species

Around 80% of the flight activity within the windfarm involved herring gull and great black-backed gull, and other gull species made up a significant proportion of the remaining species. However, as species-specific data were not available regarding the corpses collected and it was stated that the majority of those collected belonged to gulls, to calculate an avoidance rate, it was necessary to consider gulls collectively.

Conditions data collected under

No details given.

Location / habitat

Blyth Harbour breakwater, Northumberland, UK (coastal).

Turbine / array specification

Nine turbines arranged in a row along a harbour breakwater. The turbines are spaced at 200 m intervals and have a hub height of 25 m with a 25 m rotor diameter.

Results

Results were presented as average collision rates and passage rates over the study period as a whole. Based on the data presented an average of 417,954 birds, most of which were large gulls, would have been expected to pass through the windfarm over the study period. Of these, approximately 3,047, assuming birds with the characteristics of a herring gull, would have been expected to collide with turbines in the absence of avoidance behaviour using option 1 of the Band Model and 3,083 using option 2 and 3,007 using option 3. Having corrected for the imperfect detection of corpses, between 148.5 and 193.5 collisions with wind turbines were expected in an average year. This suggests a within-windfarm avoidance rate of 0.3966-0.5369 using option 1, 0.4037-0.5423 using option 2 and 0.3886-0.5308 using option 3.

Assessment of methodology

The fatality searches were intensive throughout the study period and followed a robust methodology to account for corpses that went undetected. In particular, the potential for corpses to wash up within the study area was tested experimentally.

The observational data were limited to a two year period in the middle of the study. The data may have underestimated gull movements within the surrounding area for two key reasons. Firstly, no corrections were applied to account for imperfect detection of birds. Secondly, by limiting observations to the period between 0800 and 1500 h, key movements of gulls to and from roost sites may have been missed during the summer and autumn. Underestimating bird activity within the area would lead to an underestimate of the number of collisions expected in the absence of avoidance behaviour, and consequently, the final derived avoidance rates would also be underestimated.

Activity data were only collected between 206 and 2008 and only between turbines 5 and 9. As the mean annual collision rates relate to the whole of the study period, and to all 9 turbines, it is necessary to extrapolate activity data both temporally and spatially to derive a flux rate. Therefore, these data have not been included when deriving representative avoidance rates.

A4.4 Blyth Offshore Windfarm

Rothery, P., Newton, I. & Little, B. 2009. Observations of seabirds at offshore wind turbines near Blyth in northeast England. *Bird Study*, 56, 1-14

Methods

Visual observations

Following the installation of the offshore turbines, observations of birds in the vicinity of the turbines were made on 177 occasions between 12 January and 30 August

2003, totalling almost 352 hours of observation. Observations were made from the shore and distances and heights of flying birds were calibrated against objects of known size and fixed locations.

Seasons / time of day

Observations were made between January and August. Data collection was focussed on the period between 1130 and 1600, consequently, during the summer movements to and from breeding colonies may have been missed.

Species

Northern gannet (1.23 birds/hr), great cormorant (1 bird/hr), common scoter (0.96 birds/hr), common eider (2.77 birds/hr), black-headed gull (2.78 birds/hr), herring gull (4 birds/hr), great black-backed gull (1.6 birds/hr), black-legged kittiwake (3.83 birds/hr), Sandwich tern (6.07 birds/hr).

Conditions data collected under

No details given.

Location / habitat

Blyth, Northumberland, UK (offshore).

Turbine / array specification

Two turbines separated by 200 metres. Each turbine had a hub height of 59.4 m above mean sea-level and a rotor diameter of 66 m.

Results

Throughout the study period, no collisions were recorded involving any of the species observed in the vicinity of the windfarm, reflecting a within-windfarm avoidance rate of 1.0000 for each species considered (Northern gannet, great cormorant, common scoter, common eider black-headed gull herring gull, great black-backed gull, black-legged kittiwake and Sandwich tern).

Assessment of methodology

No corrections were applied to account for the imperfect detection of birds during the survey. Consequently, the true level of bird activity within the study area was likely to have been underestimated. Additionally, it was not possible to search for carcasses, meaning that inferences about avoidance behaviour can only be drawn from the failure of observers to detect a collision from a total of 352 hours of monitoring. Given the low probability of a collision occurring, and the levels of flight activity recorded, this outcome is unsurprising. It is also important to note that the size of the OWF was very small (two turbines) and therefore caution must be applied when considering how applicable these avoidance rates are for much bigger arrays.

As insufficient observational data have been collected to record a collision, these data have not been included in those used to derive representative avoidance rates.

A6.5 Boudwijnkanaal

Everaert, J. 2008. *Effecten van windturbines op de fauna in Vlaanderen: Onderzoeksresultaten, discussie en aanbevelingen*. INBO, Brussels

Everaert, J. & Kuikjen, E. 2007. *Wind turbines and birds in Flanders (Belgium): Preliminary summary of the mortality research results.* INBO, Brussels

Everaert, J., Devos, K. & Kuijken, E. 2002. *Windturbines en vogels in Vlaanderen: Voorlopige onderzoeksresultaten en buitenlandse beviningen*. INBO, Brussels

Methods

Visual observations and fatality searches.

Systematic fatality searches were carried out once every 14 days between 2001 and 2006. Searches were carried out within a circular area, with a radius of 100 m, centred on each turbine. Corrections were applied to the data to account for imperfect detection and searcher efficiency.

Observational data describing the number of birds passing the turbine hub were collected between September and December 2005 between turbines 8 and 14. The resultant data were used to extrapolate the total number of birds likely to have passed the turbines over this period. Observational data are presented as a mean daily total collected during the period from two hours before dawn to four hours after dusk in October, reflecting a total of 17 hours of observations.

Seasons / time of day

Fatality data were collected throughout the year, behavioural data were collected between September and December.

Species

Gulls (1,075 birds/day).

Conditions data collected under

No details given.

Location / habitat

Boudwijnkanaal, Brugge, Belgium (terrestrial).

Turbine / array specification

A row of 14 turbines, each with a hub height of 55 m and a rotor diameter of 48 m. Distance between turbines within each row is not described.

Results

Collisions involving gulls were recorded in each year of the study, with a minimum of 21.2 collisions occurring in 2001 when only five of the 14 turbines were operational and a maximum of 264.6 collisions occurring in 2003, when all 13 turbines were operational. Behavioural data were only collected between September and December 2005 from between turbines 8 and 14. Extrapolating from these data to estimate the total number of collisions expected in each year in the absence of any avoidance action gives predictions of 550 collisions in 2001 using option 1 of the Band model, 252 using option 2 and 227 using option 3, and 3,262 collisions in each year between 2002 and 2006 using option 1, 1,497 using option 2 and 1,348 using option 3. Based on these analyses, within-windfarm avoidance rates would have been 0.9615 in 2001, 0.9299 in 2002, 0.9189 in 2003, 0.9284 in 2004, 0.9287 in 2005 and 0.9338 in 2006 using option 1. Using option 2, meso-micro avoidance rates would have been 0.9160, 0.8472, 0.8232, 0.8440, 0.8446 and 0.6990. Using option 3, meso-micro-avoidance rates would have been 0.9067, 0.8302, 0.8037, 0.8268, 0.8273 and 0.6656 respectively.

However, bird activity was only recorded around turbines 8 and 14 in October 2001 and October 2005. If we consider collisions recorded around these turbines in each of these time periods, the predicted number of collisions is 103 herring gulls in October 2001 and 145 black-headed gulls, 90 herring gulls and 260 birds in total during October 2005. The actual number of collisions recorded was 1, 6, 4 and 11 respectively, reflecting avoidance rates of 0.9903, 0.9586, 0.9556 and 0.9577 using option 1 of the Band model, 0.9789, 0.3658, 0.7865 and 0.8077 using option 2 of the Band model and 0.9765, 0.1886, 0.7629 and 0.7865 using option 3 of the Band model.

Assessment of methodology

Fatality data have been collected on a regular basis and following a robust methodology. Corrections have been applied to these data to account for the imperfect detection of corpses due to scavenger behaviour and searcher efficiency.

The observational data that have been collected are extremely limited. Data collection has been restricted to the September to December period in a single year. It is unclear how accurately this reflects bird movements within the windfarm over the rest of the study period. This may have a significant, but unquantifiable impact on the final, derived within-windfarm avoidance rates. In addition, it is unclear whether corrections have been applied to the observational data to account for the imperfect detection of birds.

Using the overall data, it is necessary to make both spatial and temporal extrapolations to estimate the avoidance rates. For this reason, we only use the data collected around turbines 8-14 in October 2001 and 2005 to derive representative avoidance rates.

A6.6 Bouin

Dulac, P. 2008. *Evaluation de l'impact du parc eolian de Bouin (Vendee) sur l'avifaune et les chauves-souris. Bilan de 5 anness de suivi*. Ligue pour la Protection des Oiseaux, Nantes.

Methods

Visual observations and fatality searches.

Weekly searches were carried out for corpses at the foot of turbines between 2002 and 2006. Searches were restricted to a 100 m^2 box centred on each turbine. To aid searching, each box was divided into a grid with squares of 25 m^2.

Observational data were collected from four points, covering 1 km each. Each month a two hour count was made from each point, with a total of 474 hours of observational data collected from the site as a whole between 2002 and 2006.

Seasons / time of day

Data were collected throughout the year and protocols were designed so that full day was covered.

Species

Black-headed gull (16.23 birds/hr), herring gull (2.26 birds/hr), other gulls (2.09 birds/hr).

Conditions data collected under

The observational protocol was designed to collect data throughout the tidal cycle and in all weather conditions.

Location / habitat

Bouin, Baie de Bourgneuf, France (Coastal)

Turbine / array specification

A single row of eight turbines, each with a hub height of 60 m and a diameter of 80 m. Distance between turbines within each row is not described.

Results

At this site, 30 gulls were recovered from turbine bases over the course of a four year study period. Of these, 28 were black-headed gulls, one was a yellow-legged gull and one was a Mediterranean gull. Using option 1 of the Band model, 584 black-headed gulls and 206 'other' gulls were predicted to collide with the turbines, reflecting avoidance rates of 0.9520 and 0.9903 respectively. For option 2, 483 and 354 birds were predicted to collide respectively, reflecting avoidance rates of 0.9421

and 0.9943. For option 3, the corresponding figures were 237 and 251 birds predicted to collide reflecting avoidance rates of 0.8820 and 0.9920. No collisions were recorded for herring gulls, despite a predicted collision rate of 216 per annum, reflecting a within-windfarm avoidance rate of 1 for options 1, 2 and 3 of the Band model.

Assessment of methodology

Fatality data were collected following a robust protocol, with corrections applied to account for birds lost to scavengers and search efficiency. The intensive nature of these searches, weekly over a four year period, is likely to mean that fatality rates were estimated with a high degree of accuracy.

Observational data were collected over a four year period. However, no corrections were applied to account for imperfect detection. Consequently, bird activity in the area and the derived within-windfarm avoidance rates were likely to have been underestimated.

As activity data were a spatial and temporal match for the period over which collision data were collected, these data were included when estimating representative avoidance rates.

A6.7 Buffalo Ridge

Johnson, G.D., Erickson, W.P., Strickland, M.D., Shepherd, M.F., Shepherd, D.A. 2000. *Avian Monitoring Studies at The Buffalo Ridge, Minnesota Wind Resource Area: Results of a 4-year study.* Western EcoSystems Technology Inc., Wyoming.

Methods

Visual observations and fatality searches.

Fatality searches were carried out within 126 m x 126 m plots, centred on 61 turbines. Searches were carried out every two weeks and observers covered the area by walking parallel transects separated by a distance of 6 m. This was combined with a series of large bird counts carried out every two weeks for a 0.8 km radius surrounding each of six observation stations. During each survey, two 30 minute observations were made, one in the morning and one in the afternoon. In total 70 hours of survey data were collected over the course of the study period.

Seasons / time of day

Fatality searches were carried out throughout the year. Large bird counts were carried out between 0800 and 1600 h and restricted to the period from 15 March to 15 November.

Species

Herring gull (0.1 birds/hour).

Conditions data collected under

No details given.

Location / habitat

Buffalo Ridge, Minnesota, U.S.A. (terrestrial).

Turbine / array specification

143 turbines arranged in 26 rows with between 100 m and 200 m between each turbine. Each 750 kW turbine had a hub height of 50 m and a diameter of 48 m.

Results

At this site, one herring gull was recovered following collision with turbine. Across the study plots as a whole, the average rate at which herring gulls passed through the windfarm was 0.03 birds per hour, reflecting a total of 625 gull movements within the area over the two year study period. Assuming no avoidance behaviour, the total number of collisions expected would have been 3 using option 1 of the Band model, 5 under option 2 of the Band model and 5 under option 3 of the Band model. The collision rate of 1 bird over the study therefore indicates a within-windfarm avoidance rate of 0.6503 using option 1, 0.8149 using option 2 and 0.7923 using option 3.

Assessment of methodology

The methodology was generally sound with a well-structured search likely to detect all corpses within the study area. Corrections were made for both corpses removed by scavengers and also searcher efficiency. The large bird survey also followed a sound methodology, with corrections applied to account for imperfect detection. However, as observations were limited to 0800 to 1600 h and November to March, it is possible that they failed to detect daily or seasonally important gull movements. This may reflect the fact that raptors were the primary concern at this site.

As it was necessary to extrapolate bird activity data spatially to estimate an avoidance rate, these data have not be included when deriving representative avoidance rates.

A6.8 De Put

Everaert, J. 2008. *Effecten van windturbines op de fauna in Vlaanderen: Onderzoeksresultaten, discussie en aanbevelingen*. INBO, Brussels

Everaert, J. & Kuikjen, E. 2007. *Wind turbines and birds in Flanders (Belgium): Preliminary summary of the mortality research results.*

Methods

Visual observations and fatality searches.

Systematic fatality searches were carried out once every 14 days between April 2005 and March 2006. Searches were carried within a circular area, with a radius of 100 m, centred on each turbine. No correction factors were used to account for scavengers or imperfect searcher efficiency.

Observational data describing the number of birds passing within 100 m of the turbine hub were collected between January and February 2006, the period in which the corpses were recovered. The resultant data were used to estimate the total number of birds likely to have passed the turbines over this period. In total 18 hours of survey data were collected.

Seasons / time of day

Observational data were collected throughout the day during January and February 2006.

Species

Black-headed gull and common gull (3,186 during the study period).

Conditions data collected under

No details given.

Location / habitat

De Put, Nieuwkapelle, Belgium (terrestrial).

Turbine / array specification

A row of two turbines, each with a hub height of 75 m and a rotor diameter of 100 m.

Results

In January and February 2006, the corpses of two gulls, one common gull and one black-headed gull, were recovered. Based on the number of birds estimated to have passed through the windfarm during the study period, the combined number of collisions predicted in these two species would be 19 using option 1 and none using options 2 and 3. The two recorded collisions therefore reflect a micro-meso avoidance rate of 0.8928 for common and black-headed gulls using option 1, -9.1051 using option 2 and -11.8383 using option 3.

Assessment of methodology

Whilst fatality searches appear to have been relatively robust and intensive throughout the study period, no corrections were applied to account for the imperfect detection of corpses, either through searcher inefficiency or through loss to scavengers. This may have led to an underestimate of the total number of collision victims.

Details of the methodology used to collect observational data of bird behaviour within the windfarm were sparse. In particular, no details were given of the length of observations used to collect data during the study. There also appears to have been no attempt to account for the imperfect detection of birds, meaning the total number passing through the study area may have been an underestimate. This, in turn would also mean that the final within-windfarm avoidance rate had been underestimated.

As bird activity and collision data have been collected concurrently, these data have been included when deriving representative avoidance rates.

A6.9 Gneizdzewo

Zielinski, P., Bela, G. & Marchlewski, A. 2008. *Report on monitoring of the wind farm impact on birds in the vicinity of Gniezdzewo (gmina Puck, pomorskie voivodeship)*

Zielinski, P., Bela, G. & Marchlewski, A. 2010. *Report on monitoring of the wind farm impact on birds in the vicinity of Gniezdzewo (gmina Puck, pomorskie voivodeship)*

Zielinski, P., Bela, G. & Marchlewski, A. 2011. *Report on monitoring of the wind farm near Gniezdzewo impact on birds (gmina Puck, pomorskie voivodeship)*

Zielinski, P., Bela, G. & Marchlewski, A. 2012. *Report on monitoring of the wind farm near Gniezdzewo impact on birds (gmina Puck, pomorskie voivodeship)*

Methods

Visual observations and fatality searches.

Collision surveys were carried out in the autumns of 2008, 2010, 2011 and 2012 (September-November). Corpse searches were carried out within 70 m radius of each turbine, on average every 2-3 days.

Over the same periods each year (mid-September – mid-November), activity surveys were carried out within the windfarm. Between 60 and 70 hours of observational data were collected each year, with observation sessions lasting up to 6 hours.

Seasons / time of day

Data were collected throughout the day during the autumn migration period in each year.

Species

Great cormorant (0.17-1.44 birds/hr), gulls (3.88-44.14 birds/hr), little gull (0.23 birds/hr), common gull (0.57 -1.73 birds/hr), black-headed gull (0.51-4.94 birds/hr), herring gull (1.06-5.39 birds/hr).

Conditions data collected under

All conditions

Location / habitat

Gniezdzewo, Poland (terrestrial).

Turbine / array specification

An array of 19 turbines arranged in four rows. Each turbine had a rotor diameter of 80 m and a hub height of 80 m.

Results

In the four autumns over which data have been collected, only a single collision involving a gull was recorded, a black-headed gull during the 2010 field season. No site specific flight height data were available, so it was necessary to use the distributions presented in Johnston *et al.* (2014a) and option 2 of the Band Model to estimate avoidance rates. In the 2010 field season, 460 black-headed gulls were predicted to have passed through the windfarm, with a predicted collision rate of 0.2 birds. The avoidance rate for black-headed gulls during autumn 2010 would, therefore, have been –3.9524, suggesting that a significant number of birds were attracted to the rotor swept area of the turbine. Using option 3 of the Band model, the collision rate was predicted to be 0.1 birds, reflecting a within-windfarm avoidance rate of -8.9238. However, it should be noted that this collision rate is based on a relatively low number of birds passing through the windfarm and as a result may be unreliable. The unusual nature of this result is confirmed as in three additional years of monitoring, no black-headed gull collisions were recorded, despite often higher levels of flight activity. The avoidance rate for cormorants and all other gull species in all years would have been 100%.

Assessment of methodology

The search for collision victims has been robust, with specially trained dogs used to increase detection. However, no corrections have been applied to account for birds lost to scavengers, potentially meaning the collision rates have been under-estimated.

No correction has been applied to the activity surveys to account for the imperfect detection of birds. As a consequence, the total number of birds passing through the area, and therefore potentially the final avoidance rates, may be under-estimated.

As collision and activity data were collected concurrently, from the windfarm as a whole, throughout the study period, they have been included when deriving representative avoidance rates.

A6.10 Greater Gabbard

RPS. 2011. *Galloper Wind farm Project Environmental Statement – Technical Appendices 2: Appendix 4: Greater Gabbard post-construction vantage point surveys*, RPS, Glasgow

Methods

Visual observations

Two surveyors collected data from 180˚ arcs to the port and starboard sides of a stationary vessel within Greater Gabbard Offshore Windfarm. Each arc had a radius of 2 km and all birds entering each arc were recorded during snapshot counts taken every 15 seconds. The location of the boat and the viewing area, which covered a total of 15.9 km^2, included seven operational turbines and a total of 36 hours of data were collected during the survey. The flight paths of each bird within the viewing area were noted, as was the proportion of time each bird spent at different heights.

Seasons / time of day

Data were collected between 1st June 2011 and 28th July 2011, with each survey lasting four hours.

Species

Northern gannet (0.14 birds/hr), Arctic skua (0.03 birds/hr), lesser black-backed gull (3.69 birds/hr), herring gull (0.11 birds/hr), black-legged kittiwake (1.28 birds/hr).

Conditions data collected under

Conditions were limited to sea-states one and two, to ensure the vessel remained as a stable observation platform.

Location / habitat

Greater Gabbard, UK (offshore).

Turbine / array specification

The survey monitored seven operational turbines, each with a hub height of 77.5 m and a rotor diameter of 107 m.

Results

The predicted number of collisions, in the absence of avoidance behaviour, within the 36 hour study period would have been less than 1 bird from each species. However, no collisions were recorded reflecting an avoidance rate of 1.000 for all species over the course of the study period.

Assessment of methodology

No corrections were applied to account for the imperfect detection of birds during the survey. Consequently, the true level of bird activity within the study area is likely to have been underestimated. Additionally, it was not possible to search for carcasses,

meaning inferences about avoidance behaviour can only be drawn from the failure of observers to detect a collision with 36 hours of monitoring. Given the low probability of a collision occurring, and the levels of flight activity recorded, this outcome is unsurprising.

Given the limited data collection during the study period, these data have not been included when deriving representative avoidance rates.

A6.11 Groettocht

Krijgsveld, K.L., Akershoek, K., Schenk, F., Dijk, F. & Dirksen, S. 2009. Collision risk of birds with modern large wind turbines. *Ardea* **97:** 357-366.

Methods

Radar observations and fatality searches.

Fatality searches were carried out within a 100 m radius around each turbine every 2-3 days. Searches were carried out by walking parallel transects, each separated by 4-6 m. Searches were carried out between October and December 2004.

Flight movements were quantified using a 12 kW x-band marine surveillance radar overnight between 1800 and 0700 h on 20 October 2004, 22 November 2004 and 22 December 2004, and the number of radar echoes up to 140 m (the maximum turbine height) were estimated as a measure of flux through the windfarm area.

Seasons / time of day

Resultant data reflect overnight collision rates of birds between October and December 2004.

Species

Key movements recorded included gulls travelling between Lake Ijsselmeer and a nearby roost site around dusk and dawn. However, amongst the five corpses encountered, there were only two gull carcasses, a common gull and a herring gull.

Conditions data collected under

Not specified.

Location / habitat

Agricultural area in the Netherlands.

Turbine / array specification

The array consists of a single line of seven turbines, each separated by 285 m. Turbines have a hub height of 78 m and a rotor diameter of 66 m. However, only the areas under five turbines were searched for carcasses.

Results

The average flux of birds through the area was 370 birds/km/hr, reflecting a movement of 873,534 birds through the study period as a whole. Site specific flight height data were not available for the site, so it was necessary to use the distributions presented in Johnston *et al.* (2014a) to estimate the proportion of birds at collision risk height, and option 2 of the Band model to estimate predicted collision numbers. In total, the remains of five birds (one herring gull, one common gull, one redwing, two unidentified species) were retrieved. Given that it is not possible to relate the radar tracks to individual species, we calculated the probability of collision based on a bird with the characteristics of first a herring gull, giving a predicted collision rate of 2131 birds over the study period, and an overall avoidance rate of 0.9991 based on option 2 and a collision rate of 1648 birds over the course of the study, with an avoidance rate of 0.9988 using option 3.

Assessment of methodology

The total collision rate may be an underestimate as the initial searching rate of once every three days was lowered to once every two days following the outcome of depredation tests. However, all corpses present were likely to be discovered as only turbines where the surrounding vegetation was low were searched for remains. With the exception of concerns over the depredation rate, the fatality searches were robust.

Flux rates were estimated using x-band radar, with the considerable disadvantage that it cannot be used to estimate the flux rates of different species. As a consequence, using individual species collision rates to estimate an avoidance rate may have led to an inaccurate estimate of the true value. In addition, as a single radar echo may represent multiple birds, there was a considerable risk that the true movement of birds through the area was underestimated and that, therefore, the overall avoidance rate was also underestimated.

As it was necessary to extrapolate activity data both spatially and temporally to estimate the avoidance rates, these data have not been used to derive representative avoidance rates.

A6.12 Haverigg

RPS. 2011. *Galloper Wind farm Project Environmental Statement – Technical Appendices 2: Appendix 3: Information on gull flight behaviour at operational wind farms and the estimation of avoidance rates for use in the Band Collision Risk Model*, RPS, Glasgow.

Methods

Visual observations

In July and August 42 hours of vantage point surveys were carried out at Haverigg Windfarm following the standard SNH vantage point methodology (SNH 2010).

Seasons / time of day

Surveys were carried out in July and August.

Species

Gulls (19.90 birds/hr).

Conditions data collected under

Not stated.

Location / habitat

Haverigg Windfarm, Cumbria, UK (terrestrial).

Turbine / array specification

Haverigg Windfarm consists of two groups of four turbines. The first four turbines have a hub height of 45 m and a rotor diameter of 42 m, whilst the remaining four, larger, turbines have a rotor diameter of 52 m.

Results

During 42 hours of vantage point observations, a total of 836 gulls, mostly herring and lesser black-backed gulls were recorded entering the windfarm at a rate of 19.90 birds/hr. However, during the observation periods, no collisions were recorded, reflecting an avoidance rate of 1 over the course of the study period under options 1,2 and 3 of the Band model.

Assessment of methodology

The evidence provided by the survey is limited as no corpse searches were carried out in the area surrounding the windfarm. Whilst 42 hours of survey effort were carried out, no collisions were recorded. However, given the likely rarity of collisions occurring, this is unsurprising. Furthermore, the levels of flight activity within the windfarm are likely to have been underestimated as no correction was made for the imperfect detection of birds.

As insufficient monitoring data have been collected to observe collisions, these data have not been included when deriving representative avoidance rates.

A6.13 Hellrigg

Percival, S. 2012. *Hellrigg Wind farm: Goose Refuge Monitoring Report Winter 2011-12*, Ecology Consulting, Durham

Percival, S. 2013. *Hellrigg Wind farm: Goose Refuge Monitoring Report Winter 2012-13*, Ecology Consulting, Durham

Methods

An area covered by a 100 m radius around the base of each turbine was searched between December and March in the winters of 2011/12 and 2012/13 on a weekly basis. Searches were carried out slowly and carefully, with particular care taken over areas containing large clumps of vegetation. The locations of each corpse were carefully noted, and each was left in place to provide information about decay rates and detectability.

Bird activity data were collected through vantage point surveys from a single point following standard SNH guidance. The flight lines of each species were noted and flight altitudes estimated. In total 38 hours of flight observations were collected in this way each winter.

Seasons / time of day

Data were collected between December and March each year, with effort made to cover dawn and dusk movements of birds as well as general daytime movements of birds.

Species

Common gull (8.47 birds/hr in 2011/12 and 507.17 birds/hr in 2012/13), lesser black-backed gull (0.3 birds/hr in 2011/12 and 0.41 birds/hr in 2012/13), herring gull (3.71 birds/hr in 2011/12 and 72.49 birds/hr in 2012/13), great black-backed gull (0.05 birds/hr in 2011/12 and 0.49 birds/hr in 2012/13), black-headed gull (4.79 birds/hr in 2011/12 and 131.48 birds/hr in 2012/13)

Conditions data collected under

Not stated.

Location / habitat

Hellrigg windfarm, onshore.

Turbine / array specification

An array of four turbines with a hub height of 80 m and a rotor diameter of 82 m.

Results

A single collision involving a herring gull was recorded in 2011/12. Based on the passage rate of 3.71 birds/hr, 13 collisions would have been expected in the absence of avoidance behaviour based on option 1 of the Band model, 3 collisions based on option 2 of the Band model and 2 collisions based on option 3 of the Band model. This reflects avoidance rates of 0.9209, 0.6635 and 0.5133 respectively.

Assessment of methodology

Analysis of the length of time corpses remained at the site, suggested that the mean time to disappearance was 22 days, well in excess of the 7 day search intervals. In combination with the systematic and methodical searches carried out at the site, this suggests it is unlikely any corpses went undetected.

Bird activity data were collected following standard SNH vantage point methodology. However, as no correction was made for imperfect detection, the levels of flight activity at the site and, therefore, the overall avoidance rates, may have been underestimated.

As mortality and activity data were collected concurrently at the site, following robust methodologies, these data were used when deriving representative avoidance rates.

A6.14 Keewaunee County

Howe, R.W., Evans, W. & Wolf, A.T. 2002. *Effects of Wind Turbines on Birds and Bats in Northeastern Wisconsin*. Report to Wisconsin Public Service Corporation and Madison Gas and Electric Company.

Methods

Visual observations and fatality searches.

Intensive searches were carried out between July 1999 and July 2001. Searches were carried out at least once a week. Surveyors visited a 60 m x 60 m area centred on each of the turbines and covered a series of nine 60 m transects in each. These searches were complemented by a series of 3,214 3 minute short counts carried out on 160 dates between 1998 and 2001, to estimate the number of birds within the area.

Seasons / time of day

Surveys were carried out between June and November, with a bias towards data collection during the morning.

Species

Herring gull (0.012 birds/hour), Franklin's gull (0.019 birds/hour), ring-billed gull (1.589 birds/hour).

Conditions data collected under

No details given.

Location / habitat

Keewaunee County, Wisconsin, U.S.A. (terrestrial).

Turbine / array specification

31 turbines with a hub height of 65 m and a rotor diameter of 47 m, within three clusters of 8, 9 and 14 turbines. Distance between turbines within each row is not described.

Results

At this site, one herring gull was recovered following collision with turbine. Across the study region as a whole, the average rate at which herring gulls passed through the area was 0.012 birds per hour, reflecting a total of 131 gull movements within the area over the two year study period. No site specific flight height data were available, meaning it was necessary to use the flight height distributions presented in Johnston *et al.* (2014a) and option 2 of the Band model. Assuming no avoidance behaviour, no collisions would have been expected under options 2 or 3 of the Band model. The collision rate of 1 bird over the study therefore indicates a within-windfarm avoidance rate of -12.0935 using option 2 and -13.5238 using option 3.

Assessment of methodology

The methodology was generally sound with a well-structured search likely to detect all corpses within the study area. Corrections were made for both corpses removed by scavengers and also searcher efficiency. However, no corrections were made to account for imperfect detectability during the bird surveys.

As it was necessary to extrapolate bird activity data spatially to estimate avoidance rates, these data have not been used to derive representative avoidance rates.

A6.15 Kessingland Windfarm

Wild Frontier Ecology. 2013. *Kessingland Wind farm Annual Post-construction Monitoring Report Year 2.* Wild Frontier Ecology, Norfolk.

Methods

Visual observations and fatality searches.

Fatality searches were undertaken around the bases of each turbine on nine occasions between November 2012 and March 2013. Surveyors walked a series of transects, separated by 10 m, within 65 m of the turbine base to search for corpses. A corpse correction factor of 1.79 was applied to account for corpses removed by scavengers.

Bird activity was monitored within the windfarm through nine two-hour vantage point surveys at each turbine carried out between November 2012 and March 2013. In total 36 hours of survey effort was completed throughout the study period.

Seasons / time of day

Data collection was carried out over winter 2012/13, between November and March. Surveys were carried out for two hour periods between 0800 and 1500 h.

Species

Black-headed gull (48.5 birds/hr), common gull (15.69 birds/hr), lesser black-backed gull (5.5 birds/hr), herring gull (28.36 birds/hr), great black-backed gull (0.14 birds/hr).

Conditions data collected under

No details given.

Location / habitat

Kessingland, Suffolk, UK (terrestrial).

Turbine / array specification

Two turbines with hub heights of 80 m and rotor diameters of 92 m. Distance between turbines within each row is not described.

Results

Black-headed, common, lesser black-backed, herring and great back-backed gulls were recorded within the study area at varying frequencies. Three gulls were found to have collided with the turbines – one black-headed gull, one common gull and one herring gull. After applying corpse correction factors, these estimates were revised to 1.79 birds of each species. No site specific flight height data were available, so it was necessary to use the modelled flight height distributions presented in Johnston *et al.* (2014a) and option 2 of the Band model. Given the number of birds likely to have passed through the windfarm during the study period, the predicted collision numbers would have been 28, 21 and 76 respectively. Using option 2, the avoidance rate for black-headed gull would therefore be 0.9367, for common gull it would be 0.9147 and for herring gull it would be 0.9764. Using option 3, the expected collision rates were 13, 12 and 51 respectively, reflecting avoidance rates of 0.8664, 0.8505 and 0.9647. No collisions were recorded involving lesser or great black-backed gulls, reflecting avoidance rates of 1.000 for these species.

Assessment of methodology

The fatality searches appear to have been robust, with corpse correction factors applied to account for loss of corpses to scavengers. However, during vantage point surveys, no corrections were applied to account for imperfect detection. As a result, bird activity within the area was likely to be underestimated, and therefore, the final, derived avoidance rates were also likely to be underestimated.

As collision and bird activity data were collected concurrently over the same area, these data were included when deriving representative avoidance rates.

A6.16 Kleine Pathoweg

Everaert, J. 2008. *Effecten van windturbines op de fauna in Vlaanderen: Onderzoeksresultaten, discussie en aanbevelingen*. INBO, Brussels

Everaert, J. & Kuikjen, E. 2007. *Wind turbines and birds in Flanders (Belgium): Preliminary summary of the mortality research results.*

Methods

Throughout 2005 and 2006, an area covered by a 100 m radius around the base of each turbine was searched for collision victims once every 2 weeks. Correction factors were applied to the resultant data to account for searcher efficiency and the removal of corpses by scavengers.

Between September and December 2005, bird activity data were collected between turbines 3 and 7. Data were collected from 2 hours before sunrise to 4 hours after sunset and presented as an average number of birds/day – reflecting an average of 16 hours of survey effort over this period.

Seasons / time of day

Bird activity data were collected between September and December, from 2 hours before sunrise to 4 hours after sunset.

Species

Black-headed gulls (345 birds/day), 'large' gulls (327 birds/day).

Conditions data collected under

Not stated.

Location / habitat

Kleine Pathoweg (Belgium), terrestrial.

Turbine / array specification

A line of 7 turbines, each separated by 280 m. Turbines had a hub height of 85 m and a rotor diameter of 70 m.

Results

In 2005, 240.9 gulls were believed to have collided with turbines once corrections had accounted for imperfect corpse detection. In 2006, this figure was 220.3. Based on a passage rate of 42 birds per hour, in 2005 these figures reflect an avoidance rate of 0.8795 using option 1 of the Band model, -0.2529 using option 2 of the Band model and -0.6887 using option 3 of the Band model. In 2006, these figures reflect an avoidance rate of 0.8898 using option 1 of the Band model, -0.1458 using option 2 of the Band model and -0.5443 using option 3 of the Band model.

Assessment of methodology

Fatality data have been collected on a regular basis and following a robust methodology. Corrections have been applied to these data to account for the imperfect detection of corpses due to scavenger behaviour and searcher efficiency.

The observational data that have been collected are extremely limited. Data collection has been restricted to the September to December period in a single year. It is unclear how accurately this reflects bird movements within the windfarm over the rest of the study period. This may have a significant, but unquantifiable impact on the final, derived within-windfarm avoidance rates. In addition, it is unclear whether corrections have been applied to the observational data to account for the imperfect detection of birds.

As it has been necessary to make spatial and temporal extrapolations to estimate avoidance rates, these data have not been used when deriving representative avoidance rates.

A6.17 Oosterbium

Methods

Visual observations and fatality searches.

Searches were carried out within a 50 m radius of the base of each turbine in autumn 1990 and spring 1991. Searches were carried out on 25 days in the spring and 40 days during autumn. All corpses were assessed in order to determine the cause of death and identify those killed by turbines. Corrections were applied to the data to account for searcher efficiency and scavenger activity.

Bird activity within the windfarm and a surrounding 500 m buffer was assessed during spring 1991 and autumn 1990. These activity levels were used to extrapolate the number of bird-days spent within the windfarm for each species or group of species.

Seasons / time of day

Data covered both the nocturnal and diurnal movements of birds in the spring and autumn.

Species

Gulls (158,600 bird days, autumn 1990; 43,800 bird days, spring 1991).

Conditions data collected under

No details given.

Location / habitat

Oosterbierum, Netherlands (terrestrial)

Turbine / array specification

A cluster of 18 turbines with hub heights of 35 m and a rotor diameter of 30 m, situated within 55 hectares of farmland. Distance between turbines within each row is not described.

Results

Gulls were recorded within the area more commonly during the autumn than the spring. However, the number of collisions was greatest during the spring, when 37 corpses were recovered in comparison to 12 in the autumn. No site specific flight height data were available so it was necessary to use the modelled distributions presented in Johnston *et al.* (2014a) and option 2 of the Band model. During the autumn, the predicted number of collisions in the absence of avoidance was 883 birds. Therefore, the 12 collisions recorded during the autumn reflects a meso-micro avoidance rate of 0.9864. Using option 3, the predicted number of collisions was 846, reflecting a meso-micro avoidance rate of 0.9858. During the spring, the predicted number of collisions in the absence of avoidance was 244 using option 2 and 234 using option 3. Therefore, the 37 collisions recorded during the spring reflects a meso-micro avoidance rates of 0.8483 and 0.8417 respectively.

Assessment of methodology

Fatality searches were carried out intensively throughout the spring and autumn seasons. They followed a robust methodology with corrections made for both searcher efficiency and scavenger activity.

Activity data were collected throughout the period covered by the fatality searches. However, it appears no corrections were made to the data to account for imperfect detection, meaning activity levels in the area may have been underestimated. As a consequence, the number of collisions predicted in the absence of avoidance, and therefore the derived avoidance rate would also have been underestimated.

As activity and mortality data were collected concurrently and no spatial extrapolation was necessary, these data were used when deriving representative avoidance rates.

A6.18 Waterkapptocht

Krijgsveld, K.L., Akershoek, K., Schenk, F., Dijk, F. & Dirksen, S. 2009. Collision risk of birds with modern large wind turbines. *Ardea* **97,** 357-366.

Methods

Radar observations and fatality searches.

Fatality searches were carried out within a 100 m radius around each turbine every 2-3 days. Searches were carried out by walking parallel transects, each separated by 4-6 m. Searches were carried out between October and December 2004.

Flight movements were quantified using a 12 kW x-band marine surveillance radar overnight between 1800 and 0700 h on 18 October 2004, 17 November 2004 and 20 December 2004, and the number of radar echoes up to 140 m (the maximum turbine height) were estimated as a measure of flux through the windfarm area.

Seasons / time of day

Resultant data reflect overnight collision rates of birds between October and December 2004.

Species

Key movements recorded included gulls travelling between Lake Ijsselmeer and a nearby roost site around dusk and dawn. However, amongst the seven corpses encountered, there was only a single gull carcass, that of a black-headed gull.

Conditions data collected under

Not specified.

Location / habitat

Agricultural area in the Netherlands.

Turbine / array specification

The array consists of a single line of eight turbines, each separated by 300 m, with a larger 1 km gap between turbines 4 and 5. Turbines have a hub height of 78 m and a rotor diameter of 66 m. However, only the areas under five turbines were searched for carcasses.

Results

The average flux of birds through the area was 251 birds/km/hr, reflecting a movement of 1,195,011 birds through the study period as a whole. In total, the remains of seven birds (one common pheasant, one oystercatcher, one black-headed gull, one skylark and two goldcrests) were retrieved. No site specific flight height data were available, so it was necessary to use the modelled distributions presented in Johnston *et al.* (2014a) and option 2 of the Band model. Given that it was not possible to relate the radar tracks to individual species, we calculated the probability of collision based on a bird with the characteristics of a black-headed gull, giving a predicted collision rate of 1,446 birds over the study period, and an overall avoidance rate of 0.9952. Using option 3, the predicted number of collisions was 1,118 birds, reflecting an overall avoidance rate of 0.9937.

Assessment of methodology

The total collision rate may have been an underestimate as the initial searching rate of once every three days was lowered to once every two days following the outcome of depredation tests. However, all corpses present were likely to be discovered as only turbines where the surrounding vegetation was low were searched for remains. With the exception of concerns over the depredation rate, the fatality searches were robust.

Flux rates were estimated using x-band radar, with the considerable disadvantage that it cannot be used to estimate the flux rates of different species. As a consequence, using individual species collision rates to estimate an avoidance rate may lead to an inaccurate estimate of the true value. In addition, as a single radar echo may represent multiple birds, there was a considerable risk that the true movement of birds through the area was underestimated and that therefore the overall avoidance rate has also been underestimated.

As it was necessary to make temporal and spatial extrapolations with these data, they were not used to derive representative avoidance rates.

A6.19 Yttre Stengrund/Utgrunden Offshore Windfarm

Petterson, J. 2005. *The impact of Offshore Wind farms on Bird Life in Southern Kalmar Sound, Sweden: A final report based on studies 1999-2003*. Lund University.

Methods

Visual observations

Field data were collected from three observation points located within the Southern Kalmar Sound – Eckelsudde in Oland in the east of the observation area, Olsang in the west of the observation area and Utgrunden Lighthouse in the centre of the Sound of Kalmar. The observation points made it possible to cover the whole of the Sound of Kalmar, including both windfarm sites. The sound was divided into four 5 km zones, each of which was further subdivided into 1-2 km wide zones. The observation point at Olsang covered the first of these 5 km zones, the Utgrunden Lighthouse covered the second and third 5 km zones and the Eckelsudde observation point, the fourth. Observers recorded to the exact minute the location of all flocks of migrating waterbirds they encountered, so that data could be combined into a single dataset at a later date.

Seasons / time of day

Data were collected throughout the spring (22 March to 8 April) and autumn (6 to 28 October) migration periods between 2001 and 2003.

Species

Conditions data collected under

All conditions.

Location / habitat

Southern Kalmar Sound, Sweden (offshore).

Turbine / array specification

Five 2 MW turbines with a hub height of 60 m and a rotor diameter of 72 m at Yttre Stengrund.

Seven 1.5 MW turbines with a hub height of 65 m and a rotor diameter of 70 m at Utgrunden. Distance between turbines within each row is not described.

Results

No collisions were recorded amongst any species during the spring migration periods, reflecting an avoidance rate of 1. No site specific flight height data were available at this site, so it was necessary to use the modelled distributions presented in Johnston *et al.* (2004). A single collision event was recorded involving four common eider during autumn 2003, reflecting an avoidance rate of 0.1861 using option 2 of the Band model and -0.1098 using option 3. No other collisions were recorded amongst other species, again indicating an avoidance rate of 1.

Assessment of methodology

Methodology is sound with careful calibration of estimates of distance between observers and co-ordination of counts to minimise double-counting. However, there was no correction applied to account for imperfect detection, meaning the total number of birds may have been under-estimated.

As insufficient data have been collected to detect a collision amongst any of the priority species, these data have not been used to derive representative avoidance rates.

A6.20 Zeebrugge

Everaert, J. 2008. *Effecten van windturbines op de fauna in Vlaanderen: Onderzoeksresultaten, discussie en aanbevelingen*. INBO, Brussels

Everaert, J. & Kuikjen, E. 2007. *Wind turbines and birds in Flanders (Belgium): Preliminary summary of the mortality research results.* INBO, Brussels

Everaert, J. & Stienen, E.W.M. 2007. Impact of wind turbines on birds in Zeebrugge (Belgium) Significant effect on breeding tern colony due to collisions. *Biodiversity and Conservation*, 16, 3345-3359

Everaert, J., Devos, K. & Kuijken, E. 2002. *Windturbines en vogels in Vlaanderen: Voorlopige onderzoeksresultaten en buitenlandse beviningen*. INBO, Brussels

Methods

Visual observations and fatality searches.

Between 2001 and 2007 systematic fatality searches were carried out within a 50 m radius around the base of turbines on a fortnightly basis, increasing to 3-4 times a week during the breeding season. Every turbine was searched, and corrections were made to account for searcher efficiency and scavenger activity.

An initial set of bird activity surveys were carried out at the site in 2000 and 2001. Bird activity within a 400 m section of the breakwater was monitored on four days between June and July in 2000 and 2001, with eight days data collected in total. An additional four days of monitoring were carried out on four days and two nights between September and October 2001.

In June 2004 and 2005, a second set of bird activity were carried out. In each year, two full days of monitoring data were collected covering the period from dawn to dusk. During this period, data were collected between turbines 7 and 12, covering a 720 m section of the breakwater.

Seasons / time of day

Fatality searches were carried out throughout the year. Activity surveys were limited to the breeding season and autumn. Data were collected throughout the day between dawn and dusk, with additional nocturnal surveys carried out during the autumn.

Species

Gulls (234 birds/day), little tern (375-1,860 birds/day), common tern (4,228-10,263 birds/day), Sandwich tern (11-12,334 birds/day).

Conditions data collected under

No details given.

Location / habitat

Zeebrugge, Belgium (Coastal)

Turbine / array specification

25 turbines arranged along Zeebrugge Harbour breakwater. Turbines vary in size from hub heights of 23-55 m and rotor diameters of 22-48 m. Details of collisions at individual turbines are not given, so avoidance rates are estimated assuming turbines with a hub height of 34 m and rotor diameter of 34 m, the most common turbine within the windfarm. Distance between turbines within each row is not described.

Results

Collisions were recorded in every year. For Sandwich terns, collisions varied from seven to 54 birds per year. Using option 1 of the Band model, the estimated number of collisions per year, in the absence of avoidance behaviour, varied from 6,383 birds to 10,299, 8,024 to 10,326 using option 2 and 5,984 to 8,035 using option 3. The meso-micro avoidance rates derived from the values are 1 between 2001 and 2003, 0.9915 in 2004, 0.9972 in 2005, 0.9992 in 2006 and 0.9993 in 2007 using option 1, and 1 between 2001 and 2003, 0.9948 in 2004, 0.9963 in 2005, 0.9989 in 2006 and 0.9991 in 2007 using option 2. Using option 3, the avoidance rates are 1 between 2001 and 2003, 0.9933 in 2004, 0.9952 in 2005, 0.9986 in 2006 and 0.9989 in 2007. Collision data were also obtained relating to June 2004 and June 2005, the periods in which bird activity data were collected and relating to only the turbines around which activity was monitored. In both years, 3 Sandwich terns were observed to have collided between turbines 7-12 in June. Given passage rates of 896 birds/hr in June 2004 and 725 birds/hr in June 2005, this reflects an avoidance rate in 2004 of 0.9895 using option 1 of the Band model, 0.9935 using option 2 of the Band model and 0.9917 using option 3 of the Band model. In 2005, the corresponding values are 0.9940, 0.9920 and 0.9897.

For little terns, collisions varied from two to 12 birds per year. Using option 1 of the Band model, the estimated number of collisions per year, in the absence of avoidance behaviour, varied from 990 birds to 1,087, 165 to 838 using option 2 and 128 to 650 using option 3. The meso-micro avoidance rates derived from the values are 0.9923 in 2001, 0.9914 in 2002, 0.9904 in 2003, 0.9950 in 2004, 0.9982 in 2005, 0.9963 in 2006 and 0.9890 in 2007 using option 1, and 0.9516 in 2001, 0.9455 in 2002, 0.9395 in 2003, 0.9940 in 2004, 0.9884 in 2005, 0.9768 in 2006 and 0.9304 in 2007 using option 2. Using option 3, the avoidance rates were 0.9516 in 2001, 0.9455 in 2002, 0.9395 in 2003, 0.9940 in 2004, 0.9884 in 2005, 0.9768 in 2006 and 0.9304 in 2007. No little tern collisions were recorded in the June 2004 and 2005 data relating to turbines 7-12.

For common terns, collisions varied from 12 to 164 birds per year. Using option 1 of the Band model, the estimated number of collisions per year, in the absence of avoidance behaviour, varied from 4,503 birds to 6,869, 2,475 to 6,530 using option 2 and 1,931 to 5,094 using option 3. The meso-micro avoidance rates derived from the values are 0.9970 in 2001, 0.9977 in 2002, 0.9951 in 2003, 0.9758 in 2004, 0.9812 in 2005, 0.9761 in 2006 and 0.9834 in 2007 using option 1, and 0.9919 in 2001, 0.9939 in 2002, 0.9871 in 2003, 0.9833 in 2004, 0.9501 in 2005, 0.9365 in 2006 and 0.9559 in 2007 using option 2. Using option 3, meso-micro avoidance rates were 0.9896 in 2001, 0.9922 in 2002, 0.9834 in 2003, 0.9786 in 2004, 0.9360 in 2005, 0.9186 in 2006 and 0.9434 in 2007. Collision data were also obtained relating to June 2004 and June 2005, the periods in which bird activity data were collected and relating to only the turbines around which activity was monitored. In 2004 6 common terns were observed to have collided between turbines 7-12 in June, in 2005, this figure was 9. Given passage rates of 603 birds/hr in June 2004 and 248 birds/hr in June 2005, this reflects an avoidance rate in 2004 of 0. 9703 using option 1 of the Band model, 0.9796 using option 2 of the Band model and 0.9738 using option 3 of the Band model. In 2005, the corresponding values are 0.9720, 0.9255 and 0.9045.

For gulls, collisions varied from 110 to 354 birds per year. Using option 1 of the Band model, the estimated number of collisions per year, in the absence of avoidance behaviour, varied from 2,334 birds to 2,537, 2,856 to 3,104 using option 2 and 2,698 to 2,932 using option 3. The meso-micro avoidance rates derived from the values are 0.8979 in 2001, 0.8481 in 2002, 0.8817 in 2003, 0.9105 in 2004, 0.9173 in 2005, 0.9547 in 2006 and 0.9092 in 2007 using option 1, and 0.9166 in 2001, 0.8758 in 2002, 0.9033 in 2003, 0.9268 in 2004, 0.9324 in 2005, 0.9630 in 2006 and 0.9258 in 2007 using option 2. Using option 3 meso-micro avoidance rates were 0.9117 in 2001, 0.8686 in 2002, 0.8976 in 2003, 0.9226 in 2004, 0.9285 in 2005, 0.9608 in 2006 and 0.9214 in 2007.

Data were also obtained relating to black-headed, lesser black-backed and herring gull collisions in June-July 2000, June-July 2001 and September-October 2001, periods corresponding to the times during which gull activity data were collected and restricted to the turbines around which gull data were collected. No collisions were reported involving black-headed gulls. In June-July 2000, a single collision was reported involving a herring gull, reflecting an avoidance rate of 0.9861 using option 1, 0.9829 using option 2 and 0.9819 using option 3. In June-July 2001 and September-October 2001, two collisions were reported involving herring gulls, reflecting avoidance rates of 0.9722 and 0.9976 respectively using option 1, 0.9659 and 0.9959 using option 2 and 0.9639 and 0.9957 using option 3. Single collisions were reported involving lesser black-backed gulls in each of June-July 2001 and September to October 2001, reflecting avoidance rates of 0.9706 and 0.9990 respectively using option 1, 0.9680 and 0.9977 using option 2 and 0.9656 and 0.9975 using option 3.

Assessment of methodology

The study at Zeebrugge offers one of the most comprehensive datasets for collisions involving marine birds. Fatality data have been collected over a seven year period following a robust methodology with corrections made to account for searcher efficiency and scavenger activity. However, a key limiting factor in the dataset is the accompanying bird activity data. In the case of terns, activity data is limited to the period of peak tern activity in June. As a consequence, extrapolating from this to cover the full period when terns are present is likely to vastly over-estimate activity in the area, and therefore the predicted collision numbers. This means that the avoidance rates derived for each year are likely to be significantly over-estimated. This reflects the limitations in the way data are presented within the reports. Ideally, collisions would be broken down on a month by month and turbine by turbine basis, so that avoidance rates could be calculated for the areas in which activity data were collected, rather than extrapolating across the windfarm as a whole.

We used only the collision data collected from gulls during the period in which activity data were collected, and from only those turbines around which activity data were collected, in deriving representative avoidance rates.

ANNEX 1

USING A COLLISION RISK MODEL TO ASSESS BIRD COLLISION RISKS FOR OFFSHORE WINDFARMS
(SOSS Guidance: March 2012)

SUPPLEMENT – AVOIDANCE RATES USING THE BASIC AND EXTENDED MODELS
March 2014 – Bill Band

This is a supplement to guidance prepared for the Crown Estate as part of the Strategic Ornithological Support Services programme, project SOSS-02[45]. That provides guidance for offshore wind developers, and their ecological consultants, on using a collision risk model to assess the bird collision risks presented by offshore windfarms. The March 2012 version of the guidance enabled use to be made of flight height distribution data.

This supplement is an addition to Stage E – Avoidance and Attraction. That section describes how, having used the collision model to calculate the potential collision rate if birds take no avoiding action, one should then apply an avoidance rate A to allow for the fact that many species of birds do in fact take avoiding action, either at long range (macro) or at close range (micro).

Paragraph 80 notes that

'if the extended model taking account of flight height distribution is used, it is important that the calculations on which avoidance rates are based also start with a no-avoidance collision rate derived using the extended model'.

Most of the published literature on avoidance rates is currently based on using the basic model. **This supplement shows how such avoidance rates may be modified to enable their application to the extended model.**

Avoidance in the basic and extended models

The two models – basic and extended – yield different predictions of the rate of collisions before avoidance is taken into account. The extended model is a more refined model which takes into account the effect of flight height distribution. It takes into account the fact that, for a given number of flights at risk height, a flight height distribution skewed towards low altitude leads to a smaller proportion of birds passing through the rotor, and bird passages through parts of the rotor with less risk, than if the distribution were uniform.

The outputs of the two models may be formally compared if the data input to the basic model on the proportion of flights at risk height (Q'_{2R}) is derived from the same

[45] Project SOSS -02: see http://www.bto.org/science/wetland-and-marine/soss/projects

flight height distribution used in the extended model, as in Option 2 of the spreadsheet accompanying the SOSS guidance. That is, the comparison should be made between the collision rate using the basic model (Option 2) in the spreadsheet, and the extended model (Option 3).

The collision rates (before avoidance) projected by the two models are:

Basic model (Option 2):

$C_{basic} = v(D_A/2R)(T\pi R^2)t \times Q_{2R}' \times p_{average} \times Q_{op}$

(guidance eq.5[46])

i.e. flux factor $\times Q_{2R}' \times p_{average} \times Q_{op}$

Extended model (Option 3):

$C_{extended} = v(D_A/2R)(T\pi R^2)t \times (2/\pi) \iint d(y)\, p(x,y)\, dxdy \times Q_{op}$

(guidance eq. 9)

i.e. flux factor x collision integral x Q_{op}

Where the bird flight height distribution is skewed towards low altitude, the extended model prediction $C_{extended}$ is usually less than C_{basic}, because this equation takes full account of the reduction in risk at lower parts of the rotor. Let g be the ratio $C_{extended} / C_{basic}$, g is thus usually less than 1. The value of g may be obtained by dividing the second of the above equations by the first:

$g = C_{extended} / C_{basic}$ = collision integral / ($Q'_{2R} \times p_{average}$) …. eq. S1

and this is readily calculated from the 'Overall collision risk' spreadsheet

g = cell D35 / (cell D33 x cell D27)

The expected collision rate must then take into account the proportion A of birds avoiding the turbines (e.g. by displacement, or by evasive action), by multiplying the above no-avoidance collision rates by the proportion (1-A) which do not avoid. Values of A are typically in the range 90-100%. It is more helpful to think in terms of the non-avoidance rate A' = 1 – A , such that A' is the small proportion of birds which do not avoid the turbines. The expected collision rate is then

$A'_{basic} C_{basic}$ in the basic model, or .. eq. S2a

$A'_{extended} C_{extended}$ in the extended model. .. eq. S2b

The two models require the use of different non-avoidance rates. The calculation of $C_{extended}$ takes account of the effect of a skewed flight distribution, such that the

[46] Strictly, equation (5) of the guidance refers to Q_{2R} derived from site survey, as used in the basic model (Option 1), rather than Q'_{2R}, derived from the assumed flight height distribution, as required here.

factor $A_{extended}$ (= 1 – $A'_{extended}$) refers only to genuine behavioural avoidance. The calculation of C_{basic} in the basic model does not, such that any such effect, in the basic model, must be covered by the avoidance factor A_{basic}.

Establishing avoidance rates from reference windfarms

Values of A'_{basic} and $A'_{extended}$ for use in the two models are obtained by monitoring collisions at one or more reference windfarms, and working back from the two models. For either model we have

Non-avoidance rate A' = Actual no of collisions / Predicted number of collisions C.

	using basic model	*using extended model*
Actual no of collisions	= A'_{basic} x C_{basic}(ref)	= $A'_{extended}$ x $C_{extended}$(ref)

thus $A'_{extended} = A'_{basic} \times C_{basic}(ref) / C_{extended}(ref)$

but $g(ref) = C_{extended}(ref) / C_{basic}(ref)$

so $\mathbf{A'_{extended} = A'_{basic} / g(ref)}$ … eq. S3

$A'_{extended}$ is the non-avoidance rate from the reference windfarm, for use with the extended model. Equation (S3) describes how it is related to the value of A'_{basic} derived using the basic model, using the g factor for this reference windfarm.

Where data from several reference windfarms are used to yield an average A'_{basic}, then the value for $A'_{extended}$ should be the average of A'_{basic} / g(ref) as calculated for each of the reference windfarms.

Applying reference avoidance rates to new or projected windfarms

Avoidance rates, derived from collision studies at one or more reference windfarms, may be used to inform the calculation of collision rate at a new or projected windfarm. The assumption in applying such avoidance rates is that the birds' behavioural response to the new windfarm will be similar to their response to the reference windfarm, and hence the proportion of birds avoiding the turbines of the new windfarm, further to the calculation of a no-avoidance collision rate, is likely to be the same as for the turbines of the reference windfarm.

Thus, having established values A'_{basic} and $A'_{extended}$ for non-avoidance, as derived from the reference windfarm, these same values may be assumed to apply to new or projected windfarms for the same bird species. If C_{basic}(new) and $C_{extended}$ (new) are the no-avoidance collision rates calculated for the new windfarm, the predicted collisions after avoidance for the new windfarm are:

basic model: $\mathbf{A'_{basic}\ C_{basic}\ (new)}$ .. eq. S4a

extended model: $\mathbf{A'_{extended}\ C_{extended}\ (new)}$ .. eq. S4b

A'_{basic} is the avoidance rate established from the reference windfarm(s) using the basic model, and $A'_{extended}$ that using the extended model; they are related as in equation (S3).

Dealing with lack of information on g(ref)

Published information on avoidance rates for reference windfarms has not so far included information on avoidance using the extended model, or on g(ref), the ratio between the outputs (before avoidance) of the extended and basic models. Calculation of g(ref) requires information on bird size and speed, turbine parameters, and the flight height distribution at the reference site; however it does not need information on bird density, levels of flight activity, or number of transits. If this limited subset of data is available, then it should be possible to calculate g(ref) for the reference windfarm, for the bird species under assessment.

Example:

Monitoring studies have established that for a certain bird species, an overall avoidance rate of 98% may be assumed. This has been derived using theoretical collision rates derived using the basic model, and comparing these with the actual collision mortality observed on an existing windfarm – the 'reference' windfarm.

$\mathbf{A_{basic} = 98\%}$ so the non-avoidance rate $A'_{basic} = 2\%$.

Using the extended model, the calculated g factor for this reference windfarm is 0.46. Thus the non-avoidance rate appropriate for use with the extended model is

$A'_{extended} = 2\%/0.46 = \mathbf{4.38\%}$

The corresponding avoidance rate for use with the extended model is

$\mathbf{1 - A' = 95.62\%}$

A developer now undertakes collision risk assessment for a proposed offshore windfarm. The CRM extended model which takes account of flight height distributions may be used, provided that it makes use of the avoidance rate appropriate for the extended model.

For the proposed windfarm, the projected collision rates are 23 (basic model) and 8 (extended model) per year. Applying the above non-avoidance rates of 2% and 4.38% respectively yields

expected collisions(basic) = 2% x 23 = 0.46 birds/annum

expected collisions (extended) = 4.38% x 8 =0.35 birds/annum

The two models yield different results because the second model takes flight height distribution into account, a factor ignored in the basic model.

It is recommended that any future publication of reference avoidance rates, derived from collision monitoring studies, should state both that for use in the basic model and that for use in the extended model. This will require application of both models to the reference windfarm.

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