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The Scientific Validity of Criticisms made by the RSPB of Metrics used to Assess Population Level Impacts of Offshore Wind Farms on Seabirds

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Natural Resources Wales and Department of Environment, Northern Ireland

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1. INTRODUCTION

The purpose of this report is to consider the validity of criticisms that have been made by the RSPB on the use of risk-based metrics to assess the response of seabird populations to impacts from offshore wind farms. Following the project start-up meeting on 3rd December 2014, it was agreed that this consideration should also include approaches that have been used to set thresholds of additional mortality including Potential Biological Removal (PBR), Acceptable Biological Change (ABC) and reduced uncertainty Acceptable Biological Change (ruABC).

A variety of metrics and methodologies have been used to assess the population level impacts of effects associated with proposed offshore wind farms (Table 1). With respect to the recent assessment of potential impacts of offshore wind farm projects on populations of seabirds in Scotland, the RSPB has made three key criticisms of the approaches that have been used (Green 2014):

1. ***Procedures for calculating effects of wind farms on seabird per capita mortality rates and breeding success as a result of collision, displacement and barrier effects do not have a firm foundation in empirical data, so scientifically robust, and therefore defensible, collision rate values and confidence intervals for collision mortalities and displacement and barrier effects cannot be calculated.***
2. ***As a consequence of (1), no probabilistic methods for assessing risk of population impacts of a given type and magnitude are scientifically robust and therefore defensible. This applies to both the Acceptable Biological Change (ABC) method, reduced uncertainty Acceptable Biological Change (ruABC) as well as other proposed methods to estimate the difference in the probability of a specified population outcome (e.g. a population decline) between impacted and unimpacted scenarios.***
3. ***Attempts to identify thresholds either side of which estimates of population-level impacts of developments are considered negligible (or non-negligible) on biological grounds are mistaken and should be abandoned.***

In addition to these general criticisms, Green (2014) makes more specific criticisms of some of the approaches used, notably PBR and ABC. In their written[1] and oral[2] responses to the Hornsea Project One Offshore Wind Farm, the RSPB indicate further criticisms of the approaches that have been used to date. The RSPB suggest that the most appropriate approach is to use PBR in an initial screening process to determine which projects will clearly have an unacceptable impact on populations. For the remaining projects, they suggest that a density-independent Leslie matrix model should be used to estimate the population size at the end of the lifetime of the project with and without the demographic impacts of the wind farm using matched simulations (Green 2014).

The British Trust for Ornithology (BTO) is an independent research organisation that provides data and analysis to inform decisions impacting on biodiversity and the environment. Our approach is strictly impartial and based on rigorous science. We have contributed to the development of guidance for the offshore wind industry through projects under the Collaborative Offshore Wind Research into the Environment (COWRIE) programme, operated the Strategic Ornithological Support

[1] http://infrastructure.planningportal.gov.uk/wp-content/ipc/uploads/projects/EN010033/2.%20Post-Submission/Representations/Comments/Other%20Comments/Royal%20Society%20for%20the%20Protection%20of%20Birds.pdf

[2] http://infrastructure.planningportal.gov.uk/wp-content/ipc/uploads/projects/EN010033/2.%20Post-Submission/Hearings/Issue%20Specific%20Hearing%20-%2029-04-2014%20-%200930%20-%20Humber%20Royal,%20Grimsby/Royal%20Society%20for%20the%20Protection%20of%20Birds.pdf

Services (SOSS) Secretariat on behalf of The Crown Estate for the period 2010-12, and provided substantial input to published reviews on seabird flight heights (Cook *et al.* 2012, Johnston *et al.* 2014), avoidance rates (Cook *et al.* 2014), and post-consent monitoring (MMO 2014). Building on this expertise and impartial approach, the BTO was invited to comment on the points raised in the RSPB report, "*Misleading use of Science in the assessment of probable effects of offshore wind projects on populations of seabirds in Scotland*" (Green 2014), and the RSPB written[1] and oral[2] representations in relation to the proposed Hornsea Project One offshore wind farm. We first identify the range of different approaches that have been taken in the assessment of offshore wind farms on seabird populations before, second, considering both the general and specific criticisms made by the RSPB of the methodologies used to assess the population level impacts of offshore wind farms on seabirds. The purpose of this report is, specifically, to consider the criticism of metrics that are used to assess population level impacts of offshore wind farms on seabirds, put forward in Green (2012) and in relation to the Hornsea Project One offshore wind farm[1,2], and how these metrics may be used in the future. We do not critique the use of these approaches in past assessments, nor analyse the casework undertaken in relation to any particular offshore wind farm applications.

As part of BTO's Quality Assurance procedures, this report has been internally reviewed by the BTO Director of Science, Dr James Pearce-Higgins, and by the Head of the Wetland and Marine Research Team, Dr Niall Burton. The report has also been commented on by the project steering group consisting of representatives from the Joint Nature Conservation Committee, Natural England, Scottish Natural Heritage, Marine Scotland Science, Natural Resources Wales and Department of Environment, Northern Ireland. In order to maintain the impartiality of this work, the BTO were under no obligation to address any of the comments raised by the project steering group, however, we incorporated those comments that improved the clarity of writing or corrected factual inaccuracies. Throughout this report, references to "we" solely reflect the views of the BTO, and not the positions of the project steering group. We consider each of the general, then specific, criticisms raised in turn, and, to aid the reader, we provide a brief summary of the key points for each in *italics*.

2. METRICS USED TO ASSESS POPULATION LEVEL IMPACTS OF OFFSHORE WIND FARMS ON SEABIRDS

There is widespread concern that offshore wind farm developments may impact on seabird populations (*e.g.* Drewitt & Langston 2006; Everaert & Stienen 2007). There is, therefore, much effort being invested in understanding the potential effects (i.e. collision, displacement and barrier effects) of such developments and their impacts on seabirds at a population level, with a range of predictive approaches being employed in an attempt to quantify these future impacts.

Having considered projects currently listed on the infrastructure planning portal website[3], the Marine Scotland Licensing Portal[4] and other recent, high profile examples, we identified 27 proposed sites at which the population level impacts of offshore wind farms on seabirds had been considered during assessment: Aberdeen Offshore Wind Farm, Beatrice, Burbo Bank Extension, Docking Shoal, Dogger Bank Creyke Beck A, Dogger Bank Creyke Beck B, Dogger Bank Teesside A, Dogger Bank Teesside B, Dudgeon, East Anglia One, Fife Wind Energy Park, Galloper, Hornsea Project One, Inch Cape, London Array Phase II, MacColl, Navitus Bay, Neart na Gaoithe, Race Bank, Rampion, Seagreen Alpha, Seagreen Bravo, Stevenson, Telford, Triton Knoll 3, Walney I & Walney Extension. Across these 27 sites, we identified 12 metrics, many of which are necessarily inter-related, that have been used to estimate population level effects of offshore wind farms (Table 1). **For the purposes of this report, we define a metric as any value or rule upon which a decision about whether or not a population level effect associated with the impacts of an offshore wind farm is deemed acceptable.**

Table 1 Description of metrics used to estimate population level impacts of proposed offshore developments.

Potential Biological Removal (PBR)

PBR was initially developed for use in marine mammal populations (Wade 1998). PBR provides an estimate of the maximum level of mortality, in addition to that expected to occur naturally, which a population can experience and still remain viable. A key advantage of PBR is that it requires very little input data, only the minimum current population size, mean age at first breeding and mean adult survival (Niel & Lebreton 2005). In addition, it is relatively simple to calculate following the methodology set out by Dillingham & Fletcher (2008). This simplicity makes PBR an extremely attractive approach with which to assess the potential population level impacts of offshore wind farms on seabirds.

Population growth rate

The population growth rate measures the extent to which the size of the breeding population changes on an annual basis. By considering the growth rate of the population in the presence of an offshore wind farm, it should be possible to consider whether the population will remain stable (growth rate=1), increase (growth rate>1) or decrease (growth rate<1) through the life time of the project.

Probability that growth rate <1

As part of the SOSS programme, guidance was produced for using Population Viability Analysis (PVA) to assess the potential impacts of collision-related mortality associated with offshore wind farms (WWT Consulting 2012). Under a PVA approach, stochastic models are used to simulate the impact of additional mortality on populations of species of interest and the proportion of

[3] http://infrastructure.planningportal.gov.uk/

[4] http://www.scotland.gov.uk/Topics/marine/Licensing/marine/scoping

simulations where the population declines (i.e. growth rate <1) calculated.

Probability that population decreases below initial size

The impact of a development is typically assessed in relation to a baseline population size, which may be either the pre-construction population size, the population size of a protected site at designation, the population size from Seabird 2000 (Mitchell *et al.* 2004), or some other appropriate value. Using stochastic models, the proportion of simulations in which the population drops below this baseline, either at any point in the lifetime of the project or by the end of the project, could be assessed. Alternative baseline populations, for example, the size of the population at designation in the case of a breeding colony at a protected site, could be used. Mathematically, this metric is nearly identical to the previous metric.

Probability of a population being a given magnitude below the median size predicted in the absence of an impact

With the simulations from stochastic models, rather than looking at the probability or magnitude of a decline, it may be more meaningful to estimate the median population size estimated across all simulations. This could be done either for a single fixed point in time, or at given intervals. A metric to assess the population level impact of a development could be derived by estimating a median size for a population in the absence of it and then calculating the proportion of simulations for a population in the presence of the development that are (a given magnitude) below this median population size.

Ratio of median impacted to unimpacted growth rate

Considering the growth rate of a population only in the presence of an offshore development enables an assessment of whether the population will remain stable, increase or decrease over time, but it does not make it possible to quantify the impact of the development on that growth rate. By comparing the growth rate of the population in the presence of a development to that expected in its absence it is principle possible to quantify what annual impact the development is having on a population.

Ratio of impacted to unimpacted population size

Population models can be used to estimate the size of a population through time both with and without the impact of an offshore development. Comparing the ratio of the size of these two populations offers a relatively easy to interpret statistic with which to assess the population level impact. The ratio could be derived either from a simple deterministic model, or taken from the mean or median values simulated using a more complex stochastic model, with or without density-dependence. The ratio of population sizes could be estimated either at a fixed point in time, e.g. the end of a project, or at a series of intervals throughout the life time of a project.

Change in probability that growth rate <1

Where simulations show that a population may already be at risk of declining in the absence of a development, for example if more than 50% of simulations have a growth rate <1, simply quantifying the probability of a population decline in the presence of an offshore development may not be meaningful. To assess the population level impact it would be necessary to determine how much greater the probability of a decline is in the presence of the development than in its absence. This could be done either at a single fixed point in time, or at intervals throughout the life time of the project.

Change in probability of a population decreasing by a given magnitude

At many colonies throughout the UK, seabird populations are already declining (JNCC 2014). As a consequence, the presence of a development is unlikely to increase the probability of the growth rate at these colonies being <1, especially if all the simulations from the baseline scenario already have a growth rate <1. However, the presence of the development may cause a further reduction in the magnitude of growth rate. It may, therefore, be more meaningful to consider the change in probability of a population decreasing by a given (though almost certainly artificial) threshold, e.g. a 10% increase in the probability of a 5% decline.

Probability of growth rate being x% less than unimpacted growth rate

With growth rates simulated from stochastic models, it may be desirable to estimate a mean or median value for the unimpacted population and calculate the proportion of simulations in which the growth rate of the impacted population is lower, or a given percentage lower, than this value. This approach has the advantage of allowing a probabilistic forecast of the impact of the offshore development on a population, e.g. there is a 50% chance that the development will reduce the population growth rate by 10%.

Acceptable Biological Change (ABC)

ABC was set out as a method for assessing the population impact of an offshore wind farm by Bennet (2013). Using terminology from the Intergovernmental Panel on Climate Change (IPCC), ABC attempts to set out an acceptable risk to a population. ABC allows for a change of up to one-third in the probability of a defined management target being achieved as a result of the impact of an offshore wind farm. For example, a management objective could be that the population size at the end of the life time of the project should be that which is more likely than not (i.e. in IPCC terminology has a probability of 0.667 or more) to occur in the absence of the project. If the impacted population size is greater than that which is expected to occur with a probability of 0.667 in its absence, the impact could be deemed acceptable. The approach allows also alternative targets, e.g. in reference to the site’s conservation status, to be set.

Reduced Uncertainty Acceptable Biological Change (ruABC)

An acknowledged weakness of ABC is that when there is larger uncertainty surrounding the input (demographic) parameters, this can result in wider confidence limits surrounding the estimated population sizes, and therefore greater declines being deemed acceptable (JNCC & SNH 2014). As a result, JNCC and SNH refined the methodology to account for this discrepancy, referred to as Reduced Uncertainty Acceptable Biological Change. The approach considers the model prediction uncertainty, taken to be the difference between the population predicted with a probability of 0.5 and the population predicted with a probability of 0.667. The incorporation of additional data from the regional population can help reduce the model prediction uncertainty. The model prediction uncertainty from the regional population can then be applied to the median population size for the colony of interest to estimate an acceptable level of biological change.

3. GENERAL CRITICISMS BY THE RSPB OF APPROACHES USED TO DATE

3.1. Procedures for calculating effects of wind farms on seabird per capita mortality rates and breeding success as a result of collision, displacement and barrier effects do not have a firm foundation in empirical data, so scientifically robust and therefore defensible collision rate values and confidence intervals for collision mortalities and displacement and barrier effects cannot be calculated – p3 of Green (2014)

Whilst this criticism is not directly related to the metrics used to assess population level effects of offshore wind farms on seabirds, it does relate to the way in which predicted impacts are incorporated into models used to assess population level effects. There is significant uncertainty associated with the assessment of offshore wind farms on seabird populations (Stewart *et al.* 2007, Masden *et al.* 2014). This uncertainty has been the subject of much discussion for some time (Fox *et al.* 2006, Drewitt & Langston 2006, Chamberlain *et al.* 2006) and has led to significant work programmes, including the COWRIE and SOSS programmes, and other associated research-based advice. This uncertainty has led to considerable debate about what constitutes a precautionary approach when estimating the impact of an offshore wind farm.

As more offshore wind farms become operational, increasing amounts of data are also now being collected in order to address these uncertainties. Whilst inconsistencies in data collection methodologies can make interpreting these data challenging (MMO 2014), they do reflect a growing evidence base from which to draw conclusions about the potential impacts of the effects associated with offshore wind farms on seabird populations.

Key to understanding the likely population level impacts of offshore wind farms on seabirds is an understanding of how many birds a development may affect. This requires a detailed understanding of how many birds are present in an area prior to construction, and therefore at risk from collision, barrier effects or displacement. As a consequence, there is a requirement to carry out detailed surveys in order to estimate the population size of birds within the proposed development area (Camphuysen *et al.* 2004). These data may be collected using boat and/or aerial surveys and sophisticated modelling approaches are being developed in order to generate robust population estimates from these data (e.g. Johnston *et al.* 2015). However, there remains uncertainty still over the breeding populations from which these birds originate, making an assessment of population-level impacts more challenging.

Using radar to study the movement of birds in and around wind farms has enabled some quantification of barrier effects (e.g. Desholm & Kahlert 2005, Masden *et al.* 2009, Plonczkier & Simms 2012). However, these studies have so far focussed on migrant waterbirds (geese and ducks) rather than seabirds foraging in and around the areas developed as offshore wind farms.

Several recent studies have attempted to quantify displacement rates of seabirds in relation to offshore wind farms. These studies have revealed that displacement may be highly species-specific with a range of responses recorded covering total avoidance, attraction and no response (Petersen *et al.* 2006, Leopold *et al.* 2011, Natural Power 2014, Vanermen *et al.* 2013, 2014). Whilst estimates of both displacement rates and barrier effects are available, interpreting this evidence has been hampered by inconsistencies and methodological issues in post-consent monitoring programmes (MMO 2014). Despite this, the growing evidence base for displacement means that rates can be estimated for a number of species.

As seabirds are typically long-lived and able to delay breeding attempts in unfavourable conditions, it is believed that the key impacts of displacement would be on productivity and over-winter survival, rather than adult survival in the breeding season (Furness 2013, Searle *et al.* 2014). Recent work by Searle *et al.* (2014) has modelled the impact of displacement and barrier effects on seabird populations. The study considered the population level consequences of 66 scenarios linked to different levels of displacement and barrier effects, prey availability and distribution on five species of seabird from four different protected sites. These analyses quantified the potential population level consequences of a range of displacement and barrier effects scenarios on seabird populations and showed the potential for decreases in the survival and productivity rates for all species linked to the amount of time spent foraging within the wind farm zones.

The potential impacts on seabird populations of the mortality associated with collision risk have been a key focus for concern (Fox *et al.* 2006, Drewitt & Langston 2006). Whilst collision risk models have been found to be mathematically sound (Chamberlain *et al.* 2005), their outputs are highly sensitive to input parameters including the avoidance behaviour, flight heights and speeds of birds within wind farms (Chamberlain *et al.* 2006, Fox *et al.* 2006, Masden 2015). Consequently, concern has been raised about a lack of knowledge of precise values for these parameters and, more recently about the uncertainty associated with them (Chamberlain *et al.* 2006, Masden *et al.* 2014, Masden 2015). However, recent work has sought to derive more robust estimates of both flight height and avoidance behaviour of birds within wind farms (Cook *et al.* 2012, 2014, Johnston *et al.* 2014), including generating estimates of uncertainty around these values. In addition, ongoing work (Masden 2015) is seeking to combine these uncertainty estimates within the Band collision risk modelling framework in order to derive robust estimates of the uncertainty surrounding collision risk estimates.

It should be acknowledged that there remains significant uncertainty surrounding some of the key parameters used to estimate these impacts and that the models themselves lack empirical validation. However, we believe that the estimates of the impacts of collision and displacement on seabird populations have been made with reference to the best available evidence and utilising mathematically sound models, and are therefore defensible, ***given the data available****.*

3.2. As a consequence of (1), no probabilistic methods for assessing risk of population impacts of a given type and magnitude are scientifically robust and therefore defensible. This applies to both the Acceptable Biological Change method, reduced uncertainty Acceptable Biological Change and proposed methods to estimate the difference in the probability of a specified population – p3 of Green (2014)

As stated above (3.1), we believe that while the robustness of the methods used to assess the population level impact of offshore wind farms needs empirical validation, given the data available at present, estimates of the magnitude of these impacts are defensible. That said, as highlighted by Masden *et al.* (2014), it is important to highlight uncertainty surrounding the estimated impacts, something which, given analytical limitations, is rarely done. However, given the data collection currently underway in and around existing offshore wind farms, reviews of the data that have been collected (Cook *et al.* 2012, 2014, Furness *et al.* 2013, Johnston *et al.* 2014) enable us to estimate variability around some of the key parameters used to estimate these impacts. These data can then be used within simulation modelling frameworks currently under development (e.g. Searle *et al.* 2014, Masden 2015) in order to estimate the uncertainty surrounding predicted impacts. Where estimates of uncertainty are obtained around predicted impacts, these can be used to make a probabilistic assessment of the population level effect of these impacts.

We believe that, whilst they have not been presented to date, it is possible to generate defensible estimates of uncertainty around the impacts associated with offshore wind farms.

3.3. Attempts to identify thresholds either side of which estimates of population-level impacts of developments are considered negligible (or non-negligible) on biological grounds are mistaken and should be abandoned – p3 of Green (2014)

In assessing the impacts of an offshore wind farm (or any other development) on seabirds, impacts must be considered at three different levels: (i) Is individual fitness (i.e. survival, productivity) impacted by predicted effects, e.g. collision, displacement or barrier effects? (ii) Do these individual impacts alter the population trajectory of the species concerned? (iii) Is the population level impact acceptable when considered in the context of the economic or societal benefits of the development? These levels can be considered as a continuum with each reflecting a more significant biological impact. Article 2 of the Directive of the European Parliament and Council on the Conservation of Wild Birds (2009/147/EC, the 'Birds Directive') states that:

"*Member States shall take the requisite measures to maintain the population of the species referred to in Article 1 at a level which corresponds in particular to ecological, scientific and cultural requirements, while taking into account of economic and recreational requirements, or to adapt the population of these species to that level.*"

and Article 3 states that:

"*In the light of the requirements referred to in Article 2, Member States shall take the requisite measures to preserve, maintain or re-establish a sufficient diversity and area of habitats for all the species of birds referred to in Article 1."*

To comply with these requirements, when assessing the impact of a project on a seabird population, it is therefore necessary to determine first whether the magnitude of the impact is such that a population will not be maintained at its current size, or that attempts to restore a population to a level corresponding to ecological, scientific and cultural requirements will not be impaired. Furthermore, it should be noted that populations of seabirds (or any other taxa) rarely occur in isolation, and movement between adjacent (or even more distant) populations may be non-negligible. Thus setting of thresholds of acceptable impact may need to take into account (indirect) impacts on other, demographically connected, populations.

Thresholds against which targets can be assessed cannot be thought of as biologically meaningful unless they are based on models which accurately depict population processes at the site concerned. Without these models it is not possible to determine, with confidence, how the impacts of an offshore wind farm on survival and productivity are likely to interact with one another at individual breeding colonies or at a meta-population level. However, such models ideally require a detailed knowledge of age-specific survival, productivity, immigration and emigration rates and any relevant density-dependent processes, all at a site-specific level. In practice, these data will not all exist and the models used to assess population level impacts will be generalisations, often based on data collected at regional or national levels, rather than a site specific level. As a consequence, these models will generally not be able to indicate with any degree of certainty whether or not the predicted impacts from offshore wind farms (or other developments) will have a significant effect at a population level. This means that thresholds applied to any metrics derived from these models will be based on a qualitative assessment of the evidence presented, rather than underlying biological processes, and hence there is a risk that inappropriate conclusions may be drawn. Therefore, any such threshold should be used as guidance by decision makers, rather than being viewed as firm predictions.

We believe that, at present, limitations in the data and models used to assess population level effects of offshore wind farms mean that biologically meaningful thresholds of impact cannot be set. Without improved data and more refined models, it is generally not possible to identify a threshold that is biologically relevant to the population concerned. For this reason, any threshold set will likely be largely subjective and based on a qualitative assessment of the evidence presented and should be acknowledged as such.

4. SPECIFIC CRITICISMS OF APPROACHES USED TO DATE

4.1 Inadequate knowledge of density dependence means models incorporating density dependence are inappropriate – *Section B.3 of Annex B of Green (2014); Paragraph 3.12 of the Hornsea Project One RSPB written representation*[5]

The risk-based metrics listed in Table 1 can either be derived from density-dependent or density-independent models. Density-dependence is also implicitly assumed by PBR (Wade 1998, Niel & Lebreton 2005, Dillingham & Fletcher 2008). Green (2012) raised concerns about the incorporation of density-dependence in population models previously in relation to the London Array Offshore Wind Farm Phase II Appropriate Assessment where a density-dependent response in wintering oystercatchers *Haematopus ostralegus* was extrapolated to red-throated divers *Gavia stellata*. In relation to the Hornsea One Offshore Wind Farm, RSPB suggested a review of density dependent relationships in seabirds. Only a single study, Cury *et al.* (2011), was presented in response to this suggestion[6]. However, ongoing work has highlighted a range of studies in which density-dependent responses in seabird populations have been identified (Horswill & Robinson 2015). These relate to both survival (e.g. Breton *et al.* 2006, Coulson 2001, Milne 1974) and productivity (Andersson & Eriksson 1982, Butler & Trivelpiece 1981, Kilpi 1989) and cover a range of species, e.g. Atlantic puffin *Fratercula arctica* (Harris 1980, Breton *et al.* 2006), black-legged kittiwake *Rissa tridactyla* (Coulson 2001), great black-backed gull *Larus marinus* (Butler & Trivelpiece 1981), herring gull *L. argentatus* (Kilpi 1989) and common guillemot *Uria aalge* (Birkhead *et al.* 1977). We believe that knowledge about the range of density-dependent responses in seabird populations may be greater than has been previously assumed, however, Horswill & Robinson (2015) suggest that density-dependent effects can vary markedly between colonies in relation to local conditions, and may result in the expected impacts being exacerbated or mitigated. Focussing on a single study, even one as comprehensive as Cury *et al.* (2011), therefore risks potentially over-looking important responses.

We believe that whilst evidence concerning density-dependent responses in seabird survival and productivity rates have not been routinely presented in impact assessment work, a detailed review of the topic suggests that there may be a useful evidence base to draw upon. With careful consideration of this evidence, it may be possible to consider models incorporating density-dependence when assessing the population-level impacts of offshore wind farms. However, in many cases, density-independent models are likely to represent a more precautionary approach where there is uncertainty about the shape or magnitude of any response, as they do not assume a compensatory increase in survival or productivity at low population sizes.

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[5] ***Summary written representation for the Royal Society for the Protection of Birds in the matter of planning application for the Hornsea Project One Offshore Wind Farm (Zone 4) located approximately 103km off the East Riding of Yorkshire Coast***

[6] http://infrastructure.planningportal.gov.uk/wp-content/ipc/uploads/projects/EN010033/2.%20Post-Submission/Representations/ExA%20Questions/Appendix%20X%20-%20PVA%20Note.pdf

4.2. The probability of a population decline cannot be reliably calculated without statistical bias – *Paragraphs 3.21 & 3.22 of the Hornsea Project One RSPB oral representation*[7]

The RSPB highlight that uncertainty surrounding both model input parameters and predicted impacts means that the probability of a population decline cannot be reliably calculated. Whilst it is acknowledged that uncertainty in the demographic parameters should be addressed through sensitivity analysis, the issue of uncertainty surrounding the predicted impacts remains. This issue also affects all probability outputs from PVAs, including those used by ABC and ruABC. Although impacts, such as collision risk and displacement, are routinely presented as a single value with no estimate of uncertainty, as outlined above (3.1) we believe that ongoing methodological advances and data collection are likely to facilitate the estimation of uncertainty, for example through the use of Monte Carlo simulations in the modelling process. However, it is important to present these estimates of uncertainty in a manner that can be easily interpreted, and to draw a distinction between uncertainty (i.e. we believe the impact will be in the range of x-y) and risk (i.e. there is an x% chance of the impact being greater than or equal to y).

Whilst it may be possible to estimate uncertainty surrounding predicted impacts, a second criticism of metrics related to the probability of a population declining remains. These metrics may be sensitive to assumptions about whether demographic rates predict future populations to be growing, stable or declining. If future populations are predicted to be growing, predicted population-level effects may be less significant than if populations are predicted to be stable or declining. This assumption is significant as policy and environmental changes including, but not limited to, climate change, fisheries discard policy and landfill closure are expected to impact populations in coming years. The exact magnitude of these changes is unknown, but may be substantial. Given the uncertain impacts of these changes on seabird population trends, understanding the sensitivity of metrics based on the probability of decline to assumptions about underlying population growth rate is necessary before they can be used with confidence.

It should also be noted that it may not be possible to use metrics related to population declines (probability of growth rate being <1, change in probability of growth rate being <1, probability of population decreasing by a given magnitude below its initial size, change in probability of a population decreasing by a given magnitude, probability of decreasing below baseline population size) to determine whether the conservation objectives of a site are being met, particularly if these are phrased relative to the population the site can potentially support. Thus, whilst a declining population is likely to reflect a site failing to reach its conservation objectives, if an impact caused a population that was previously increasing to stabilise, but at a lower level, then, although the population is not declining, the site may still not be achieving its conservation objectives. Similarly, PBR, which determines a level beyond which additional mortality will be unsustainable (i.e. the population is likely to become extinct), is generally not suited for defining acceptable population impacts. Aside from the practical challenges of defining a single sustainable level of impact in a continually changing environment, a population that is merely viable would have little resilience to any additional adverse factors that may be imposed.

We believe that in the light of ongoing methodological advances and data collection it is possible to provide some estimate of the uncertainty surrounding predicted impacts. We further suggest that the utility of metrics related to probability of population decline will depend on the context of existing population trajectories.

[7] ***Summary of Oral Case presented at the April Issue Hearings by the Royal Society for the Protection of Birds in the matter of Planning Application for the Hornsea Project One Offshore Wind Farm (Zone 4) located approximately 103km off the East riding Of Yorkshire Coast***

4.3 Models do not incorporate additional sources of mortality (e.g. drowning in fishing gear) – *Section B.2 of Annex B of Green (2014); Paragraph 3.11 of Annex V of the Hornsea Project One RSPB written representation[6]; Paragraph 3.2 of the Hornsea Project One RSPB oral representation[7]*

It is clear that the impact of offshore wind farms is only one of a number of pressures facing seabirds in northern Europe (Burthe *et al.* 2014), each of which may contribute to additional mortality. Criticisms of a failure to incorporate all sources of additional mortality when assessing the population level impacts associated with offshore wind farms typically focus on the PBR method (Green 2014), but may be equally applicable to other metrics, for example those based on PVA.

When used to determine whether population declines are attributable to anthropogenic causes, Abraham & Richard (2013) highlight that PBR should be compared to total anthropogenic mortality, as opposed to just that associated with a single source, such as an offshore wind farm. Failure to do so may result in the total anthropogenic impact on a species being underestimated. For example, in the Baltic, Zydelis *et al.* (2009) highlight a lack of data on the number of long-tailed ducks *Clangula hyemalis* killed by Russian hunters and oil pollution as an explanation for why the population is declining despite their estimates of additional mortality not exceeding that allowable by PBR. This criticism is equally applicable to metrics derived from PVAs, especially those related to the likelihood of population declines. Unless the demographic parameters used in a PVA reflect the conditions at the site in question over the period of interest, it is likely that other anthropogenic impacts will not be accounted for. However, as PVA-based metrics can be used to compare population trends with and without an impact applied (rather than simply being a binary assessment of whether additional mortality is above or below a given value), this omission may be less important in the context of PVAs.

The impact of an offshore wind farm on its own may not exceed PBR, or trigger a population decline; however, in combination with existing sources of additional mortality not accounted for in the baseline demographic rates considered, this may not be the case. It may be possible to address this issue if it can be demonstrated that the demographic parameters used in PBR or PVA calculations already account for these additional sources of mortality. If this is not the case, then outputs must be interpreted and presented more carefully. For example, it may be possible to say that *on their own* impacts from offshore wind farms will not cause a population decline or exceed PBR, but this may not be true when considered in combination with the cumulative effects of other sources of anthropogenic mortality.

We believe that the failure to incorporate additional sources of anthropogenic mortality, for example drowning in fishing nets, is likely to impact both PBR and also PVA-based metrics, particularly those linked to the probability of decline. However, given that PBR focuses on whether a certain level of mortality is exceeded or not, we believe that failure to incorporate this additional mortality is a more significant issue for PBR than for PVA- based metrics.

4.4 The recovery factors used in PBR calculations are not based on empirical evidence – *Section B.4 of Annex B of Green (2014); Paragraph 3.12 of Annex V of Hornsea Project One RSPB written representation[6]*

A key parameter in the PBR method is the recovery rate (f). Wade (1998) states that f can be seen as, "*both an additional factor to hasten the recovery of a population and as a safety factor to account for additional uncertainties other than the precision of the abundance estimate*". As f is, at least partly, designed to allow for unknown biases it is difficult to choose a value objectively, however, it is typically assumed that f takes a value between 0.1 and 1, reflecting the ability of a population to recover, with lower values set to enable a population to recover more quickly (Wade 1998). Thus, Dillingham & Fletcher (2008) recommend that *f* values should be set with reference to the conservation status of a species as assessed by IUCN and Birdlife International (IUCN 2001). Following this guidance, a value of 0.1 is used for all threatened species, 0.3 for species assessed as near threatened, 0.5 for species of least concern and 1.0 for species of least concern whose populations are known to be increasing or stable. Such definitions are generally not suitable for use in relation to a species at a single colony as the population size and trend of the species at the site of interest may not represent the conservation status of the species population as a whole. However, a similar logic of assigning *f* values based on the population status of a species at a colony concerned could be applied. For example, where a population of one species at a site is declining, it would be appropriate to select a lower *f* value than is used for a population of another species which is stable or increasing. In some ways, the use of *f* mirrors the debate surrounding the avoidance rate used in collision risk modelling (Cook *et al.* 2014) which has (incorrectly) been used as a "fudge-factor" to account for error and uncertainty in the model input parameters. Simulation approaches have been applied to guide the choice of *f* values most appropriate given biases thought to exist in parameter estimates and tolerance for risk (Richard & Abraham 2013). Such an approach is consistent with using *f* as a safety factor to account for uncertainties in the model as suggested by Wade (1998). However, using *f* as a "fudge-factor" in this way is not desirable as it potentially obscures the influence that variation in other parameters may have on the final estimate of PBR.

The final PBR values are sensitive to the *f* value assumed, with an increase in *f* from 0.1 to 0.5 reflecting a five-fold increase in the PBR value estimated. However, the value selected is rarely based on empirical evidence. Indeed, without observing populations going extinct, it is debatable whether it is possible to obtain empirical evidence in support of a particular *f* value. Consequently, while Dillingham & Fletcher (2008) argue that the selection of *f* is a management decision which should be made with reference to conservation goals, stakeholder desires and the ability to monitor a population, it should be emphasised that ultimately the value selected will necessarily be subjective.

We agree that the f values are not based on empirical evidence. Indeed Wade (1998) emphasises that f should be taken as both the recovery factor and also as a safety factor to account for uncertainty in input parameters. Therefore, without detailed quantification of the uncertainty in the model parameters, it is difficult to see how f values could be based on empirical evidence.

4.5 PBR does not quantify the impact of additional mortality on population size – *Section B.5 of Annex B of Green (2014)*

If the aim of metrics is to test whether or not the conservation objectives of a site will be met, any approach used must typically be capable of assessing whether the resultant additional mortality will mean a population can be maintained at its current level. Both Wade (1998) and Niel & Lebreton (2005) make a distinction between additional mortality exceeding PBR and a population undergoing a significant decline. Niel & Lebreton (2005) explicitly state that "*It [PBR] could be used to predict whether an additional source of mortality is unsustainable, but it cannot be used the other way around (i.e. to predict that it is sustainable).*" Indeed, the simulations of Wade (1998) demonstrate that if the additional mortality resulting from a project is equal to that obtained from estimates of PBR, populations can reach equilibrium at a point well below the carrying capacity of the available habitat (see Figs. 2, 3 & 6 of Wade 1998).

We do not believe that PBR is suitable for use in quantifying the impact of additional mortality on population size.

4.6 PBR has not been adequately validated by empirical studies – *p17, Section B.4 of Annex B of Green (2014); Paragraph 3.13 of Hornsea Project One RSPB written representation*[6]

As with methodologies used to assess collision risk, displacement and barrier effects, a key criticism of PBR is that it has not been validated by empirical studies. Green (2014) suggests that this validation could be achieved by comparing reliably measured population trends in species where additional mortality was less than the PBR value with those in species where the additional mortality was greater than the PBR value. In order to show empirical support for PBR, these studies should demonstrate that the latter populations, where the additional mortality was greater than the PBR value, were declining whilst others were not.

There are relatively few studies available with which PBR could be validated. Thompson *et al.* (2007) found that the number of grey seals *Halichoerus grypus* shot by fisheries managers in the North Sea exceeded that allowable under PBR and that this was sufficient to explain a localised population decline. However, Zydelis *et al.* (2009) in a study of fishing mortality in the Baltic and North Seas found only limited support for the use of PBR. They investigated the population effect of the additional mortality in three species. The additional mortality only exceeded PBR for greater scaup *Aythya marila* and this population was indeed declining. The long-tailed duck population was also declining despite the mortality not exceeding PBR, although they attribute this to an underestimate of the additional mortality. Conversely, however, common guillemot populations remained stable, despite additional mortality exceeding PBR. These data would suggest that PBR does lack adequate validation by empirical studies and that those studies which do exist highlight the need for improved understanding of additional sources of mortality.

We believe that where PBR has been used to assess impacts on seabirds, results have been inconsistent and do not offer empirical support for the approach.

4.7 ABC uses an arbitrary and inappropriate threshold probability value for the acceptable population size – *Section A.2 of Annex A of Green (2014)*

As outlined above (3.3), any attempt to derive a threshold with which to assess the impact of additional mortality on a population is likely to be subjective (i.e. based on an individual perception of the evidence presented), as opposed to arbitrary (i.e. based on no obvious reasoning or system), rather than having a firm biological basis, and should be acknowledged as such. Bennet (2013) uses terminology from the IPCC to identify this threshold. However, the IPCC terminology regarding the likelihood of an event occurring is inconsistent with the way in which risk is assessed at other stages of the planning process for offshore wind farms, for example, in an EIA where impacts may be categorised in relation to their magnitude, rather than their likelihood. Masden *et al.* (2014) highlight the use of consistent language as a key step in reducing the uncertainty associated with environment assessment. As such, introducing new terminology is undesirable unless it facilitates greater clarity, or allows useful new insights. Thresholds could be related to those used to assess magnitude as part of an EIA, however, it should be emphasised that, given current knowledge, these would be subjective thresholds and generally not biologically meaningful; there is thus a risk that the consequences for the seabird population may not be fully considered. We therefore feel that the task of determining whether or not they are appropriate in a particular case should be the responsibility of decision makers, once they have carefully considered all of the evidence they are presented with, and that their subjective nature be made clear.

We agree that the thresholds used to define ABC are not biologically meaningful.

4.8 Use of ABC results in perverse consequences of measurement errors – *Section A.3 of Annex A of Green (2014)*

Under ABC, if the predicted size of the impacted population is equal to, or greater than, that 66.7% likely to be achieved in the absence of an impact, the development should be deemed 'acceptable'. However, where there is greater uncertainty surrounding the demographic parameters, the distribution of predicted population sizes will have a wider spread around the median estimate. As a consequence, where there is greater uncertainty around the model input parameters, more substantial impacts will be deemed acceptable. Therefore, whether or not an impact is deemed acceptable is likely to reflect the limitations in the data rather than the ecology of the species concerned, potentially misrepresenting the population-level consequences of any impact. In recognition of this, JNCC and SNH (2014) proposed a modification to this approach, referred to as reduced uncertainty Acceptable Biological Change (ruABC). Using ruABC, uncertainty is reduced by using a regional population model which absorbs error resulting from sampling variation, but attempts to retain natural variation in demographic rates. The model prediction uncertainty from this model can then be applied to predicted population sizes at a colony level in order to identify an acceptable level of biological change. As seabird demographic parameters and population trends vary spatially (Frederiksen *et al.* 2005, Cook *et al.* 2011), it is important to ensure that the regional population is representative of the colonies concerned. If the regional populations are not representative of the colonies concerned, it may lead to bias and inaccurate estimates of the scale of impacts that a population could withstand. However, even with ruABC there is a significant risk that whether an impact is deemed acceptable or not will reflect limitations in the data, rather than the vulnerability of the species concerned.

We agree that using both ABC and ruABC, there is a significant risk that projects are deemed acceptable as a result of limitations in the data rather than the magnitude of predicted impacts. We believe that much better, colony-specific estimates of impacts and demographic parameters are needed if these approaches are to be used.

4.9 ABC uses the wrong exceedance probability distribution to define acceptable risk – *Section A.4 of Annex A of Green (2014)*

The ABC (and ruABC) approach uses a threshold determined by the uncertainty in the population size of a species predicted in the absence of any impact from an offshore wind farm. However, there is also likely to be significant uncertainty surrounding the magnitude of any predicted impact from an offshore wind farm. Disregarding this uncertainty risks giving a misleading impression of the confidence associated with the assessment of any impact and an incomplete picture of the likely risks of population-level impacts.

We agree that the ABC approach needs to consider the uncertainty in population size in both the presence and absence of a development.

5. CONCLUSIONS

The RSPB criticisms make reference to whether "the best scientific knowledge" has been used to assess the population level impacts of offshore wind farms on seabirds (Green 2014). We believe that, overall, the best scientific knowledge available is being used to assess the magnitudes of impact of individual effects on a species (e.g. collision, displacement, barrier effects). Work programmes, such as the COWRIE and SOSS programmes, other, methodological advances and reviews (Cook *et al.* 2012, 2014, Davies *et al.* 2013, Horswill & Robinson 2015, Johnston *et al.* 2014, 2015 Masden 2015) and ongoing data collection are helping to improve the knowledge base with which these impacts can be assessed, and better account for uncertainty in these impacts. With regards to the assessment of the consequences of these impacts at a population level, the situation is less clear. A range of different metrics have been proposed with which to assess population-level effects, and, as discussed above, we believe that some are more appropriate than others.

We agree with RSPB that PBR generally cannot be used to assess whether the population-level effects of offshore wind farms mean that the conservation objectives (whatever they may be) of protected sites are (or are not) being met. This is because PBR considers only whether a pre-determined level of mortality is exceeded, rather than the biological impact of any additional mortality at a population level. For similar reasons, we have concerns about metrics related to the probability of a population decline. However, these may have some merit, subject to the outcome of an assessment of their sensitivity to demographic parameters and projected population trends.

We also agree that neither ABC nor ruABC are suitable metrics given the risk that whether an impact is deemed 'acceptable' or not may reflect uncertainty in the data rather than the status of the population concerned. Of the remaining metrics, we believe that those linked to population size at a given point (e.g. the end of the lifetime of a project) and population growth rate have the most promise, subject to a careful consideration of the sensitivity of metrics to model input parameters, model assumptions and uncertainty about the population trajectory and demographic characteristics of the species concerned. Those linked to population growth rate may be of particular value in that predicted growth rates could be compared to observed growth rates from the outset of the project. This may allow a rapid assessment of the impact of an offshore wind farm on seabird populations. These metrics may be calculated from either stochastic or deterministic models, although the benefits of each of these approaches would need to be considered as part of a wider sensitivity analysis.

We agree with Green (2014) that any thresholds applied to the metrics described above are likely to be subjective, rather than biologically meaningful, and that, where they are defined, they need to be acknowledged as such, as should the risk that inappropriate conclusions may be drawn.

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