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Seabird–wind farm interactions during the breeding season vary within and between years: A case study of lesser black-backed gull *Larus fuscus* in the UK

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ABSTRACT

The marine environment is increasingly pressured from human activities, such as offshore renewable energy developments. Offshore wind farms may pose direct risks to seabirds at protected breeding sites. However, changes in food availability may influence foraging behaviour and habitat use during the breeding season or between years. Consequently, seabird–wind farm interactions, and risks posed to populations, may vary over longer time scales, but this has seldom been quantified. We used GPS-telemetry to study the movements of 25 lesser black-backed gulls from the Alde–Ore Special Protection Area (SPA), UK between 2010 and 2012, while birds were associated with their breeding colony. Variation in movements away from the colony, offshore, and in operational, consented and proposed Offshore Wind Farm Areas ("OWFAs") was investigated: (1) between years and (2) across the breeding season, addressing: (3) sex-specific, (4) individual and (5) diurnal/nocturnal differences. The extent of overlaps with OWFAs varied between years, being greatest in 2010 (7/10 birds showing connectivity; area overlap: 6.2 ± 7.1%; time budget overlap: 4.6 ± 6.2%) and least in 2012. Marine habitats close to the colony were used before breeding. Birds spent little time offshore as incubation commenced, but offshore usage again peaked during the early chick-rearing period, corresponding with use of OWFAs. Individuals differed in their seasonal interactions with OWFAs between years, and males used OWFAs significantly more than females later in the breeding season. This study demonstrates the importance of tracking animals over longer periods, without which impact assessments may incorrectly estimate the magnitude of risks posed to protected populations.

1. Introduction

The marine environment is under increasing pressure from human activities, such as fisheries, shipping and boat traffic, oil and gas, and renewable energy (Syvitski et al., 2005; Halpern et al., 2008). Offshore wind farms are a key part of the UK Government's plan to obtain 15% of energy from renewable sources by 2020 (DECC, 2009). It is therefore important to properly quantify the potential impacts that proposed offshore wind farms, alongside those operational or consented (hereafter together termed as Offshore Wind Farm Areas, "OWFAs"), could have on marine wildlife and habitats.

Seabirds are key components of marine ecosystems, and may be affected by offshore wind farms through direct collision mortality, displacement from foraging areas, diversion of flight paths, or through changes to habitats and prey (Garthe and Hüppop, 2004; Masden et al., 2009; Furness et al., 2013). In the UK, full consideration is given to each of these effects through the Environmental Impact Assessment (EIA) process. The potential impacts on populations of birds at protected sites, for example sites classified as Special Protection Areas (SPAs) under the European Union's Birds Directive (Directive 2009/147/EC), are given consideration through Habitats Regulations Assessments (HRA). Specific data on the links ("connectivity") between a particular SPA and the development of interest are often lacking, meaning that precautionary information, such as representative foraging ranges (Thaxter et al., 2012) may be required to evaluate potential impacts. Consequently, there is a pressing need to directly demonstrate connectivity between

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breeding sites (where breeding seabird species are classified as a SPA feature) and areas used at sea.

Greater availability and affordability of tracking technologies have offered an increasing number of ways to assess the likely impacts of offshore renewable energy developments on wildlife (Carstensen et al., 2006; Desholm et al., 2006; Scheidat et al., 2011). For seabirds, telemetry is a particularly useful tool in this regard (see Gyimesi et al., 2011; Langston et al., 2013; Bogdanova et al., 2014; Wade et al., 2014). Species-specific requirements and economic restrictions determine when and how many birds to tag and the types of tracking devices (hereafter "devices") used. Affordable short-life devices are very informative, but for seabirds their use is normally restricted to periods when devices can be deployed and retrieved from birds. Furthermore, information obtained for short periods may not be representative of a species' movements within the breeding season, or typical of movements in comparison to other years. It is well known that changes in food availability may alter the behaviour of birds through the course of a single breeding season and between years (Bearhop et al., 2001; Hamer et al., 2007; Pettex et al., 2012), which may lead to use of alternative foraging areas and different commuting patterns. Therefore, the movements of birds needs to be characterised over longer time periods (Bogdanova et al., 2014) to fully appreciate the potential for variation in seabird–wind farm interactions, and properly evaluate the risks posed to protected sites within EIAs.

For generalist species, some birds may exhibit dietary specialisation (Bolnick et al., 2003; Martins et al., 2008), resulting in individual differences in foraging behaviour. For instance among seabirds, different foraging tactics or particular trips may be required for certain prey (McCleery and Sibly, 1986; Watanuki, 1992; Woo et al., 2008). Foraging behaviour may also differ between sexes and therefore influence habitat use (Lewis et al., 2002; Thaxter et al., 2009). The relative probability of a flying bird colliding with an offshore wind turbine may also be greater at night than during the day due to reduced visibility (Garthe and Hüppop, 2004). However, differences in the movements of birds between daytime and night-time (Camphyusen, 2011), may also determine the risk of collision. Quantifying these additional sources of variability is necessary to build up a coherent picture of seabird–wind farm interactions.

This study focuses on the lesser black-backed gull, UK sub-species *Larus fuscus graellsii*, a breeding feature at 10 SPAs in the UK (Stroud et al., 2001). The foraging distribution and habitat associations of this species have been studied using at-sea surveys (e.g. Kubetzki and Garthe, 2003), and the species is increasingly being tracked from breeding colonies in Europe (Shamoun-Baranes et al., 2011; Klaassen et al., 2012). Lesser black-backed gulls may forage up to 180 km offshore during the breeding season (Thaxter et al., 2012). Hence, there is potential for birds from several UK colonies to forage in areas of OWFAs. Lesser black-backed gulls are considered at high collision risk (Furness et al., 2013), flying at heights (during commuting and foraging) within the rotor sweep zone (Johnston et al., 2014; Corman and Gathe, 2014), making it necessary to characterise their total area usage away from the breeding colony.

We used a long-term GPS system (Bouten et al., 2013) to investigate the movements of lesser black-backed gulls from an SPA in the UK. Using data collected over three separate years, we investigated whether time budgets, area utilisation, and in-turn the likelihood of interactions with OWFAs varied significantly: (1) between years, (2) during the breeding season, while also addressing potential (3) sex-specific, (4) individual, and (5) day-time and night-time variations in behaviour. There were very few constructed offshore wind farms at the time of the study to investigate any effects on behaviour before or after construction (4cOffshoreWind, 2015). Therefore our main aim was to assess potential exposure to OWFAs (proposed, operational and consented) and the sources of variation that can influence this.

2. Materials and methods

2.1. Study site and period

Lesser black-backed gulls were studied at a colony of 550–640 apparently occupied territories (AOTs) at Orford Ness (Marsh, 2013), part of the Alde–Ore Special Protection Area (SPA), Suffolk, UK (52°06′N, 1°35′E). The study took place from June 2010 to October 2012, from first recording of individuals at the colony to departure. This covered pre-breeding (return to colony to first egg, *ca.* February to May), breeding (incubation and chick-rearing, *ca.* May to July), and post-breeding periods (post-fledging or failed to departure, *ca.* July to October). The OWFAs within the potential foraging range of lesser black-backed gulls (Thaxter et al., 2012) at Orford Ness are given in Fig. 1 (see also Appendix A in Supplementary Information).

2.2. Capture methods and attachment of devices

Birds were caught at the nest site during early incubation using a walk-in wire mesh trap. During 2010, GPS devices (Bouten et al., 2013) were attached to 11 birds using either: a leg-loop harness ($n = 3$ birds), body harness with a breast strap ($n = 4$ birds), or wing harness ($n = 4$ birds). During 2011, devices were attached to a further 14 lesser black-backed gulls using a wing harness (Thaxter et al., 2014). Birds were sexed using head and bill length measurements (2010, n = seven males, three females; 2011, n = seven males, six females; two unidentified, one in each year, due to uncertainty) that were recorded along with body mass on capture (Coulson et al., 1983; Camphyusen, 2011). One GPS device deployed in 2010 provided no data (male bird), giving a total sample size of 24 birds across all years for further analysis. The total weight of devices (plus harness) was 21 g (<3% body mass, mean weight 851 ± 85 g, range: 710–955 g). The potential effects of the GPS devices and harnesses used in this study were assessed through comparison with a separate group of control birds. There were no significant differences between harness and control groups in measures of productivity or over-winter survival ($P > 0.05$), indicating that the GPS device and wing harness had negligible effects for the species in this study (C.B. Thaxter Unpublished data); thus behaviour is considered representative.

2.3. Productivity and breeding periods

The nests of tagged individuals were monitored during their breeding season of capture through approximately weekly visits in order to assess the variation in the productivity of the colony between years and the timing of breeding periods. Tall vegetation and mobility of chicks prevented the following of nest survival to fledging. For the same reasons, the nests of tagged birds could not be monitored beyond the first season in which they were tagged. Therefore, the productivity of additional ("control") nests of unmarked birds was monitored in 2011 ($n = 46$) and 2012 ($n = 51$). Productivity was assessed through: (i) number of eggs hatched per nest, and (ii) number of chicks present at the end of monitoring (up to mid-July). Where there was uncertainty (between colony visits), mean minimum and maximum estimates were calculated. Chicks were monitored up to 11 July in 2010, 9 July in 2011 and 23 May 2012; in 2012 subsequent visits in early June to July revealed the colony had suffered a breeding failure.