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Research article

Towards a framework for quantifying the population-level consequences of anthropogenic pressures on the environment: The case of seabirds and windfarms

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1. Introduction

Human activity is having a substantial impact on the natural environment (Vitousek et al., 1997) and where this affects biodiversity it can have knock-on consequences for ecosystem stability (Hautier et al., 2015). Whilst there is a tendency to focus on rare species, impacts on more common species can also have significant consequences at a population or ecosystem level (Gaston and Fuller, 2008; Inger et al., 2015). Consequently, there is a need to develop methods and approaches which can be used to quantify the impacts on individuals in order to understand their consequences at a population-level, whilst accounting for any uncertainty in the available data. A key challenge is to ensure that any methods can be incorporated into policy decision frameworks, such that they provide clear guidance on the relative risks of realistic management options (Bakker and Doak, 2009). This concept applies equally to interventions that are expected to have a positive effect on populations, such as population translocation (e.g. Canessa et al., 2016) or reserve designation (e.g. Fenberg et al., 2012), as to those with negative impacts. Integrating ecological processes with the social and economic goals associated with resource management represents a significant challenge because of the complexity of the systems concerned (Dale and Beyeler, 2001).

The pace of change has been particularly noticeable in the marine environment (Halpern et al., 2015), with key concerns about the rapid increase in the number of offshore structures and their potential to impact wildlife (e.g. Bailey et al., 2014). The development of large offshore wind farms, often seen as a key part of strategies to reduce reliance on fossil fuels (Toke, 2011), has been particularly significant. These potentially have a number of negative effects on seabird populations including: displacement from foraging areas, collision with turbines, and the wind farm acting as a barrier to migrating or commuting birds (Everaert and Stienen, 2007; Masden et al., 2009; Vanermen et al., 2013). Each impact can be estimated prior to construction by characterising the environment (e.g. Johnston et al., 2015) and applying tools, such as collision risk models (Masden and Cook, 2016). Whilst these approaches can be used to assess the total number of individual birds which may be affected by a development, understanding the implications at a population level is more complex (Maclean et al., 2014).

Tools such as Population Viability Analysis (PVA) are widely used in population management, for example; in order to predict the likely success of different conservation interventions (e.g.Lindenmayer and Possingham, 1996), to determine whether levels of population harvesting are sustainable (e.g. Tufto et al., 1999) and to investigate the efficacy of pest control programmes (e.g. Brook et al., 2003). However, predicting the viability of populations after interventions requires some understanding of the underlying demography of the population, so that a population model can be constructed (e.g. Oro et al., 2004). Our confidence in estimates of the demographic parameters varies considerably by species and population, and must be accounted for if any assessment process is to be effective (Horswill and Robinson, 2015).

In the context of wind farms, for example, a key aim of PVA may be to demonstrate whether or not the level of additional mortality, for example resulting from collisions with turbines, will have an adverse effect at a population level. The predicted impacts can be

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incorporated into PVA models as an additional level of harvesting to understand the population-level consequences associated with the presence of a wind farm. From these models a range of metrics can be calculated, such as population size at time, t (N_t), which may reflect the end of the operational life of the wind farm, the (mean) population growth rate (λ), or probability of extinction (Brook et al., 2000; Ellner et al., 2002) (Fig. 1). These metrics have been applied to assess the impact of management interventions in a variety of different ways. These include, comparing the impact of timber management strategies on the probability of extinction (Lindenmayer and Possingham, 1996), the probability of detecting a given population decline under different conservation strategies (Thompson et al., 2000) and the probability of a population of a reintroduced species being above a given size through time (Wood et al., 2007). When used in a management context, the implications of different (or no) interventions must be presented in a way which is clear so that they can be easily interpreted by decision makers. This requires that these metrics (e.g. population size, population growth rate or probability of a given outcome) be assessed in relation to some criterion, which may be a simple threshold defining a desirable outcome (which may be no effect), perhaps relative to a population not subject to any intervention, or a probabilistic assessment of a range of outcomes.

Recently, Green et al. (2016) summarised three criteria which may be derived from PVAs in order to quantify population level effects arising as a result of the impact of offshore wind farms – Acceptable Biological Change (ABC), Decline Probability Difference (DPD) and Counterfactual of Impacted to Unimpacted Population (CIU) (Table 1). However, interpreting these criteria can be fraught with difficulty and has been the subject of much debate between stakeholders (e.g. Court of Session of Scotland, (2016)), contributing to costly delays in the decision making process (Masden et al., 2015).

The theoretical basis underpinning the models and criteria used to make decisions about the population level effects of management interventions has been criticised (e.g. Coulson et al., 2001; Green et al., 2016). To inform their use in the decision-making process there is a need for an assessment of the strengths and weaknesses of these different approaches. We present a unified framework (Fig. 1) for assessing the impact of developments at a population level using criteria informed by metrics from Leslie matrix models (LMMs) of populations with and without the impact of a management intervention. We test this framework using the example of an offshore wind farm, which may affect survival rates of a seabird population through collision-related mortality or productivity rates through displacement (Drewitt and Langston, 2006). We aim to derive measures within this framework that reflect the impact of the intervention in a clear and unequivocal fashion, rather than differences in model properties (e.g. density dependence, stochasticity) or knowledge of the values of demographic parameters. We discuss how conclusions about the acceptability of any impacts may be influenced by uncertainties either in data availability, or in the modelling process, and what implications this may have for the consenting process.

2. Methods

We considered a generic seabird species (Table 2) with life history traits informed by a recent review of seabird demography (Horswill and Robinson, 2015) and constructed Leslie matrix models with four age-classes, and reproduction confined to the adult age class (≥four years). To simulate the impacts associated with the presence of a wind farm, we modelled a broad range of reductions in productivity of 0–40% and increases in mortality of 0–40% across all age classes, taken as reflective of the range of impacts considered in Environmental Impact Assessments. Models were run for 25 years, taken as the typical life span of an offshore wind farm. All analyses were carried out using R 3.1.1 (R Core Team, 2015).

2.1. Decision criteria

In order to understand how decision criteria behave when applied to the three different metrics we test two variants of each,

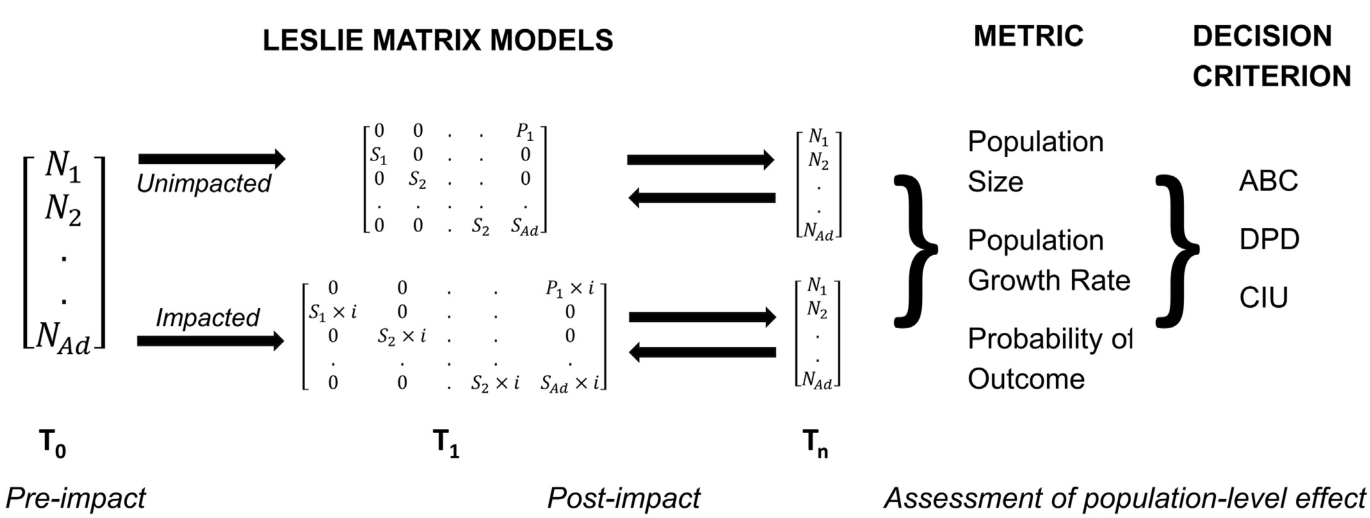

Fig. 1. Framework used to derive decision criteria to assess the population level effects associated with management interventions. At T_0, pre-intervention, there is a population with N individuals in each age class. Using a "matched runs" approach, population changes are projected over the lifetime of the project from T_1 to T_n. Two populations are modelled, the first in which no impact from the management intervention is assumed with demographic parameters S_1 (first year survival), S_2 (sub-adult survival), S_{Ad} (Adult survival) and P_1 (Productivity). The second population is modelled with each parameter modified by an impact associated with the intervention, i. For simplicity, we assume equal impacts for each parameter. In practice, this is unlikely to be the case. Three metrics can be derived from these models, population size, population growth rate and the probability of an outcome (e.g. extinction, or a population decline of a given magnitude). The metrics from the impacted and unimpacted populations can then be compared using three decision criteria – Acceptable Biological Change (ABC), Decline Probability Difference (DPD) and Counterfactual of Impacted to Unimpacted Populations (CIU) –to quantify the population-level effect associated with the management intervention.