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TITLE: Lack of sound science in assessing wind farm impacts on seabirds

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ABSTRACT:

Journal of Applied EcologyVolume 53, Issue 6 p. 1635-1641 Practitioner's PerspectiveFree Access Lack of sound science in assessing wind farm impacts on seabirds Rhys E. Green, Corresponding Author Rhys E. Green [email protected] Conservation Science Group, Department of Zoology, University of Cambridge, David Attenborough Building, Pembroke Street, Cambridge, CB2 3QZ UK RSPB Centre for Conservation Science, RSPB, The Lodge, Sandy Bedfordshire, SG19 2DL UKCorrespondence author. E-mail: [email protected]Search for more papers by this authorRowena H. W. Langston, Rowena H. W. 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Wilson orcid.org/0000-0001-7485-5878 RSPB Centre for Conservation Science, RSPB Scotland, 2 Lochside View, Edinburgh Park, Edinburgh, EH12 9DH UKSearch for more papers by this author First published: 25 June 2016 https://doi.org/10.1111/1365-2664.12731Citations: 33AboutSectionsPDF ToolsRequest permissionExport citationAdd to favoritesTrack citation ShareShare Give accessShare full text accessShare full-text accessPlease review our Terms and Conditions of Use and check box below to share full-text version of article.I have read and accept the Wiley Online Library Terms and Conditions of UseShareable LinkUse the link below to share a full-text version of this article with your friends and colleagues. Learn more.Copy URL Introduction Electrical power generation from wind farms has grown rapidly in the UK and European Union (EU) in the last decade and is set to grow further. By 2020, the EU proposes to source 20% of energy from renewable sources (Directive 2009/28/EC). Wind energy is expected to provide 9?14% of global electricity generation by 2050 (IPCC 2011). This may eventually reduce climatic change and its negative impacts on biodiversity, but there are also several poorly quantified negative effects on wild species of renewable energy generation, including wind turbines. For example, birds and bats are killed by colliding with turbine blades or towers and there may be effects of wind farms on mortality and reproductive rates of a wide range of species from avoidance and displacement. Birds may incur additional costs or forego benefits because of reduced transit or foraging within or near to wind farms (Drewitt & Langston 2006; Searle et al. 2014). Depending upon the strength of density-dependent compensatory processes, these effects could reduce the population to a lower stable level or cause its extinction (Wade 1998; Niel & Lebreton 2005). Except in the rare circumstances where density dependence is exactly compensating, such effects would always diminish population size. Positive effects of renewable energy infrastructure on populations of wild species have also been proposed and, in a few cases, quantified. These include possible enhancement of food resources of seabirds by protection from fishing from the presence of offshore installations and the provision of artificial substrates as habitat for fish and invertebrates (Inger et al. 2009; Langhamer, Wilhelmsson & Engström 2009). The UK has the best wind resources in Europe (DECC 2011). Although the cost per megawatt-hour of electricity generation from offshore wind turbines averages about twice that for onshore installations (Bilgili, Yasar & Simsek 2011; Chu & Majumdar 2012), offshore wind power is currently favoured over onshore by the present UK government because of public perceptions of nuisance and landscape consequences of onshore turbines. The UK also has internationally important breeding populations of seabirds. It holds more than 10% of the world's breeding population of eight species, of which three have more than half of their global breeding population in the UK (Brown et al. 2015). Because seabirds range over long distances, there may be cumulative impacts on a breeding colony from several wind farms (Masden et al. 2010). Seabirds are long-lived and late-maturing, which renders their population growth rate particularly sensitive to additional mortality from collisions or displacement (Niel & Lebreton 2005). The importance of these seabird populations and their sensitivity places a heavy responsibility on those conducting and acting upon scientific assessments of the impacts of offshore wind farms on seabirds to comply with the protection measures and the precautionary principle enshrined in the EU Birds and Habitats Directives (Directive 2009/147/EC and Council Directive 92/43/EEC). For the UK, and other countries within the European Union, the regulation of wind farm construction requires the assessment of possible damage to the integrity of sites and populations under the EU Habitats and Birds Directives. Consideration must be given to impacts on bird populations of a project on its own and in combination with others already in existence, given consent or planned. Governments give or refuse consent for the construction of wind farms after taking into account the scale and level of certainty of the impacts indicated by these assessments. However, there are no definitive quantitative thresholds or criteria defining how large or likely expected impacts must be for damage to the integrity of sites and populations to be anticipated and for consent for wind farm construction to be denied or limited. Consent can be granted only if it is ascertained that there will not be an adverse effect on the integrity of a Natura site, excepting in cases where there are imperative reasons of overriding public interest for consent and no alternative solutions (Article 6(4) of Directive 92/43/EEC). In recent years, several plans for large offshore wind farms have been approved and some built in UK and EU waters close to large seabird populations because the competent authority judged there was no expected adverse effect on the integrity of the Natura sites involved. For example, in 2014 approval was granted for several extensive wind farms at Hornsea (England, UK Government) and the Firth of Forth (Scotland, Scottish Government), close to internationally important breeding populations of seabirds. This approach contrasts with that in some other EU states. In Germany and Denmark, for example, offshore wind farms have been subject to rigorous marine spatial planning with the aim of avoiding potential conflict with nature conservation as part of the required Strategic Environmental Assessment (SEA) process recommended in EU Commission guidance (European Commission 2011). The German Cabinet approved Europe's first maritime spatial plan in September 2009, after a considerable effort in terms of surveys and research to identify marine sites of high nature value and potential conflict areas with wind farms and to establish zones for various activities and infrastructure. The offshore SEA covering UK waters is not of comparable quality. In this perspective, we argue that the methods and data used in these cases for estimating effects upon seabird demographic rates and translating them into potential impacts on seabird populations do not allow adequate assessment of effects on site integrity. As a result, sound science and its logical interpretation are lacking in Environmental Impact Assessments of this large and expanding industry. Estimates of the effects of wind farms on seabird demographic rates are neither robust nor validated Collision risk models (CRMs) are used to predict the number of fatal collisions of flying birds with wind turbines and per capita additional mortality rates. In the UK, the most widely used CRM is that of Band (2012) (see review by Masden & Cook (2016)). The model requires estimates or assumptions about bird numbers and ages at the wind farm, attribution of birds at the wind farm to source populations, sizes and age structure of source populations, flight behaviour and avoidance rates. Data specific to the project and species being assessed are usually collected on seabird numbers and flight heights, judged by eye, but these estimates are subject to substantial uncertainties, variability and potential biases (Johnston et al. 2014), including: accuracy of input variables is rarely quantified, is often poor, and the CRM outputs are highly sensitive to the values used, including flight speed (Masden 2015), and avoidance rate estimates; in many cases, birds at risk are not attributed to source populations because recently developed tracking technologies are either not deployed at all or not on a sufficient scale for robust estimation; count and flight height data are usually insufficient in quantity and quality for precise estimation of seasonal variation, age structure and age differences (Band 2012). Total avoidance rates used for CRM calculations for seabirds, including within-wind farm avoidance of individual turbines and macro-avoidance by movement of birds around the turbine array, are most often based upon judgement or extrapolation from other contexts rather than pertinent data. Empirical values are only available from a few species (mostly gulls and terns) and usually extrapolated from studies of onshore wind farms, where different circumstances prevail (Cook et al. 2014). Robust direct estimates of within-wind farm avoidance rates are lacking for seabird species frequently present in and near planned and consented offshore wind farms in the UK, such as northern gannet Morus bassanus and black-legged kittiwake Rissa tridactyla (Cook et al. 2014). Macro-avoidance and displacement rates have been estimated using radar, visual surveys and imaging, but robust quantitative estimates with confidence intervals are generally not used in impact assessments. Estimates of macro-avoidance for the same species can be highly variable (e.g. Petersen et al. 2006; Krijgsveld et al. 2011; Vanermen et al. 2012, 2013 for northern gannet). This may well be because macro-avoidance varies with the relative positions of nesting and foraging sites, foraging site quality and seasonal timing of studies. At onshore wind farms, carcasses of some of the birds killed by collisions with turbines can be collected during systematic searches and probabilities of their detection can be estimated. This allows estimation of numbers of deaths per unit time and confidence intervals, even if with low precision (e.g. Bellebaum et al. 2013). These methods help to quantify uncertainty and remove bias, but are currently impractical for offshore wind farms. Alternatives that use video or thermal camera systems have not yet been deployed sufficiently to substitute for them. Where direct measurements of avoidance rates are lacking, Band (2012) recommends use of a range of plausible values. However, this can result in a 20-fold variation in assumed per capita mortality rates (APEM 2015). Overall, CRM outputs are sensitive to the combined effects of multiple assumptions of unknown accuracy, sampling errors and unquantified biases. Only for species that almost completely avoid entering wind farms can the annual per capita mortality rate from collisions be estimated reliably and with robust confidence limits (Desholm & Kahlert 2005). Validation tests of offshore seabird CRM outputs, in which expectations from pre-consent data and modelling are compared with independent robust post-construction measurements of numbers of collision deaths, have not been conducted. Estimation of effects on seabird demographic rates of the displacement and barrier effects of wind farms is even less well developed. Avoidance of wind farms by foraging and migrating birds can be substantial and operate over long distances from the turbines (Desholm & Kahlert 2005; Petersen et al. 2006; Percival 2010), but the degree to which this affects travel times and costs, access to food and mortality and reproductive rates of breeding seabirds has not been measured reliably. In the case of migrating birds, the displacement and increased travel costs caused by avoidance of a single wind farm may be trivial relative to the total length and cost of the journey (Masden et al. 2009), but effects on demographic rates have not been robustly quantified by empirical studies for central-place foraging breeding seabirds repeatedly subjected to barrier or displacement effects. Simulation modelling has been performed of potential effects of displacement by as yet unconstructed wind farms on seabird time and energy budgets and demographic rates (Searle et al. 2014). Modelled potential effects of displacement included considerable declines in adult survival of up to 2·1% for black-legged kittiwake and up to 4·9% for Atlantic puffin Fratercula arctica (both for the Forth Islands cumulative effects: table 3·3 of Searle et al. 2014), though simulated effects on survival for other species and sites and for breeding productivity generally were small. The species for which collision mortality can be reliably estimated as low, because of strong avoidance, are those for which displacement and barrier effects upon demographic rates are potentially the largest, but currently unquantified. In summary, the procedures currently used to calculate expected effects of proposed wind farms on seabird per capita mortality rates and breeding success largely involve modelling with little firm empirical data. Moreover, actual outcomes at wind farms that have been constructed have not been measured, so model predictions are not tested and there is no adaptive improvement of the decision-making process (Nichols et al. 2015). As a result, scientifically robust and defensible calculations of effect sizes for changes in seabird demographic rates caused by collision, displacement and barrier effects of offshore wind farms, with confidence intervals, are currently lacking. Procedures for translating effects on demographic rates into projected impacts on seabird population size and trends are inappropriate and untested Assessments of the impacts of offshore wind farms in the UK on seabirds require that the highly uncertain estimates of effects on demographic rates are translated into projections of impacts on population size or trend. Decisions about UK offshore wind farms have been based upon, or influenced by, the following effect?impact translation procedures. Potential biological removal (pbr) The recommended and robust application of this method is to identify a level of additional mortality above which a decline of the affected population to eventual extinction would be likely (Niel & Lebreton 2005). In recent cases, such as Hornsea, the UK statutory conservation agencies advised using this method in wind farm assessments to identify demographic rate thresholds below which additional mortality estimated from CRMs and related methods is unlikely to adversely impact the population (Natural England 2014). This reverse application involves faulty logic because PBR's value of maximum potential excess growth may not be realizable in the ecological circumstances of a particular population of interest. In addition, PBR does not estimate the effect of additional mortality on population size. Potential biological removal provides thresholds of additional mortality that are sensitive to assumptions made about the form of density dependence. The studies of Wade (1998) and Bellebaum et al. (2013) show that the shape parameter of the generalized logistic equation has a strong effect on PBR results. Details of the form of density-dependent relationships are rarely known for animal populations and are unknown for any of the UK seabird populations to which PBR has been applied. These uncertainties have prompted the use of ?recovery factors?, which are constants by which the maximum possible value of the PBR threshold is multiplied to give a safety margin (Dillingham & Fletcher 2008). The values used for these recovery factors are based upon judgement. There has been no empirical validation of their safety by observation of the effects on population size of known additional mortality rates from any source in any bird species. Acceptable biological change (abc) This method, which has not yet been published in the peer-reviewed scientific literature, was developed by Marine Scotland, a Scottish government agency, and used in a recent assessment of the impact of wind farms on internationally important seabird populations in the Firth of Forth (Marine Scotland, 2015). It uses probabilistic forecasts from stochastic seabird population models to assess the probability of a particular level of population size occurring at some future time, such as the end of the period of operation of a wind farm, in the absence of the wind farm. In practice, this probability is obtained from a simulation model of the population in which variation in expected future population size arises from supposed future demographic and environmental stochasticity in demographic rates, when applied to the population of a specified initial size over a period of 25 years, which is the usual licence period for an offshore wind farm. If the best estimate of future population size, after the expected effects of the wind farm on demographic rates are taken into account, equals or exceeds the population size that is 66·7% likely to be equalled or exceeded in the absence of the wind farm, then ABC deems that the impact of the wind farm is acceptable. The weaknesses of this approach are severe. First, the accuracy of projections of the demographic rates used in the model of the unimpacted seabird population long into the future is highly uncertain and untested. Perversely, the greater the estimated uncertainty, the larger the acceptable population decline. Secondly, it does not address the uncertainties in size of the effects of the wind farm on demographic rates, which are mostly unquantified. Hence, ABC does not assess the risk or probability that the wind farm itself will cause a particular specified outcome or change at all. It simply proposes that an event half as likely to occur as not if there is no wind farm should be the threshold for acceptability. Thirdly, the threshold probability for acceptance is arbitrary and is plucked from an unrelated context: IPCC guidelines about the appropriate language to describe the likelihood of an event or outcome of at least given size happening, based upon available evidence (Mastrandrea et al. 2010). The threshold chosen for ABC is described as ?unlikely? in the IPCC lexicon. However, this lexicon was not developed for the purpose of determining acceptable levels of risk, which also requires that the societal costs and benefits of possible outcomes are evaluated. It is not only the chance of being wrong that is important, but also the scale of the damage caused by being wrong. No justification is given by the proponents of ABC for using as a tolerable risk threshold for damage to important nature conservation sites and their species a term selected arbitrarily from a lexicon developed by IPCC for a different purpose. Decline probability difference (dpd) method Large uncertainties in predicting future seabird population changes might not matter if differences in the probability of a specific population outcome between scenarios with and without wind farms could be predicted reliably and used as criteria for acceptability. This focus on differences in risk has been proposed by the Joint Nature Conservation Committee & Natural England (2012). It was suggested that assessments of acceptable impact should be based upon an arbitrary threshold level of absolute difference between the impacted and unimpacted scenarios in the probability that a population decline by an arbitrary proportion of the initial level would occur. In principle, this approach is preferable to ABC because it takes the uncertainty in the predicted magnitude of the effect of the wind farm into account. However, the results of this procedure are sensitive to the selection of unpredictable baseline (unimpacted) demographic rates. For example, in a model in which the selected values of baseline demographic rates imply a rapid increase in projected population size, it is unlikely that even large additional mortality would give rise to an appreciable absolute difference in the probability of population decline between impacted and unimpacted scenarios. Both probabilities would be very small. If the selected rates were inaccurate and the true values instead led to the unimpacted population being approximately stable, the same level of additional mortality could result in a large difference in the probability of population decline between impacted and unimpacted scenarios. In practice, uncertainties in future projections of both unimpacted and impacted populations are mostly unquantified, so the probability distribution of an outcome for population size cannot be calculated. This problem makes approaches, such as ABC and DPD, which are based upon assessments of probability or difference in probability unworkable, given present knowledge. The danger of acceptability thresholds without a logical or empirical basis All the effect?impact translation procedures described above have a built-in threshold for an acceptable impact. Such thresholds are naturally attractive to decision-makers because they appear to offer a clear-cut, evidence-based way to establish whether damage to the integrity of a designated site will or will not occur. However, in the case of ABC and DPD, the thresholds offer only false security because they are arbitrary, have no foundation in population biology and embed the acceptance of some adverse impact on population size. Whilst PBR does identify a threshold based upon population biology, it is one that is misapplied to the problem at hand. PBR could be used to identify a threshold level of effect of wind farms on demographic rates above which a decline of an affected closed population to eventual extinction would be almost certain. However, population declines of a wide range of magnitudes, short of extirpation, could be caused by effects of wind farms on demographic rates well below this. How large these declines would be depends upon the form and strength of density dependence, which are unlikely to be measured with sufficient precision, and the magnitude of such declines has not been quantified using PBR in any UK wind farm assessment. We argue that such declines would constitute adverse effects on site integrity. Hence, PBR is not an appropriate method for assessing population impacts of a development in a manner that is relevant to the concerns of the public and decision-makers. A robust effect?impact translation procedure without a built-in threshold A more robust procedure for evaluating population-level impacts of wind farms on seabirds is to calculate, using a density-independent Leslie matrix model (LMM), expected population sizes, with and without the expected effects on demographic rates of the wind farm, at the end of its lifetime. The ratio of the expected population size with the wind farm to that without it (the counterfactual of population size) is a robust metric for likely population-level impact of a specified set of effects of the wind farm on seabird demographic rates. This LMM-ratio approach is relatively insensitive to the assumptions made about the magnitude, variability and trends of demographic rates in the model from which it is calculated, because the same uncertainties apply to both the impacted and unimpacted scenarios. Hence, this effect?impact translation procedure contributes little to the uncertainty in the difference in population size caused by the wind farm. Density dependence tends to reduce the impact on population size of a given effect of the wind farm on demographic rates, so the LMM ratio calculated from the density-independent model is a precautionary worst-case outcome. We think it probable that density-dependent compensation occurs in UK seabird populations and that including it in LMMs (e.g. Miller, Jensen & Hammill 2002) could lead to more accurate estimates of population impact than those based upon density-independent LMMs. However, accuracy would only be increased if robust estimates of the form and strength of density dependence were available or population outcomes could be shown to be insensitive to assumptions made about density dependence in the absence of reliable quantification. In practice, no assessments of population impacts of additional mortality from wind farms on UK seabirds have included empirical estimates of the form and strength of density dependence because applicable estimates seem not to be available. Until adequate quantification of density dependence is available, we recommend the use of density-independent LMM ratios. Whether density dependence is included or not, there is no threshold value of acceptability built into the LMM-ratio metric. Population estimates from Leslie matrix models, for example Trinder (2014), and population models fitted using a Bayesian approach (Marine Scotland, 2015) have been calculated as part of offshore wind farm impact assessments, but their results have not been used explicitly as counterfactuals in decision-making about the acceptability or otherwise of UK offshore wind farm projects. Based upon the documentation of UK wind farm assessments, we believe that methods such as PBR, ABC and DPD have been used in preference to LMMs because they provide thresholds which can be used to argue that site integrity will not be affected by the project, whilst LMMs deliberately do not provide a threshold. We argue that, because the thresholds offered by the other methods are arbitrary and invalid, LMMs should be used as the standard, best-practice method, and we note that any of the potential positive effects of offshore wind farms on seabird demographic rates, if quantified, could be included in an integrated assessment using an LMM-ratio metric. Conclusions Current procedures for collecting empirical data, modelling effects on demographic rates and translating those effects into projected impacts of offshore wind farms on seabird populations are inadequate. Empirical measurements of effects of offshore wind farms on seabird demographic rates from fieldwork are not sufficiently precise and unbiased. In the case of some important parameters such as turbine avoidance rates and the strength of density-dependent compensation, estimation is rarely even attempted. As a result of these holes in the evidence base, the magnitude of effects of wind farms on seabird demographic rates cannot be estimated accurately and the level of bias and precision in the estimates used cannot be calculated. To overcome these problems, responsible governments should require the renewable energy industry to co-fund an adequate level of field-based research to estimate effects of wind farms on seabird demographic rates more reliably. The Offshore Renewables Joint Industry Partnership (ORJIP) intends to address this need (Carbon Trust 2015), but the objectives of its project need to be greatly expanded with regard to the number of species covered, proximity to their breeding colonies and robustness of estimation. Further development and deployment of radar, imaging and tracking techniques are likely to be required, including remote download 3D tracking (Cleasby et al. 2015). A defensible approach is then needed to translate these effect measurements, and their uncertainties, into expected impacts on populations. We propose that the counterfactual population ratio from a density-independent Leslie matrix model would be an appropriate method for this translation. Quite separate from these problems of measurement, estimation and modelling, there is a fundamental logical flaw in the link between scientific assessment and decision-making about the acceptability of wind farm impacts. Modelling approaches have been contrived that seek to define an acceptable threshold for a projected negative impact of a wind farm on seabird populations, below which this negative impact is regarded as causing no adverse effect on site integrity. However, the emperor has no clothes: the thresholds used to define the acceptability of projected offshore wind farm impacts are arbitrary, poorly reasoned, not designed for the purpose and have no valid biological basis. Hence, it is necessary to revise decision-making procedures, regardless of what effect-to-impacts translation procedure is

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