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Seabird ecology in high-energy environments: approaches to assessing impacts of marine renewables

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Abstract

It has been widely acknowledged that a global change in energy production, from fossil fuels to renewable sources, is required in order to reduce carbon dioxide outputs and help mitigate anthropogenic climate change. The UK is recognised as having one of the largest practical marine energy resources in Europe, including ‘wet renewables’ energy sources; tidal-stream and wave energy. Scotland, as well as having some of the best marine energy resources, also holds internationally important numbers of breeding seabirds. Both wave energy and tidal stream devices have the potential to place a new anthropogenic pressure on already declining seabird populations. Wet renewables are predicted to impact seabird populations through collision, disturbance, habitat loss and changes to food availability. However, few devices have been deployed to enable monitoring of impacts and there have been few studies of the ecological implications and magnitude of any impacts to these populations. The aim of this thesis is to address key knowledge gaps relating to our understanding of seabird ecology in high-energy marine environments, specifically exploring spatial, temporal and habitat drivers for their use of these areas. This is to help expand the scientific basis used to assess the impacts of marine renewable energy devices on marine birds. This thesis also aims to increase our understanding of bird survey requirements for wet renewable energy schemes to help develop methods for environmental impact assessment. Chapters 2, 3 and 4 demonstrate that marine birds show differing spatial and temporal patterns of abundance and behaviour within high-energy marine environments. By using shore-based vantage point methods in three high-energy areas in the Northern Isles, I explore some variations in vantage point methods. Chapter 5 provides the most extensive and up-to-date review of diving and foraging behaviour for UK marine bird species. I identify knowledge gaps including the species and parameters for which there are a paucity of studies. I demonstrate a more robust approach to data collection in these high-energy tidal stream, which can shed insight on how they are being used by marine birds, while the diving and foraging behaviour synthesis provided many of the values that are required to undertake underwater collision risk modeling. It is hoped that developers can adopt the approaches identified here, which will improve the accuracy of their assessments and improve monitoring of any potential impacts.

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This thesis has been written in loving memory of my beloved best friend, devoted field assistant and foot-warming study buddy, Rissa. I finally did it, boy.

Rissa Robbins - 2006-2015

“The winds, the sea, and the moving tides are what they are. If there is wonder and beauty and majesty in them, science will discover these qualities. If they are not there, science cannot create them.”

Rachel Carson (1907-1964)

Author's Declaration

I declare that the work described in this thesis is of my own composition and has been carried out entirely by myself unless otherwise cited or acknowledged. No part of this thesis has been submitted for any other degree or qualification.

Alexandra Robbins
University of Glasgow, July 2017

Chapter 1: General Introduction

1.1 Marine renewable energy

It has been widely acknowledged that a global change in energy production, from fossil fuels to renewable sources, is required in order to reduce carbon dioxide outputs and help mitigate anthropogenic climate change (e.g. IPCC 2011; Rosenzweig et al. 2008). Climate change has been recognised as an internationally important concern and contentious political issue, with much debate over predictions of global temperature rises and climatic variability. Consequently, in 2009 the Renewables Directive set targets for EU member states to reduce its carbon emissions by 20% and to obtain 20% of energy from renewable sources by 2020 (European Parliament and Council 2009). The Scottish Government have increased their original target of 50% to 100% of electricity generation from renewable sources, also by 2020 (The Scottish Government 2013), with the expectation that one third of this target will be sourced through marine renewable energy schemes.

The key marine energy resources are tidal currents, wave, and offshore wind. The UK is recognised as having one of the largest practical marine energy resources in Europe (Pelc and Fujita 2002; Portman 2010), with an estimated 10% and 25% of Europe’s wave and tidal power (respectively) (Marine Scotland 2010). This has led to a strong emphasis on development and growth within the marine renewable sector, with Scotland at the forefront. The offshore wind industry has developed ahead of “wet renewables” (wave and tidal stream energy schemes), as the latter are still emerging industries, with many technologies in the early stages of development (Marine Scotland 2010). There are currently few marine renewable energy devices (MREDs) deployed in waters around the UK, apart from test sites such as the European Marine Energy Test Centre (EMEC) in Orkney. At the time of writing, there is one operational offshore wind farm (Robin Rigg in the Solway Firth) and three consented offshore wind developments in or overlapping Scottish Terrestrial Waters (STW), with a further two at application stage and one at scoping. There are also four consented offshore wind developments that have been challenged and are under appeal. Two tidal stream energy schemes have been consented in STW, with a further two at application stage and one at scoping. Unfortunately, the key wave energy developments that have been consented or are near application have stalled due to companies entering administration (Marine Scotland 2016).

The principle behind tidal stream energy generators is akin to that of a wind turbine, i.e. converting the energy in the tidal current velocity akin to how a wind turbine uses airflow (Fraenkel 2006). However, there is far greater variation in design and technologies used to harness the tidal flow energy compared with offshore wind; from horizontal axis and vertical axis turbines to venturi effect devices, oscillating hydrofoils and tidal kites (Fig. 1.1). The flow of the currents can be restricted by land and seabed topography e.g. around peninsulas and within channels and sounds, where water is pushed through and accelerated, increasing the kinetic energy (Fraenkel 2006), making these ideal locations for maximising tidal stream resources. Optimal site requirements for tidal turbines are more specific than either wave or wind devices: tidal stream energy technologies require peak spring tidal current velocities that are greater than 2.5ms^{-1} (Fraenkel 2006) and most devices require water depths of between 20-40m but some developers have noted depths of 70m (Denny 2009; Roberts et al. 2016).

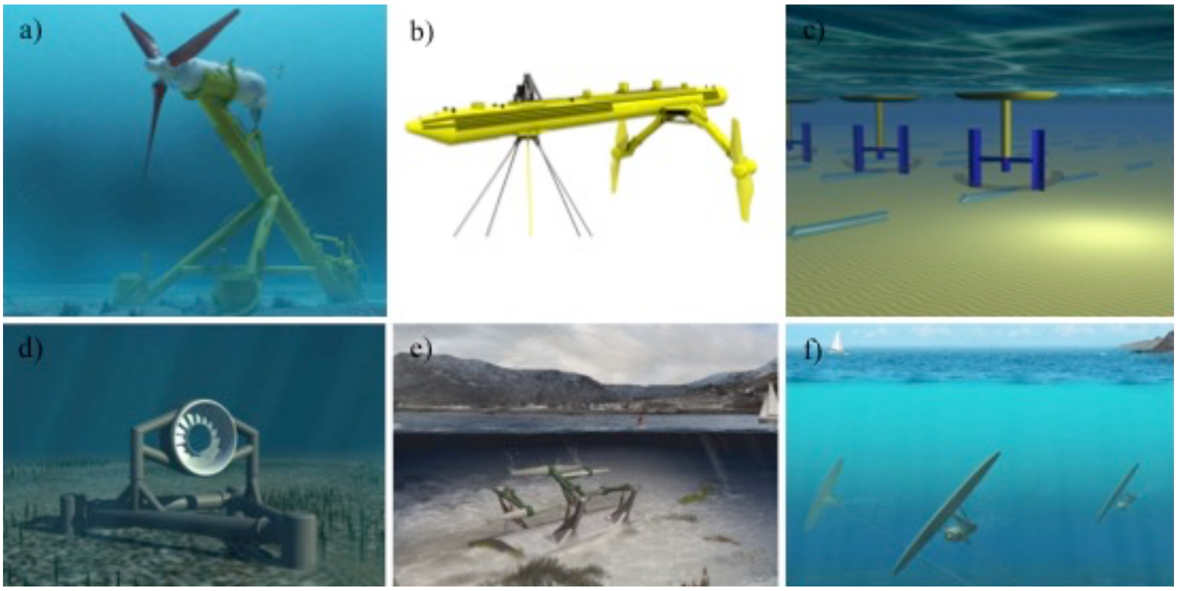


Figure 1.1 Examples of tidal stream converters with commercial example given in brackets: a) horizontal-axis turbine (Andritz Hydro Hammerfest: http://www.andritzhydrohammerfest.co.uk); b) floating horizontal-axis turbine (Scotrenewables Tidal Power: http://www.scotrenewables.com); c) vertical-axis turbine (http://www.aquaret.com/images/); d) venturi effect (OpenHydro: http://www.openhydro.com); e) oscillating hydrofoils (Pulse Tidal: http://www.pulsetidal.com); f) tidal kites (Minesto: http://minesto.com)

Wave energy uses the movement of the water near the surface of the sea, formed by winds moving across the surface. Waves can vary in both height and the time between peaks (period), which means different waves contain varying amounts of energy. Wave energy converters can either be located offshore, nearshore or fixed to, or embedded into the shoreline (Clément et al. 2002). Like tidal-stream devices, there is a plethora of wave energy converter designs and technologies that have been developed, the main groupings include attenuators, oscillating water column (OWC) devices, oscillating wave surge converters (OWSC), overtopping devices, point absorbers and submerged pressure differentials (EMEC 2016a) (Fig 1.2). Offshore devices are likely to require water depth of up to 100m (Falcão 2010). Both wave and tidal stream schemes, like offshore wind farms, are likely to include a number of devices to maximise the generation from any location, forming an array.

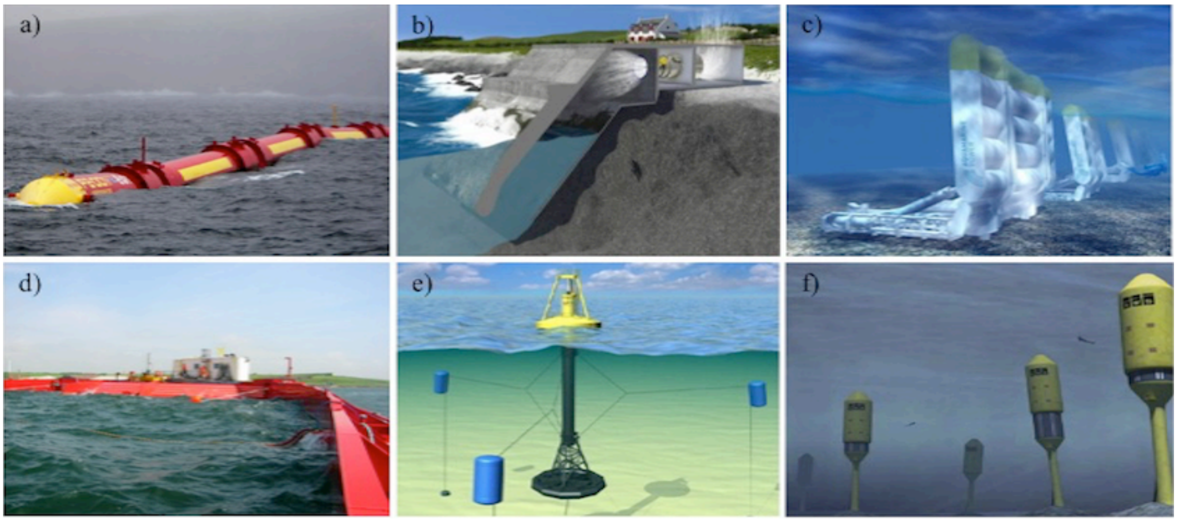


Figure 1.2 Examples of wave energy converters with commercial example given in brackets: a) attenuator (Pelamis: http://www.emec.org.uk/about-us/wave-clients/pelamis-wave-power/); b) OWC (Voith Hydro WaveGen, LIMPET: http://voith.com/); c) OWSC (Aquamarine Power, Oyster: http://www.aquamarinepower.com); d) overtopping device (Wave Dragon: http://www.wavedragon.net); e) point absorber (Ocean Power Technologies, Power Buoy: http://www.oceanpowertechnologies.com); f) submerged pressure differentials (Archimedes Wave Swing: http://awsocean.com)

Marine renewable energy schemes are subject to consenting and licensing processes under both European and national legislation. Developers for any marine renewable scheme in Scotland need to obtain a Marine Licence (under the Marine (Scotland) Act 2010) and a Section 36 consent (under the Electricity Act 1989). They therefore need to undertake Environmental Impact Assessments (EIA) (required under Section 36 of the Electricity Act 1989) and a Habitats Regulation Appraisal (HRA) (which may be required under the Conservation (Natural Habitats, &c.) Regulations 1994), to assess interactions and potential impacts of the development on the environment. However, for developing industries, such as both wave and tidal energy, potential impacts can remain largely unknown until there have been enough deployments to study, and knowledge gaps can present obstacles to the consenting process. Therefore there is an urgent need for applied ecological research to address policy needs and key consenting issues (e.g. Maclean et al. 2014).

Undertaking environmental impact assessments for marine renewable energy schemes is a complex process, in part due to the stochasticity within these high-energy marine environments and also the difficulties in measuring, recording and analysing both bird usage and environmental variables of these areas without introducing bias. Inaccuracies in data collection can be introduced by observer and detection biases, such as identification of species, availability of the species (pertinent to diving species) and distance decay; environmental biases can include collecting data in favourable weather conditions, e.g. no glare, low sea state; as well as temporal biases such as ease of collecting data in slower tidal states, or particular times of day and season. Whereas uncertainty or imprecision can be introduced randomly through the natural variability of ecological system being studied or systemically as a function of how we measure and analyse the system being studied based on our perception of it (Masden et al. 2015). These can contribute towards the variation in quality and quantity of information provided within assessments and the inherent uncertainty that enters this process through all stages, from data collection to decision-making (Masden et al. 2015). We therefore strive for *robustness* within these assessments but as Masden et al. 2015 found with the term “uncertainty” this can be defined in different ways, particularly in a subject area that overlaps academia, industry and policy. *Robustness* can refer to a dataset, a statistical approach and also a methodological approach: Robust statistics are statistical tests that perform well even if assumptions are broken, e.g. non-parametric tests. A robust dataset has a good signal to noise ratio and even under different tests, a pattern should be detected if it is present. Collection of robust data requires clarity and understanding on sampling methods, outliers, etc. However, the marine system in which we are required to collect data for marine renewable energy scheme assessments is highly stochastic resulting in highly variable quality of data, which inevitably fails to encompass the multitude of environmental variables affecting the dataset. Therefore decision makers require a *robust approach* to the assessment process, i.e. they need to be systematic and transparent in articulating the robustness and issues around the process from collection to analysis, enabling variable quality of data to be used but in such a way that comparisons, and indeed cumulative assessments, can be undertaken. Similarly, policy makers also require robust and transparent policies to aid decision makers. It is crucial in this developing field that methods of data collection are transparent in assumptions and strive to overcome variability arising from data collection.

1.2 Anthropogenic pressures on seabirds

Scotland, as well as having some of the best marine energy resources, also holds internationally important numbers of breeding seabirds (Mitchell et al. 2004). However, around the UK, particularly the North Sea, seabird populations are already facing a number of threats, with sustained declines seen in a number of species over the last 20 years (Foster and Marrs 2012). Seabirds as apex predators, have often been used as indicators of environmental change within marine ecosystems (e.g. Cairns, 1987; Furness and Camphuysen 1997; Piatt et al. 2007a;b). They are typically long-lived, with low birth rates and delayed maturity, and so consequently their populations are more susceptible to changes in adult survival rates than breeding productivity (Gaston, 2004). Studies have identified that changes in food availability (Frederiksen et al. 2004; Furness 2007; Votier et al. 2004; 2010), predation (Suryan et al. 2006; Votier et al. 2008a), climate (Votier et al. 2005; Frederiksen et al. 2008; Rolland et al. 2008; Grémillet and Boulinier, 2009) and pollutants (Votier et al. 2005; 2008b) can affect seabird productivity, and behaviour. In the UK, seabirds have been severely impacted by overfishing, particularly of lesser sandeels, *Ammodytes marinus*; the main prey species for many UK seabirds. More recently, rapidly rising sea temperatures have been linked to reduced sandeel recruitment (van Deurs et al. 2009), which would further depreciate seabirds' food supplies; and also extreme weather events have resulted in mass mortalities or 'wrecks' of seabirds (Frederiksen et al. 2008).

1.3 Impacts of wet renewables on seabirds

Wave energy devices and tidal stream turbines have the potential to place a new anthropogenic pressure on already declining seabird populations, with many licensing areas in close proximity to internationally important seabird colonies. Wet renewables are predicted to impact seabird populations through collision, disturbance, habitat loss and changes to food availability (e.g. Inger et al. 2009; Grecian et al. 2010; Langton et al. 2011). However, with the industry in its infancy, few devices have been deployed to enable monitoring of impacts and there have been few studies of the ecological implications and magnitude of any impacts to these populations, with little empirical data to quantify any impacts (Wilson et al. 2007; Witt et al. 2012).

It is important to identify species that are likely to be affected by wet renewables, providing focus for research and assessments for developments; but also those species that are unlikely to be affected, to ensure they can be scoped out of assessments in a scientifically robust way (Furness et al. 2012). The foraging patterns, habitat preferences and seasonal distributions of a species are all likely to influence if and how seabirds might interact with devices. Unfortunately, these aspects of bird behaviour are poorly understood for high-energy systems, particularly in the UK, making it difficult to predict the consequences of installing renewables (Hunt et al. 1999). Furness et al. (2012) explored peer-reviewed literature specifically on drowning risk, diving depths, benthic foraging, use of tidal race for foraging, feeding range, disturbance by traffic, and habitat specialization to identify which Scottish seabird populations were likely to be most vulnerable to tidal-stream developments. Risk of collision mortality, exclusion from foraging habitat, disturbance by structures and by ship traffic, flexibility in habitat use and benefit from roost platforms and fish attraction devices or biofouling were then considered as the key factors in determining vulnerability to wave-energy installations. Furness et al. (2012) identified black guillemot, razorbill, European shag, common guillemot, great cormorant, divers and Atlantic puffin as the species most vulnerable to adverse impacts from tidal-stream devices, while divers were identified as most vulnerable to wave developments. Wave-energy devices were concluded to represent less of a hazard than tidal-stream turbines.

There are a number of different parts that make up the overall functioning device of marine renewable installations, (e.g. the moorings, surface and underwater structures and onshore infrastructure), which individually can present a novelty and/or obstacle within the birds' environment. Some wet marine renewables designs, in particular wave devices, extend above the surface of the water as well as below. However, the existing wave devices have considerably lower profiles than wind turbines, which is likely to lower the risk of mid-air collision (Michel et al. 2007). Marine birds are only likely to interact with sub-surface marine renewable devices while they are foraging (or undertaking avoidance actions from boats). Turbines are the one part of a renewable structure that are most commonly associated with negative impacts, namely collision (Hüppop et al. 2006). While parallels are made with both wind turbines and ship propellers, there are differences in the speed at which these turbines rotate i.e. tidal turbines will rotate more slowly (Fraenkel 2006), and the difference in medium (water and air) may alter hydrodynamic flows taking energy out of the current. It has also been suggested that seabirds may be swept through the blades of a tidal turbine due to the flow of water around the blades (Fraenkal 2006). However, as noted by Furness et al. (2012) this assumes birds move passively with the water flow instead of swimming against or across the tidal current. However, evidence suggests birds swim more actively within the water, with studies such as Heath et al. (2006) showing common eider normally dive against the current and surface upstream of their dive entry location. Marine birds are able to undertake avoidance measures to try and escape collision with renewable energy devices. For devices located on or above the water, they can employ similar tactics to those used avoiding wind farms by taking alternative flight routes and avoiding obstacles during poor visibility and during darkness (Desholm and Kahlert, 2005; Wilson et al. 2007). Fraenkel (2006) suggested that diving birds, that have fast bursts of speed, may be able to move out of the path of the blades of tidal turbines. However, diving birds' fastest swimming speeds are slower than the speed of the outer edge of turbine blades (Fraenkel, 2006). There is currently no empirical data that can be used to test these ideas.

Disturbance can lead to displacement, which can arise when birds actively avoid areas where marine renewable devices (or other man-made structures) are found, or if alterations to the hydrological process or environment make the location unsuitable for the animals' prey species (Langston and Pullan 2003; Gill, 2005; Larsen & Guillemette, 2007). Displacement has been seen around wind farms, as a number of bird species have been observed exhibiting avoidance behaviours towards the wind turbines (Desholm and Kahlert, 2005). Such avoidance behaviour can create a barrier to movement, which discourages them from using the surrounding area, forcing them to navigate around the development (Desholm, 2003). Evidence from wind farms, suggest this could be 2-4km from the array site (Dierschke et al. 2016). While this will reduce the risk of negative interactions, i.e. collision, it also limits the habitat available for them to use (e.g. Larsen & Guillemette, 2007; Inger et al. 2009). Many breeding marine birds are confined to centrally placed foraging areas, therefore marine renewable devices may pose a barrier to movement if they are placed on regular commuting routes between breeding and feeding grounds (Langston & Pullan 2003, Desholm & Kahlert 2005, Masden et al. 2010b). Masden et al. (2009) highlighted that seabirds avoiding offshore wind farms may have to fly increased distances to reach foraging grounds, and if this additional energetic cost could not be met, a reduction in condition of the breeding bird and/or its offspring could follow. However, Masden et al. (2009) calculated that to achieve body mass loss, such avoidance behaviours would need to be repeated regularly. Grecian et al. (2010)

considered that the lower surface profile of most wet renewable devices is likely to result in a negligible impact, but called for more research into the re-directional effect of devices with differing heights.

Habitat will invariably be lost as a result of marine renewable energy devices being installed, however the extent of this will vary depending on the siting, design of devices, and scale of arrays and associated works (e.g. cable laying) (Inger et al. 2009). Mueller and Wallace (2008) identified that wave energy converters will have a smaller foundation footprint on the seabed than tidal devices. However, Inger et al. (2009) also highlighted that there are potentially beneficial impacts as well if exclusion of fishing within arrays creates *de facto* marine protected areas.

There are a number of ways in which food availability could be altered by localised habitat changes around marine renewable arrays. Changes in turbulence may lead to stratification and movement of prey within the water column (Grecian et al. 2010); or if the devices are placed in an area with little or no hard substrate, e.g. sandy bottoms, the infrastructure on the seabed can create an artificial reef (Whitmarsh et al. 2008; Boehlert and Gill, 2010; Grecian et al. 2010). This could provide anchorage for bivalves (Langhamer et al. 2009), providing additional food resources benefitting seaduck species, for example. The renewable devices may also act as an attractant for fish (e.g. fish aggregation devices or FADs). Alternatively, devices may have negative effects on fish populations. Migratory fish were found to avoid the Bay of Fundy tidal turbines area, which can in turn have a negative impact on birds reliant on the fish population (Dadswell and Rulifson 1994). Grecian et al. (2010) also suggested that fish mortality through collision risk or entrapment may benefit scavenging birds in the short term, however if this was to have a negative impact on the fish population, then there could be long-term consequences, in relation to food supplies.

1.4 Thesis motivation and aims

The concept for this thesis was initially developed while working for Scottish Natural Heritage on Noss National Nature Reserve in 2010 with my colleagues and University of Glasgow supervisors. The marine renewable energy industry was undergoing rapid growth, particularly in the Northern Isles, but very little was understood about how marine birds use high-energy areas and there were also no standardised methods for assessing potential impacts or even quantifying usage of these areas. In the intervening years research in this area has grown, with progress towards both predicting and monitoring impacts of wet renewable energy schemes, however, my initial research aims are still just as relevant today. The two aims of thesis are:

1. To expand our knowledge base relating to our understanding of seabird ecology in high-energy marine environments, aiding the assessment of the impacts of marine renewables on seabirds. As UK seabirds have been rarely studied in this environment, I intend to explore spatial and temporal use of these areas, including potential habitat drivers.

2. To increase our understanding of how land-based bird surveys can be used to assess wet renewable energy schemes, in particular to help develop a robust approach to undertaking environmental impact assessments. The thesis achieves this by using shore-based vantage point methods to address some methodological biases, as well as identify spatial and temporal patterns of use and behaviour in these high-energy environments.

1.5 Thesis structure

This thesis contains four data chapters, which have been written as stand alone chapters. Some information within these chapters may be duplicated. Within each chapter I will identify which of the thesis research aims and key questions will be addressed.

Chapters 2 and 3 explore two existing long-term wildlife monitoring datasets from wet renewable test sites in Orkney (see Fig. 1.3). The two test sites comprise a wave energy and a tidal-stream site, with slightly different approaches to monitoring. Data were collected on behalf of EMEC at the two test sites and I undertook the analyses. This work was undertaken during the early part of my PhD and was published as two separate SNH commissioned reports. The reports have been amended slightly (e.g. excluding marine mammal results) for inclusion within this thesis. Initially it was hoped to explore changes in usage (at both sites) with device presence, however this was not possible due to commercial sensitivities. Instead, these chapters aim to identify relationships between the more frequently observed bird species' site usage and habitat and environmental variables. Some of the methods used in these chapters are now considered outdated, however analysis of these data and review of the survey methods proved to be useful in development of the survey design for my own fieldwork.

In **Chapter 4**, the thesis moves on to focus on the marine bird usage of a high-energy tidal stream in Shetland (see Fig. 1.3). I identify habitat preferences, foraging patterns and behaviour within the sound for key diving bird species, and explore implications for vulnerability to encountering tidal arrays. I also identify a potential solution to a methodological issue that arises through surveying birds across tidal currents.

In **Chapter 5** I undertake a review of published literature on foraging and diving behaviour for twenty-two diving marine bird species. The aim of this review is to provide values for parameters related to bird's foraging and diving behaviour for use within underwater collision risk modelling. This work provides a key resource to the consenting process as it can be used in the assessment of environmental impacts of marine renewable developments.

During this PhD I have also contributed to two papers as a named co-author (Furness et al. 2012 and Waggitt et al. 2017). The first uses data from scientific literature on seabird ecology to identify which species are most vulnerable to tidal stream turbines and wave energy devices. The second uses six shore-based datasets from across tidal streams in Scotland, including a subset of data collected as part of my Bluemull Sound fieldwork, to compare variability in the use of tidal streams by black guillemot and European shag. Both paper and manuscript are included in Appendix 6.

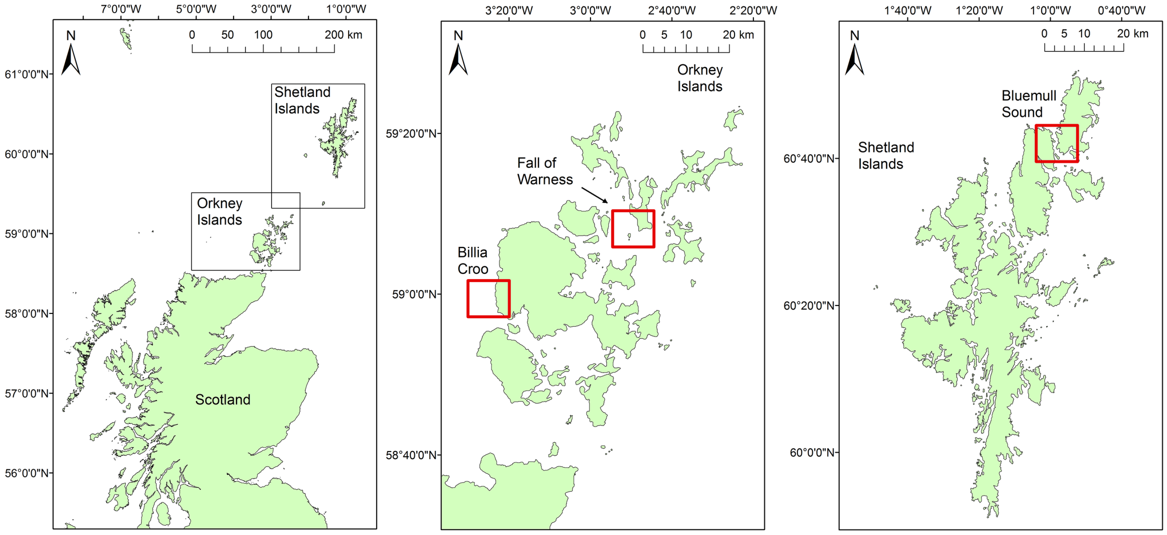


Figure 1.3 An overview of the three Northern Isles study sites used in this thesis. Chapter 2 focuses on the EMEC tidal-stream test site at Fall of Warness, Orkney; Chapter 3 on the EMEC wave-energy test site at Billia Croo, Orkney; and Chapter 4 on Bluemull Sound, Shetland.

Chapter 2: Monitoring marine bird use of a tidal stream test site: an Orkney case study

This chapter has been published as a commissioned report to Scottish Natural Heritage as ‘Robbins, A. 2012. Analysis of bird and marine mammal data for the Fall of Warness Tidal Test Site, Orkney. *Scottish Natural Heritage Commissioned Report No. 614*’.

This chapter comprises data owned and collected for EMEC as part of their wildlife monitoring programme at the Fall of Warness. I have undertaken all the analyses of these data presented within this chapter. Marine mammal data included in the original analysis have been omitted and some of the text and figures has been amended to fit within the context of this thesis.

Supplementary materials for this chapter are in Appendix 1.

Abstract

The marine renewable energy sectors, including both wave and tidal-stream energy are in their infancy with few devices deployed globally. The European Marine Energy Centre (EMEC) provides one of the first areas in the world for these devices to be deployed and tested. It also provides an invaluable opportunity to collect comparatively long-term data on the most common bird species in potential development areas (e.g., if they interact with or are otherwise affected by the test devices, and whether such interactions would be detrimental to them). Therefore, this study aimed to identify the distribution, abundance and species composition of marine birds using the EMEC Fall of Warness tidal test site, and explore relationships between the more frequently observed bird species' site usage and habitat and environmental variables. This chapter analyses observations of the eight most commonly encountered diving bird species at Fall of Warness conducted from 2005 to 2011.

All species showed seasonal variation, while daily temporal variation was exhibited by all species except gannets, common guillemots and razorbills. Species varied in their use of the environment and some were influenced by tidal state and wind strength and direction. Species more commonly observed using the pelagic environment and waters between 21-30m in depth included red-throated divers, gannets, common guillemots, razorbills and puffins while eider, *Phalacrocorax spp.* and black guillemots preferred the shallower coastal environment where water was between 1 and 10m deep. Only half the species were influenced by tidal state and significant relationships were found between some weather variables and a few species, including common guillemots, *Phalacrocorax spp.* and black guillemots.

2.1 Introduction

The European Marine Energy Centre (EMEC) in Orkney, Scotland is the one of the first centres of its kind in the world, providing developers of both wave and tidal energy converters with purpose-built, accredited open-sea testing facilities (EMEC 2016b). These renewable energy arrays harness the power of waves and tidal streams to generate electricity. Orkney is renowned for its harsh marine conditions and the tidal test site, Fall of Warness, off the island of Eday, has high velocity marine currents reaching nearly $4 ms^{-1}$ during spring peaks, making it an optimal location to trial these energy devices (EMEC 2016b).

Tidal-stream energy arrays are typically constructed in near-shore areas that may be important to seabirds and potentially impact them via collision, disturbance, changes to food availability, and habitat loss (e.g. Inger et al. 2009; Grecian et al. 2010; Langton et al. 2011). These impacts are known to influence the manner in which seabirds utilise an area and could limit or prevent their access to necessary resources. Thus, there is a legal requirement to assess potential impacts of these developments by collecting data on the most common bird species in potential development areas. These data are used to determine how seabirds use the marine environment and if they interact with, or are otherwise affected by, the proposed development. In this way we can understand whether such interactions are detrimental to marine bird populations and what preventative measures could be put in place to mitigate any deleterious effects.

There are internationally important numbers of breeding seabirds and aggregations of other marine birds on or around the Orkney Islands (Mitchell et al. 2004; Forrester et al. 2007). These often overlap with many high-energy areas that are also suitable for marine renewable energy development. The Fall of Warness test centre is located to the west of the island of Eday, and a nearby, shore-based vantage point 50 m above the test site, provides an ideal location to determine whether seabirds interact with these tidal stream devices. This vantage point should also enable monitoring of any interactions between marine birds and tidal energy devices. Until now, there has been limited analysis of the five-year wildlife monitoring dataset, and marine bird species that use the immediate proximity around the test site.

In this chapter I aim to obtain baseline data on the spatial and temporal distribution, abundance and species composition of seabirds that use the EMEC Fall of Warness tidal test site; identify those most commonly occurring; and explore relationships between the more frequently observed bird species' site usage patterns and habitat and environmental variables such as water depth, wind strength or direction, tidal state and cloud cover. In addition, I provide recommendations on improving wildlife monitoring protocols and data management for the Fall of Warness tidal test site, as the quality of the monitoring protocols used up to now by the ornithologists working under contract to EMEC has not been subject to critical review. This chapter, alongside chapter 3, will set the foundation and provide a better understanding of appropriate methodologies, wildlife monitoring protocols and data management that can be used in future studies determining the level and type of interaction between seabirds and the tidal energy systems.

2.2 Methods

2.2.1 Data collection

Land-based vantage point observations take place from a trailer shelter on Ward Hill, Eday, Orkney (59˚08.975'N, 002˚47.396'W), approximately 50m above sea level. These data were collected on behalf of EMEC by two experienced surveyors from Eday. The survey area was defined by grid squares approximately 500m x 500m (Fig. 2.1). An additional row (-1) was added to the north end of this grid in August 2005 to accommodate the most northerly berth. The observations for birds commenced at the Fall of Warness on 11th July 2005, with a four-hour watch format, 5 days per week (i.e. approximately 80 hours of observation per month) and concluded in December 2010. The rota was designed to ensure relatively even coverage across daylight and tidal states. Where watches were unable to be carried out due to weather they were typically undertaken at the next weekend or following opportunity.

The observers scanned the site using a telescope (Opticron GS815) using a 20x magnification to detect birds on the surface of the water. When necessary for species identification the magnification was increased up to 60x. The area was scanned in a pre-defined manner grid square, recording species within the cell. The geographical locations of sightings were estimated by overlaying the associated grid over a map.

Birds were only recorded as sightings when they were on the water or hovering directly above. Groups of animals were defined as all animals within approximately 100m of each other. If groups of animals were recorded then the location was based on the centre of the group. Observers recorded bird behaviour as:

Diving from flight	One or more birds diving underwater from a hovering or flying position.
Diving from water	One or more birds diving underwater from a position on the water surface.
Swimming at surface	The birds are making progress at the surface.
Stationary at the surface	The birds are stationary at the surface.

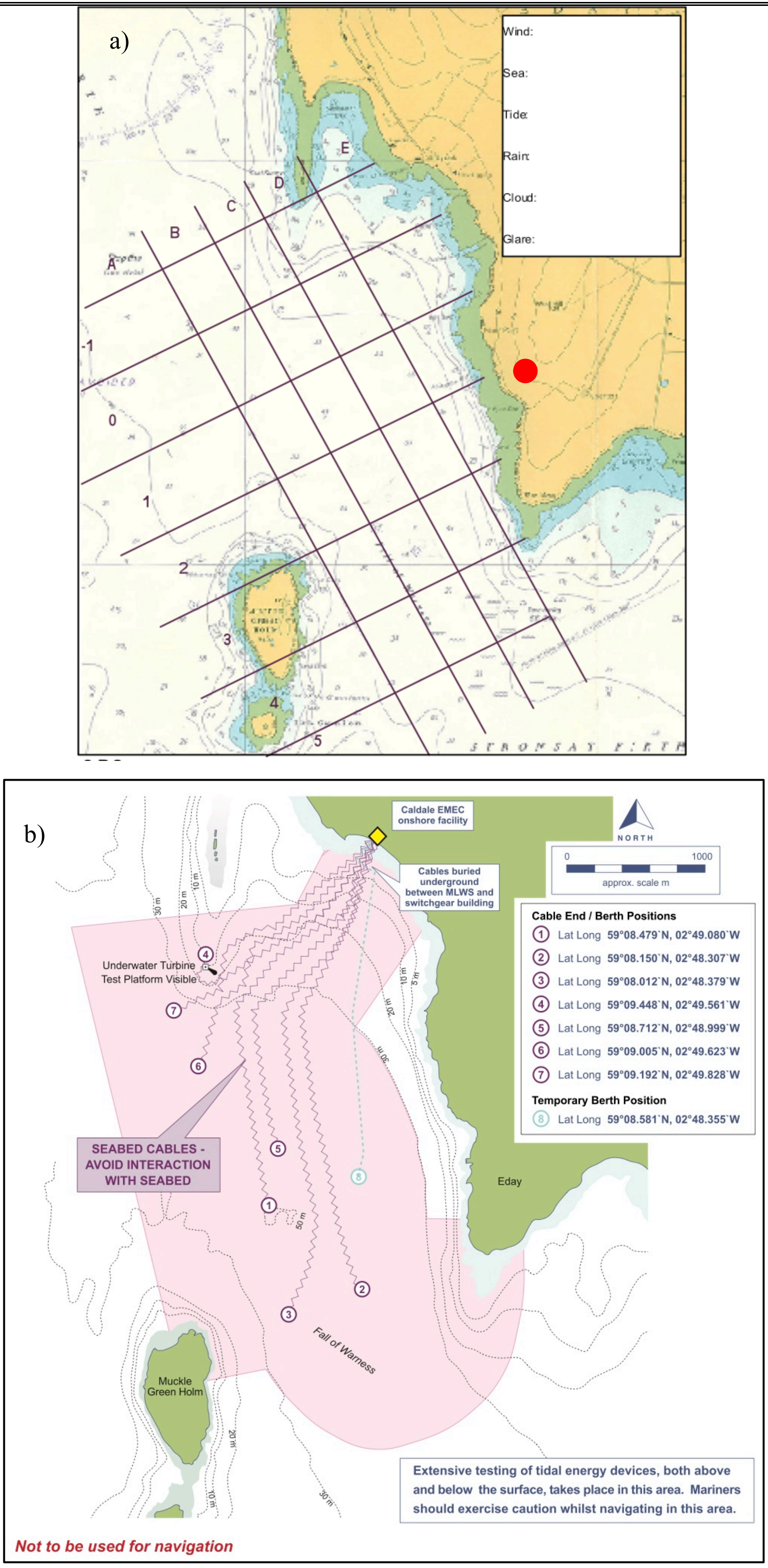
2.2.2 Data Preprocessing

These data required preparation prior to analysis, which included:

- Alteration of misspelled species names, codes and other categorical variables.
- Alteration of reversed grid squares (letters/numbers), removal of misidentified grid squares.
- Ensuring consistency in naming of categorical variable levels.

Data and analytical limitations:

- Only actively foraging birds (i.e. birds interacting with the water surface) were recorded at the Fall of Warness. Due to the lack of variation in behavioural observations, these data could not be used to analyse behavioural disturbance.
- Due to the commercial sensitivity of the test site devices, data on when these devices were present and operational were not available at the time of analysis and could not be included.
- Due to both the scale of the grid system used for the wildlife monitoring and the use of shore-based vantage point surveys, too many assumptions were violated to undertake a distance analysis.
- Due to the number of bird sightings exceeding the Excel spreadsheet row limit, only data between 11th July 2005 and 19th December 2010 were included in the bird analyses. 68 of the most recent observation days were not included in the analysis, however this was only 5% of observations for the 5-year dataset.



Cable End / Berth Positions

①	Lat Long	59°08.479`N, 02°49.080`W
②	Lat Long	59°08.150`N, 02°48.307`W
③	Lat Long	59°08.012`N, 02°48.379`W
④	Lat Long	59°09.448`N, 02°49.561`W
⑤	Lat Long	59°08.712`N, 02°48.999`W
⑥	Lat Long	59°09.005`N, 02°49.623`W
⑦	Lat Long	59°09.192`N, 02°49.828`W

Temporary Berth Position

⑧	Lat Long	59°08.581`N, 02°48.355`W

Figure 2.1 a) Map of the EMEC Fall of Warness tidal site, showing the vantage point location at Ward Hill (red dot) and study area extending from the shoreline to approximately 2km offshore and divided into grid squares of approximately 500m^2. Note: this figure is the datasheet for environmental variables and therefore the legend within the figure is for data collection purposes only. (Taken with permission from the EMEC Fall of Warness Observation Protocol). b) Map of the EMEC Fall of Warness tidal site, showing the berth positions. (Adapted from the EMEC Tidal Test Site Awareness Chart: http://www.emec.org.uk/facilities/tidal-test-site/)

Table 2.1 The total number of hours of survey that birds were observed in between 11th July 2005 and 19th December 2010.

Month	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**
January		12	22	16	24	38
February		16	26	26	31	48
March		23	35	26	28	52
April		37	29	26	46	52
May		20	35	37	36	28
June		32	35	39	28	42
July	24	47	36	34	33	30
August	18	36	27	29	31	47
September	18	38	27	23	32	49
October	17	32	32	31	55	45
November	22	25	17	25	47	38
December	11	16	21	23	38	20
Total	**110**	**334**	**342**	**335**	**429**	**489**

2.2.3 Data Analysis

All data analyses were completed using the statistical data package R, with relevant packages (R Development Core Team, 2011). The total survey effort (in hours) is summarised for the entire data set (refer to table 2.1). For all bird species with sufficient data, modelling of the effects of habitat and environmental variables was then undertaken.

2.2.3.1 Environmental Variables

In addition to the marine bird count data, a range of environmental variables were monitored during observations (Table 2.2).

The identity of each observer is an important element in modelling of observation data to account for variation between individuals because experience and skill can influence observations. Unfortunately, this information was not available for this study but should be obtained in future studies.

Table 2.2 Environment variables monitored at Fall of Warness tidal test site.

Environmental Variable	Description
Wind Direction	This was subdivided into the following six categories; "North", "South", "East", "West", "Variable" and "None". Due to the latter two categories this variable was not treated as a circular variable.
Wind Strength	This was defined by observations using the Beaufort scale descriptions.
Sea State	This was defined also by observations using the Beaufort scale descriptions.
Cloud Cover	This was recorded as a percentage.
Precipitation	A range of descriptive categories and values has been applied to this variable during data collection. Previous reports have analysed precipitation in the binary form of "dry" or "precipitation". The latter approach was adopted for this analysis, but, as mentioned in previous reports, an intermediate approach with a manageable number of categories may be more informative in any future analysis.
Tidal State	This was subdivided into "Ebb", "Flow" and "Slack".
Tidal Flow Speed[1]	Flow speed has been recorded with the following categories "Very Fast", "Fast" Moderate/Fast", Moderate", "Slow", "Slack", "High" and "Low".
Habitat Type	Limited data are available on the habitat types within the Fall of Warness monitoring site (Finn 2009) and are not available for every grid square. Distance for coast was not included as it confounds with the detectability of birds. It was therefore only possible to describe each cell as either "Coast" (if coastline was present in the cell) or "Pelagic" (if there was no coastline).
Depth	Depth data was only available from the MCA Bathymetry Survey in Finn (2009). The cell depths were grouped as "1 – 10m", "11 – 20m", "21 – 31m", "32 – 42m", and "43 – 54m".
Grid square	The study site is divided into rows (-1 to 5) and columns (A to E). Each grid square is 500x500m in size.
Season[1]	The months were grouped in to "Winter" (December, January and February); "Spring" (March, April and May); "Summer" (June, July and August) and "Autumn" (September, October and November).
Time of Day	This was recorded as the hour in which the observation occurred, i.e. 10:30 was "10" and 14:15 was "14" using GMT.

[1] Ordinal variables were treated as factors.

2.2.3.2 Co-linearity

Similar variables were tested for co-linearity using Variance Inflation Factors (VIF). Tide state, flow direction and flow speed were all found to be co-linear. Tide state was found to be the most representative variable and was used in the subsequent analyses. Wind strength was used as a proxy for sea state, the method for recording both was using the Beaufort scale, an empirical measurement, which is used to describe wind intensity based on observed sea conditions. Although, noting that both wind strength and tidal conditions affects sea state and that the numerical values between the two differ due to the lag effect between the wind strength increasing and the sea state increasing (Met Office, 2016).

2.2.3.3 Modelling

Modelling of marine bird abundance with the EMEC Fall of Warness tidal test site was achieved by using two extensions of generalized linear model ("GLM") techniques. The counts of each species were used as the response variable to investigate the influence of the different habitat and environmental conditions. The dataset was zero-truncated and therefore the analysis was of presence only data. The model interpretation therefore represents the impact of environmental covariates on the absolute number of birds. Another inherent issue with this dataset is the temporal auto-correlation, i.e. where counts that occur in consecutive hours or days may not be independent, which if unaccounted for may result in detecting an effect that does not exist. Therefore, generalized linear mixed models (hereafter "GLMM") were used with a Poisson error distribution and log link, and the inclusion of a random effect variable to allow for correlation between observations the same day and grid square.

The top five models were selected using the Analysis of Variance (ANOVA) function and comparing Akaike Information Criterion (AIC) values - the measure of the quality of the model. I compared the variables that were consistently selected in these top 5 models in order to best understand the variables that were most likely to be influencing results for each species. These variables were then used in the subsequent models. GLMM model validation was undertaken by plotting and reviewing the distribution of the selected models Pearson's residuals. In addition, another GLM extension, generalized estimation equations (hereafter "GEE") were also used to estimate robust standard errors, adjusted for temporal autocorrelation. The variables included in the GEE models were either those selected in best GLMM for any given species or a collection of the variables that occurred most frequently across the top 5 models. In this way I created the most robust predictions of marine bird species distributions and was able to determine the most influential environmental variables in the Fall of Warness test site area. Observations that occurred within the same day and grid square were assumed to have an AR1 autocorrelation (auto-regressive model of order 1), which models a residual at time t as a function of time $t-1$

along with noise, so the further the residuals are away from each other in time the lower their correlation. The results reported include the GEE model coefficient estimates and standard errors. The plots presented within this report incorporate GEE model coefficient estimates, the encounter rate (mean number of birds observed per hour) or percentage of overall observations. The higher model coefficient values represent a greater number of predicted birds.

2.3 Results

The top model results are presented for each of the foraging bird groups and species observed and pertinent figures. The top 5 GLMM models are included within Appendix 2.

2.3.1 Common Eider *Somateria mollissima*

A total of 66,254 common eider were observed between July 2005 and December 2010 at the Fall of Warness tidal test site. Eider numbers were found to be a function of season, time of day, tidal state, depth, habitat type and cloud cover (Table 2.3 and S.Table 1.1). Greater numbers of eiders were observed in the spring months and fewer in the summer (Table 2.3; Fig. 2.2).

Time of day was selected for in four out of the top five models (Table 2.4) and eiders were seen in greatest numbers at 09:00-10:00 (peak mean = 28 birds per hour; Fig. 2.4), after which the hourly encounter rate (mean number of birds per hour) decreased. The GEE and GLMM coefficient estimates both indicate that eiders were observed significantly more frequently during flooding tides and slack tides (Fig. 2.3).

The depth of the grid squares and habitat type (pelagic v. coastal) were also selected for, in all of the top models. Eiders were most commonly seen in 1-10m depth (mean = 53.53; Fig. 2.5) and of the 1028 days eiders were observed at the Fall of Warness, they were seen in coastal grid squares on 1011 days and in pelagic grid squares on 359 days.

Table 2.3: Parameter estimates, standard errors, probability values for GEE investigating eider counts as a function of season, time of day, tide state, depth, habitat type and cloud cover.

	Estimate	**Std. error**	**Wald**	**Pr (>|W|)**	**Signif.**[1]
Intercept	2.1308	0.1200	5773 287.02	< 2e-16	***
Spring	0.6496	0.0535	147.23	< 2e-16	***
Summer	-0.9005	0.0602	224.12	< 2e-16	***
Autumn	-0.1703	0.0453	14.17	0.00017	***
Time of day	0.0174	0.0087	3.97	0.04643	*
Flood	0.5004	0.0504	98.55	< 2e-16	***
Low	-0.9914	0.4485	4.89	0.02708	*
Slack	0.4599	0.0642	51.3	7.90E-13	***
Depth	-0.0541	0.0021	640.96	< 2e-16	***
Pelagic	0.6901	0.0673	105.17	< 2e-16	***
Cloud cover	0.0035	0.0008	18.97	1.30E-05	***

[1] Significance codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' '

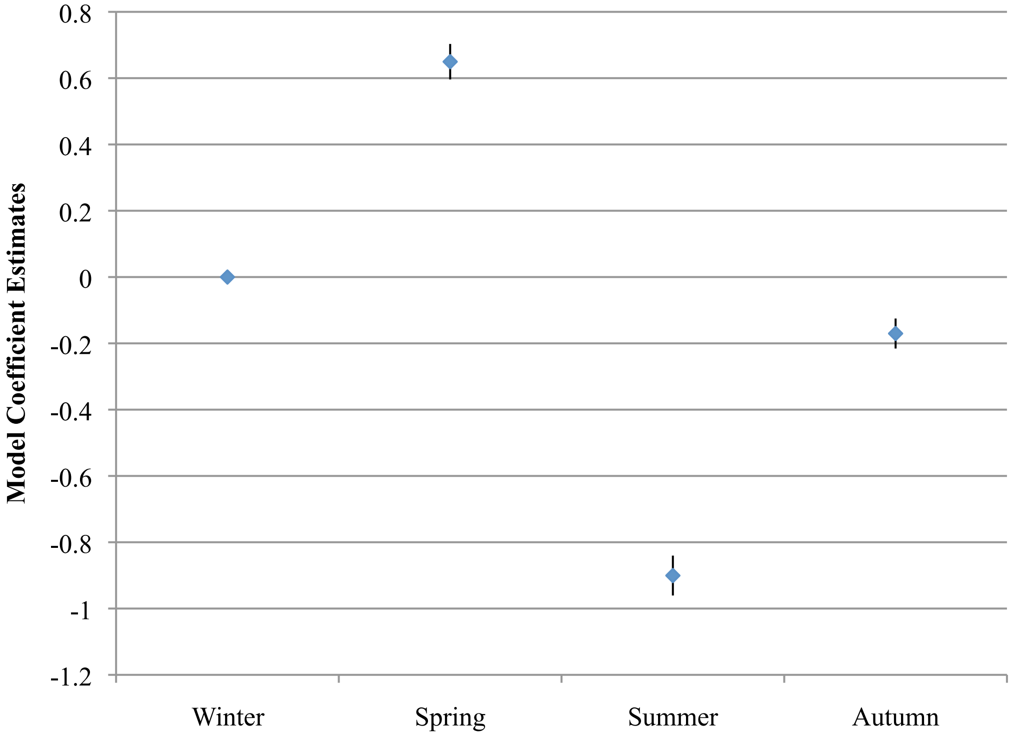


Figure 2.2: GEE coefficient estimates (and standard errors) for common eiders observed by season at the Fall of Warness.

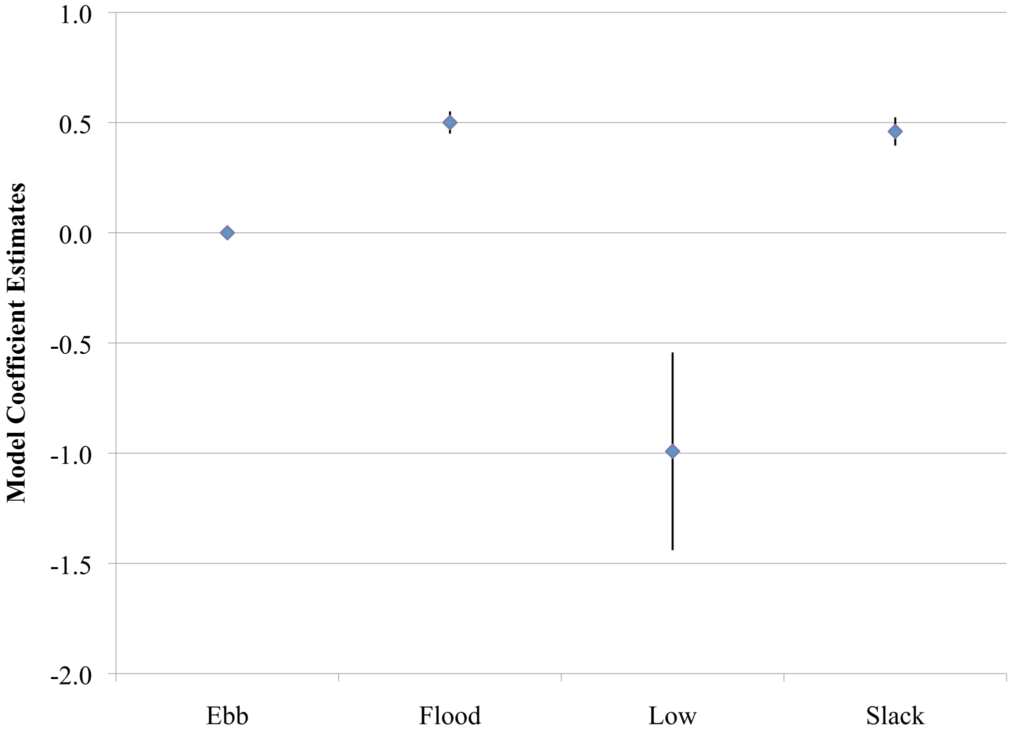


Figure 2.3: GEE coefficient estimates (and standard errors) for common eiders observed by tidal state at the Fall of Warness.

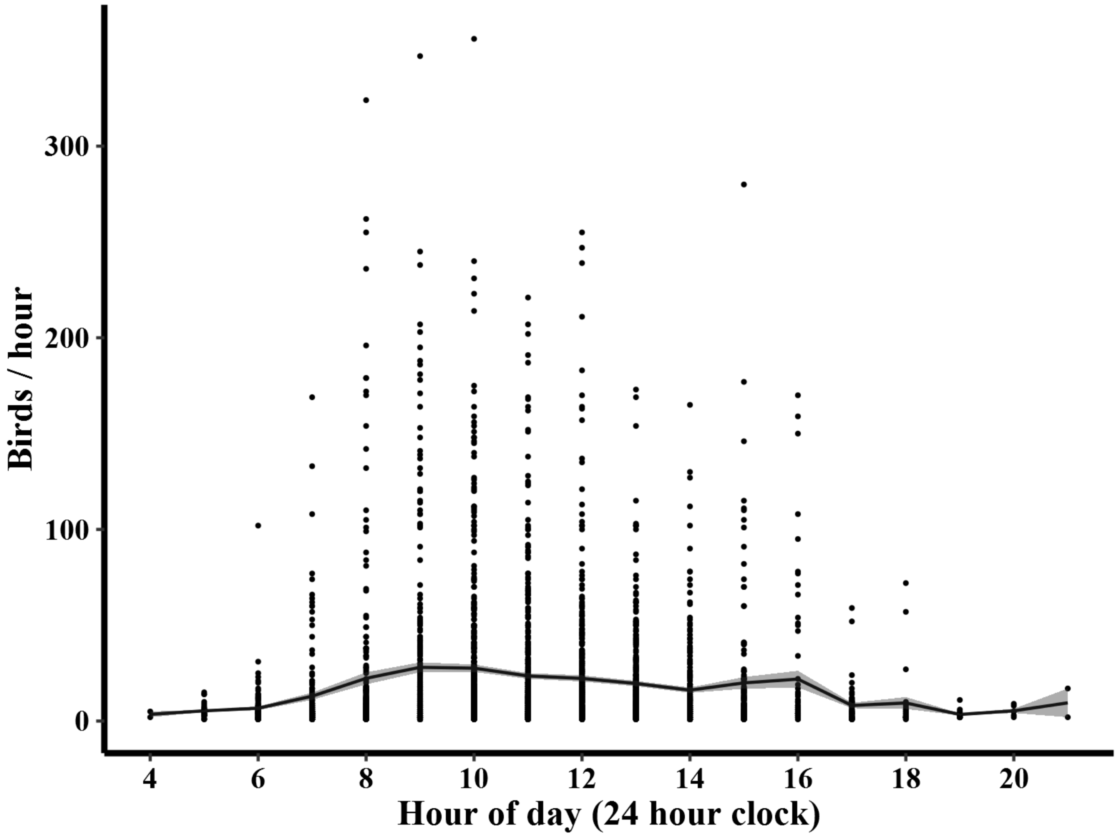


Figure 2.4: Mean number of common eider observed per hour, throughout the day at the Fall of Warness, with raw data overlaid and standard error around the mean shown by the grey area.

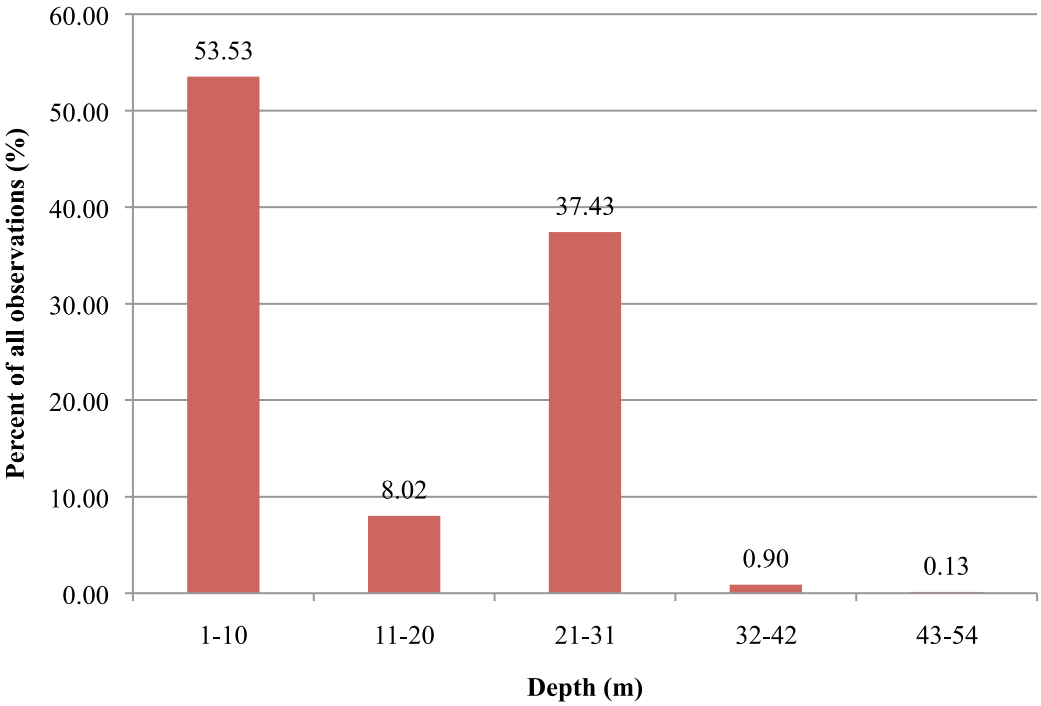


Figure 2.5: The proportion of observations common eider were observed at different depths, at the Fall of Warness.

2.3.2 Divers *Gavia spp.*

Of the 4,638 divers observed between July 2005 and December 2010, <70% were red-throated divers (*Gavia stellata*), 24% were great northern divers (*G. immer*) and >1% were black-throated divers (*G. arctica*), the remaining 5% were unidentified to species (Fig. 2.6). Of the top five GLMM models exploring the influence of environmental variables on numbers of divers observed, season, depth, wind strength and tide state were most frequently selected, however the top model also included time of day (S.Table 1.2). The GEE model also identified season, time of day, depth and wind strength to be significant (Table 2.4). The GEE model estimated over dispersion (scale parameter) to be 0.52 and correlation between observations on the same day and grid squares to be 0.55.

Figure 2.7 indicates that higher model coefficients in summer, with lower in winter and spring; while figure 2.8 highlights that a greater proportion of diver observations occurred in autumn (38.98%) compared with only 8.45% in the summer.

Divers were found to show several temporal patterns. They were observed more frequently in the early morning and then again in the evening: the mean number of divers per hour increases and peaks at 06:00 (2 divers/hour) before decreasing during the day and peaking again at 18:00 (3 divers/hour; Fig. 2.10). The encounter rate of divers per observation was greater during slack tides, compared with ebb and flood tides (Table 2.4; Fig. 2.9).

Greater than 55% of observations recorded divers in depths of 21-31m (Fig. 2.11). The mean number of divers encountered per hour decreases as wind strength increases from Force 0 to 5, with the mean hourly encounter rate peak at Force 0 (3 divers/hour) (Fig. 2.12).

Table 2.4: Parameter estimates, standard errors, probability values for GEE investigating diver counts as a function of season, time of day, depth, wind strength and tide state.

	Estimate	**Std. error**	**Wald**	**Pr (>|W|)**	**Signif.**
(Intercept)	0.363679	0.067786	28.78	8.10E-08	***
Spring	0.046002	0.028213	2.66	0.103	
Summer	0.137771	0.040708	11.45	0.00071	***
Autumn	0.085281	0.02753	9.6	0.00195	**
Time of day	0.013138	0.004761	7.62	0.00579	**
Depth	-0.004374	0.000948	21.28	4.00E-06	***
Wind strength	-0.045514	0.012141	14.05	0.00018	***
Flood	-0.038923	0.023694	2.7	0.10044	
Slack	0.041555	0.030931	1.8	0.17912	

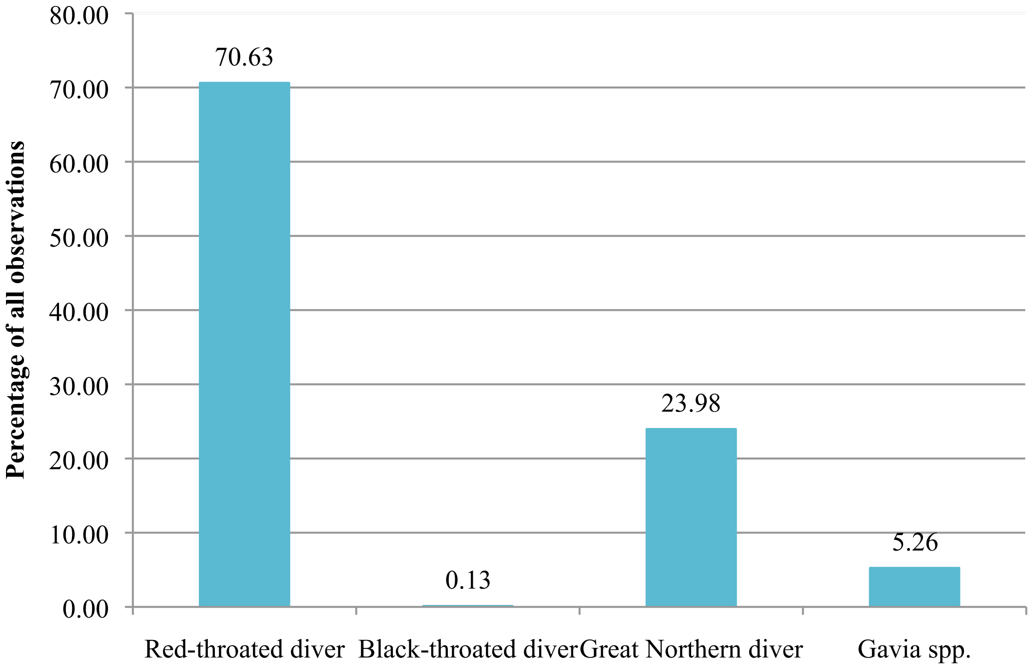


Figure 2.6: Proportion of all *Gavia spp.* sightings by species at the Fall of Warness.

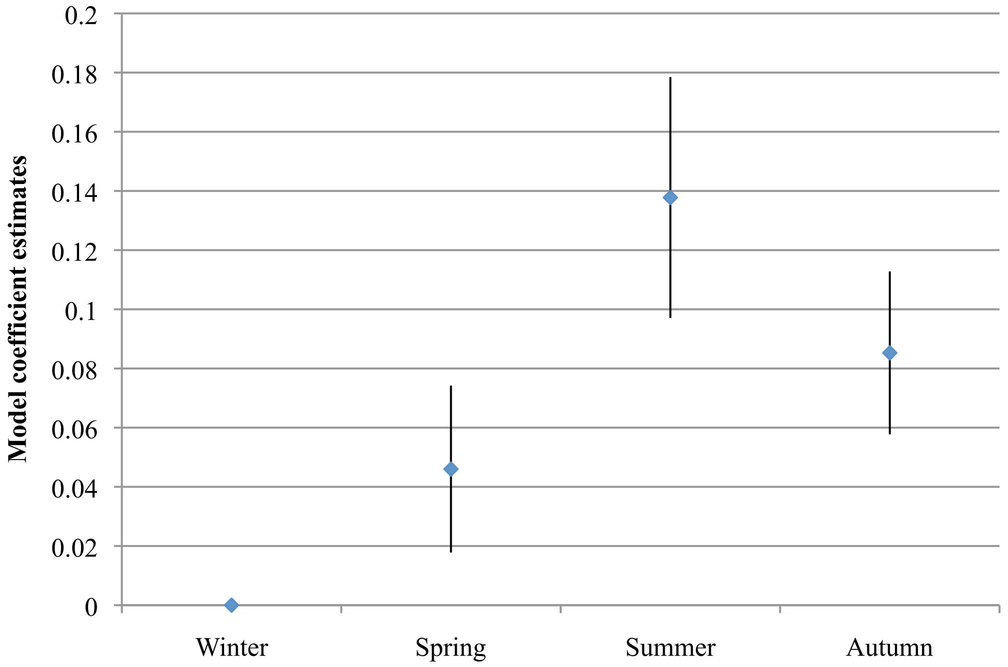


Figure 2.7: GEE coefficient estimates (and standard errors) for *Gavia spp.* observed by season at the Fall of Warness.

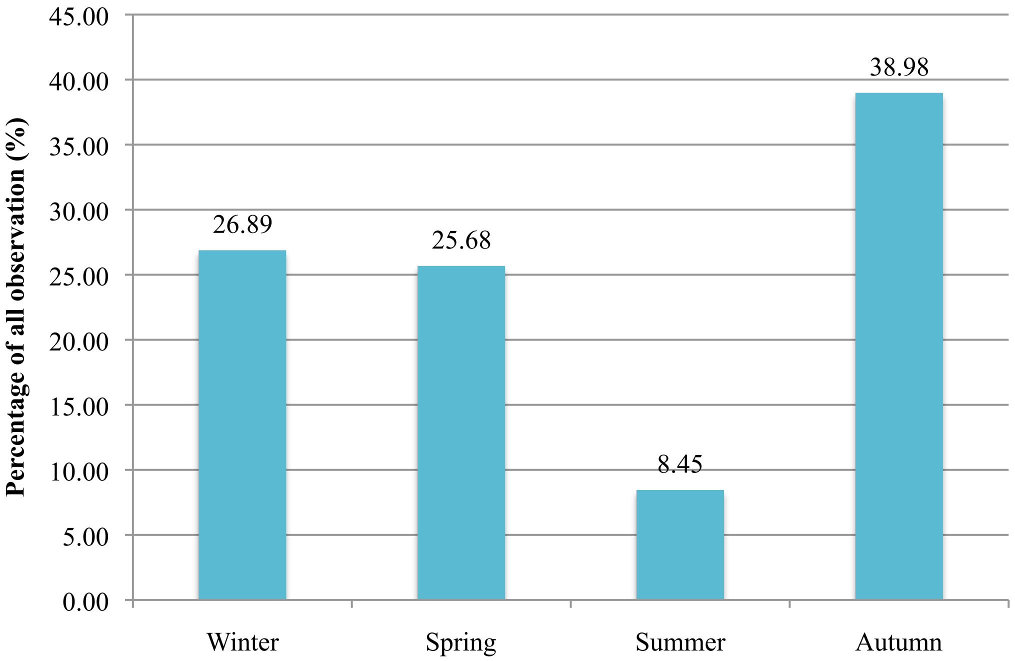


Figure 2.8: The proportion of *Gavia spp.* observed, by season at the Fall of Warness.

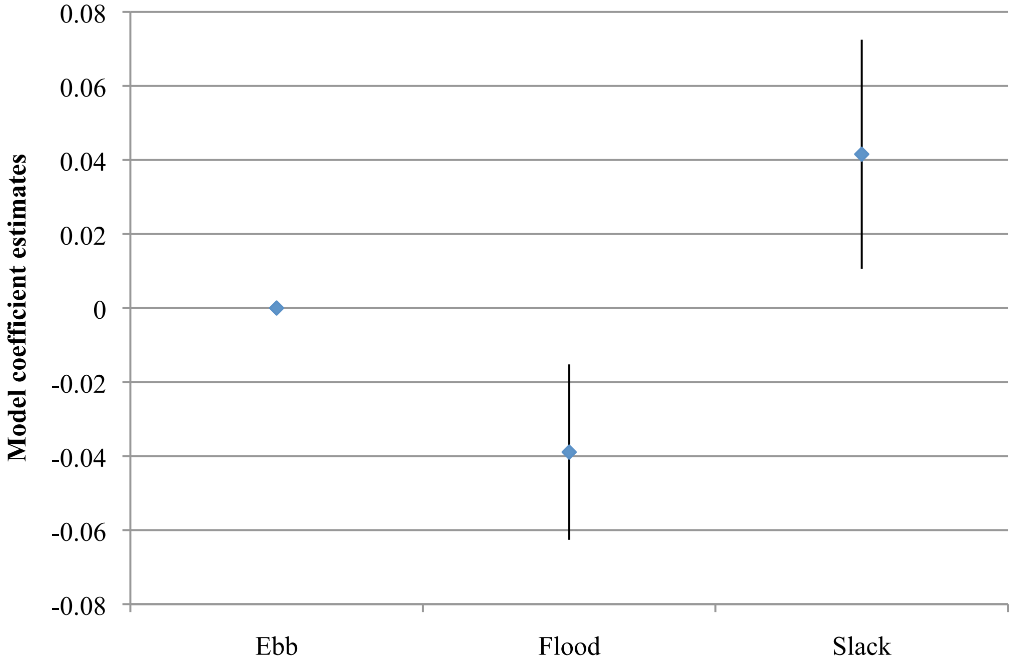


Figure 2.9: GEE coefficient estimates (and standard errors) for *Gavia spp.* observed by tidal state at the Fall of Warness.

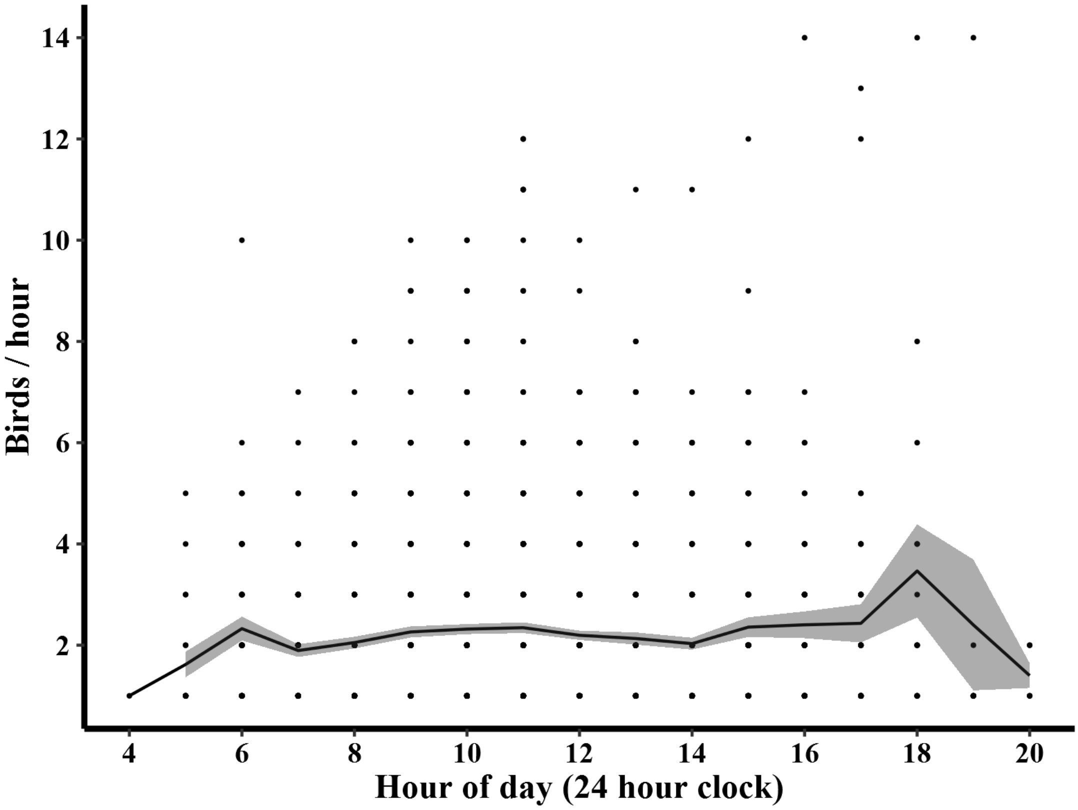


Figure 2.10: Mean number of *Gavia spp.* observed per hour, throughout the day at the Fall of Warness, with raw data overlaid and standard error around the mean shown by the grey area.

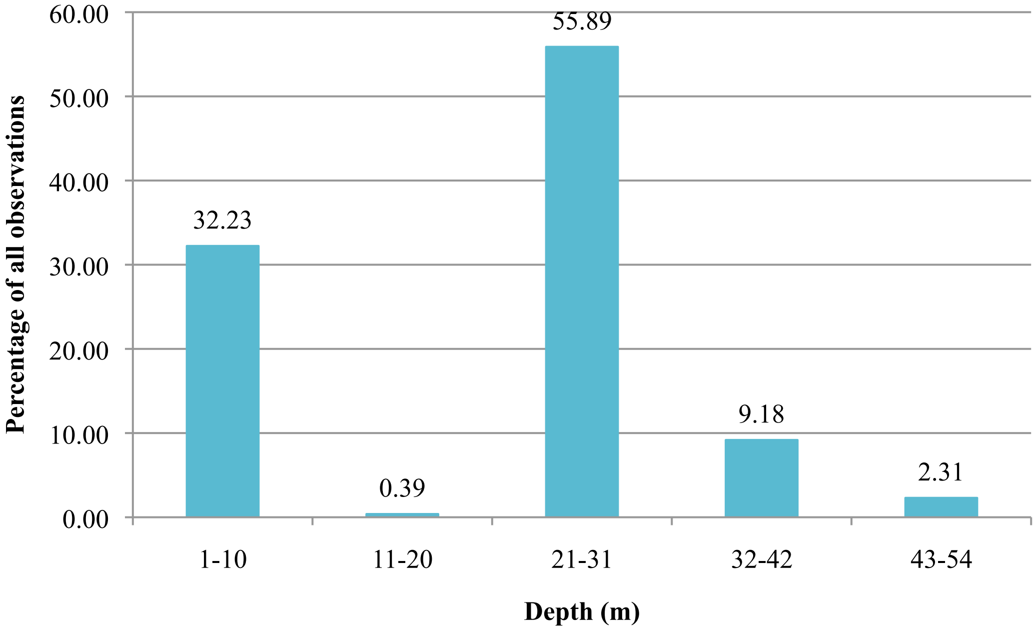


Figure 2.11: The proportion of observations *Gavia spp.* were observed at different depths, at the Fall of Warness.

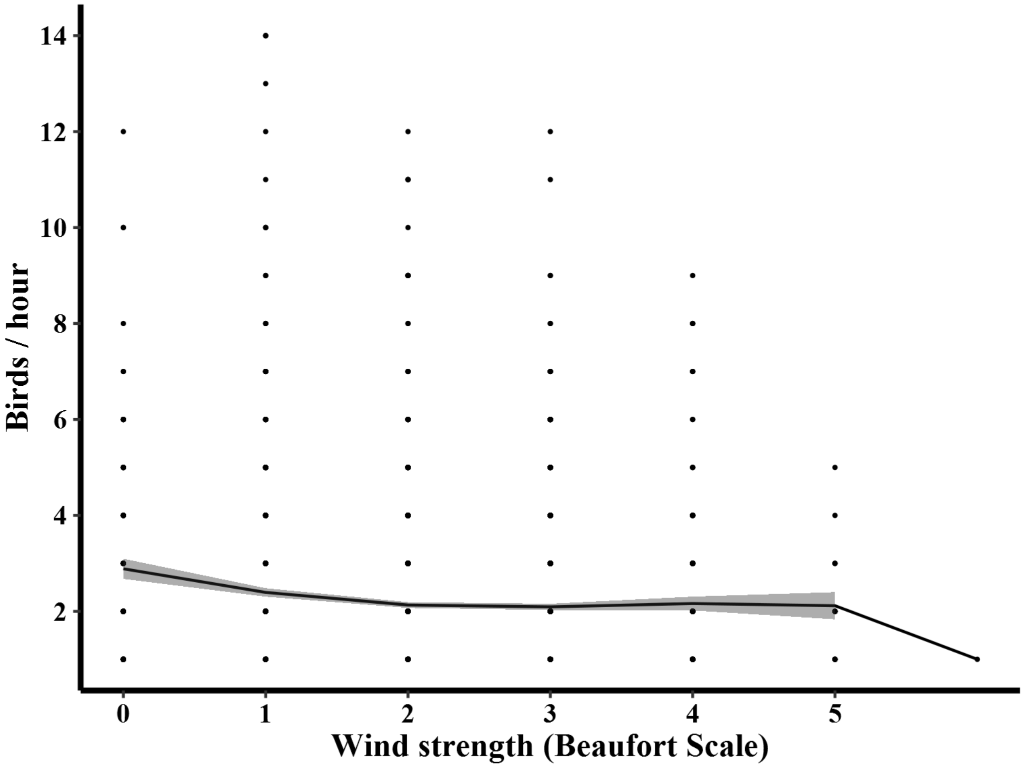


Figure 2.12: Mean number of *Gavia spp.* observed per hour during different wind strengths, using the Beaufort Scale, at the Fall of Warness, with raw data overlaid and standard error around the mean shown by the grey area.

2.3.3 Northern Gannet *Morus bassanus*

A total of 10,623 gannets were counted during the observation period. Of the top five GLMM models exploring the influence of environmental variables on numbers of gannets observed, season, habitat type and tidal state were most frequently selected and the GEE model also highlighted these as significant (Table 2.5). Time of day, depth, wind strength and cloud cover were also selected for in the top five GLMM models (S.Table 1.3). The GEE model estimated over dispersion (scale parameter) to be 3.4 and correlation between observations on the same day and grid squares to be 0.165.

Gannets were encountered more frequently during observations in the summer and autumn months, compared with winter and spring Fig. 2.13). The mean number of gannets observed per hour showed peaks and troughs throughout the day, with greater mean numbers observed around 07:00 (7 gannets/hour) and 20:00 (13 gannets/hour; Fig. 2.15). The peak at 04:00 was due to a single observation.

Gannet models consistently selected for habitat type, gannets being more frequently observed in coastal waters (75% of observations) compared with pelagic (25% of observations; S.Table 1.3). Furthermore, 62% of all observations found gannets in water 21-31m deep with only 23% in 1-10m waters (Fig. 2.16). Gannets were also more frequently observed during flood (43% of observations) and ebb tides (44% of observations) compared with slack tide (<13% of observations) (Fig. 2.14).

Table 2.5: Parameter estimates, standard errors, probability values for GEE investigating gannet counts as a function of season, habitat type and tide state.

	Estimate	**Std. error**	**Wald**	**Pr (>|W|)**	**Signif.**
(Intercept)	0.22111	0.03312	44.57	2.50E-11	***
Spring	0.00877	0.05241	0.03	0.867	
Summer	0.51166	0.05503	86.46	<2.00E-16	***
Autumn	0.44116	0.03756	137.92	<2.00E-16	***
Pelagic	0.28083	0.05795	23.48	1.30E-06	***
Flood	0.0075	0.04446	0.03	0.8661	
Slack	-0.12617	0.04685	7.25	0.0071	**

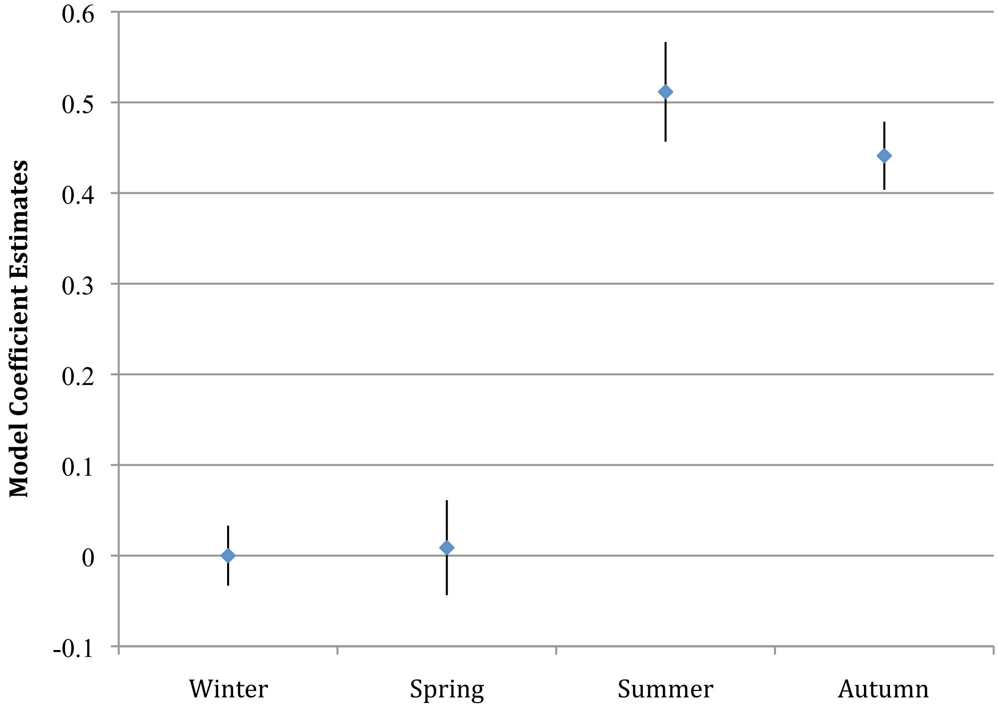


Figure 2.13: GEE coefficient estimates (and standard errors) for northern gannets observed by season at the Fall of Warness.

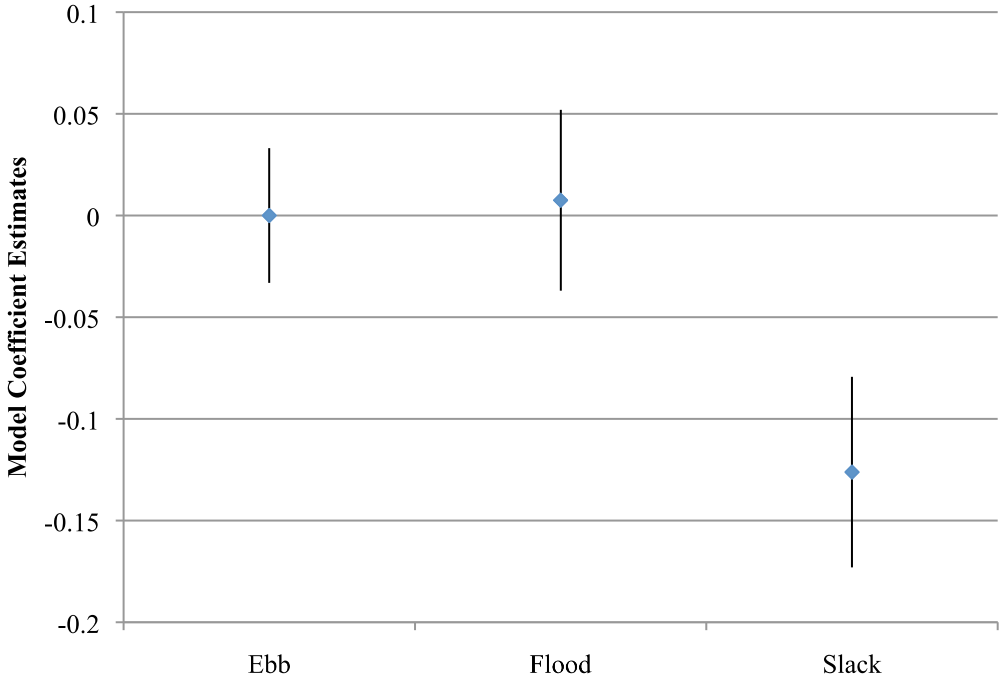


Figure 2.14: GEE coefficient estimates (and standard errors) for northern gannets observed by tidal state at the Fall of Warness.

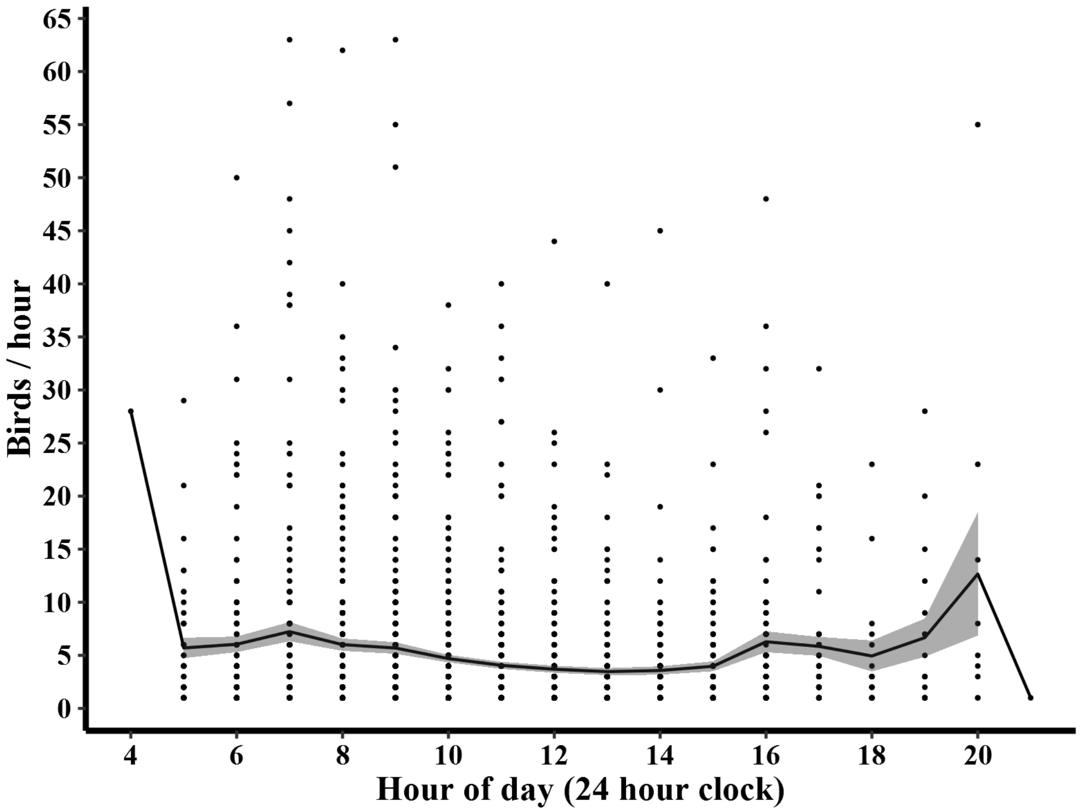


Figure 2.15: Mean number of northern gannets observed per hour, throughout the day at the Fall of Warness, with raw data overlaid and standard error around the mean shown by the grey area.

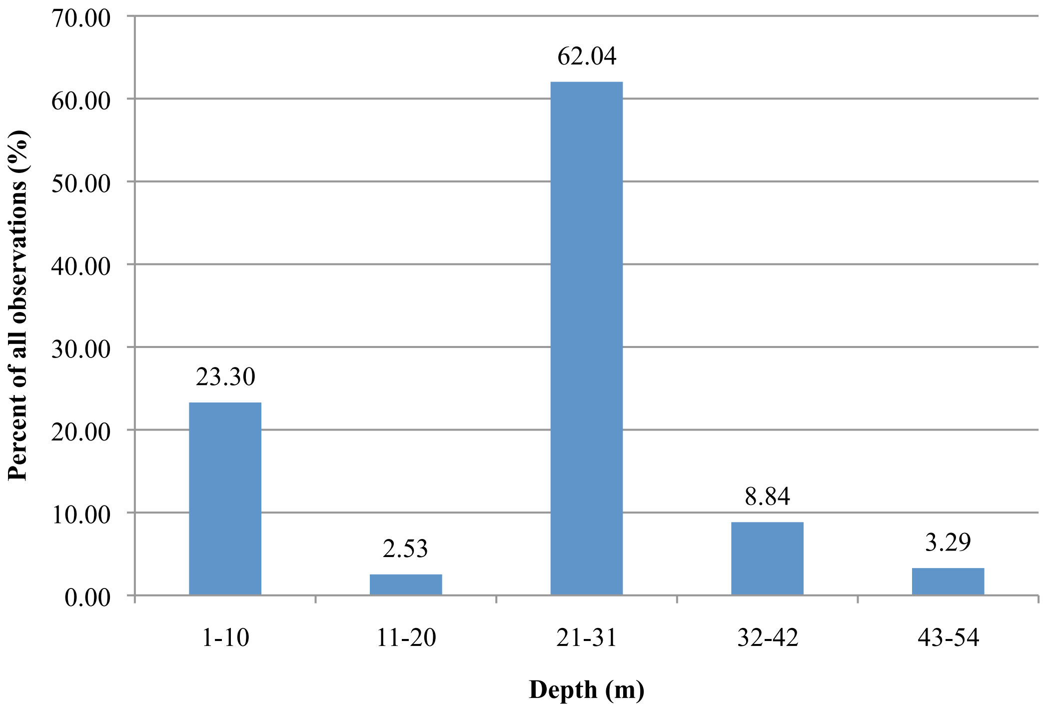


Figure 2.16: The proportion of observations northern gannet were observed at different depths, at the Fall of Warness.

2.3.4 Cormorants *Phalacrocorax spp.*

Phalacrocorax spp. (shags and cormorants) were by far the most numerous of the seabirds observed at the Fall of Warness during the study period, with 145,613 individuals counted. Of these, 30% were identified as European Shags (*P. aristotelis*), 7% were great cormorants (*P. carbo*) and the remaining 63% were unidentified to species. The best *Phalacrocorax spp.* GLMM models contained the variables for season, time of day and tide state (with an interaction), depth, habitat type, wind strength and cloud cover (Table 2.6 and S. Table 1.4). The GEE model indicated that season, time of day, depth, wind strength and cloud cover were significant. The GEE model estimated over dispersion (scale parameter) to be 38.1 and correlation between observations on the same day and grid squares to be 0.094.

Phalacrocorax spp. showed a number of temporal patterns. The encounter rate of *Phalacrocorax spp.* observed varied between seasons, with greater numbers observed during the autumn and winter months (Fig. 2.17), while mean hourly encounter rate peaked between 09:00 and 14:00 followed by a steady decrease (Fig. 2.19)

48% of the shags and cormorants observed were active over depths of between 1-10m deep, while >44% were observed within 21-31m deep (Fig. 2.20). Similarly, the GEE model coefficients indicate that a greater number of shags and cormorants were observed in coastal areas, as opposed to pelagic (Table 2.6).

Fewer cormorants were encountered during flood and slack tidal states (Fig. 2.18). The top two GLMM models included an interaction between time of day and tidal state showing that the mean hourly encounter rate was higher during ebbing tides compared with flooding or slack tides (Fig. 2.19). The mean hourly encounter rate of shags and cormorants was found to decrease with increasing wind strength (Fig. 2.21).

Table 2.6: Parameter estimates, standard errors, probability values for GEE investigating *Phalacrocorax spp.* counts as a function of season, time of day*tidal state, depth, habitat type, wind strength and cloud cover.

	Estimate	Std. error	Wald	Pr (>|W|)	Signif.
(Intercept)	3.23055	0.12499	668.03	<2.00E-16	***
Spring	-1.03995	0.04698	489.91	<2.00E-16	***
Summer	-1.43267	0.05081	794.94	<2.00E-16	***
Autumn	0.07514	0.05067	2.2	0.1381	*
Time of Day	-0.05051	0.00841	36.08	1.90E-09	***
Flood	-0.09722	0.15366	0.4	0.5269	
Slack	-0.21697	0.20434	1.13	0.2883	
Depth	-0.0213	0.00203	110.26	<2.00E-16	***
Pelagic	-0.06708	0.07167	0.88	0.3493	
Wind Strength	-0.04697	0.01894	6.15	0.0132	**
Cloud Cover	0.00222	0.0007	10.04	0.0015	***
Time of Day*Flood	-0.01456	0.01282	1.29	0.256	
Time of Day*Slack	0.00101	0.01632	0	0.9504	

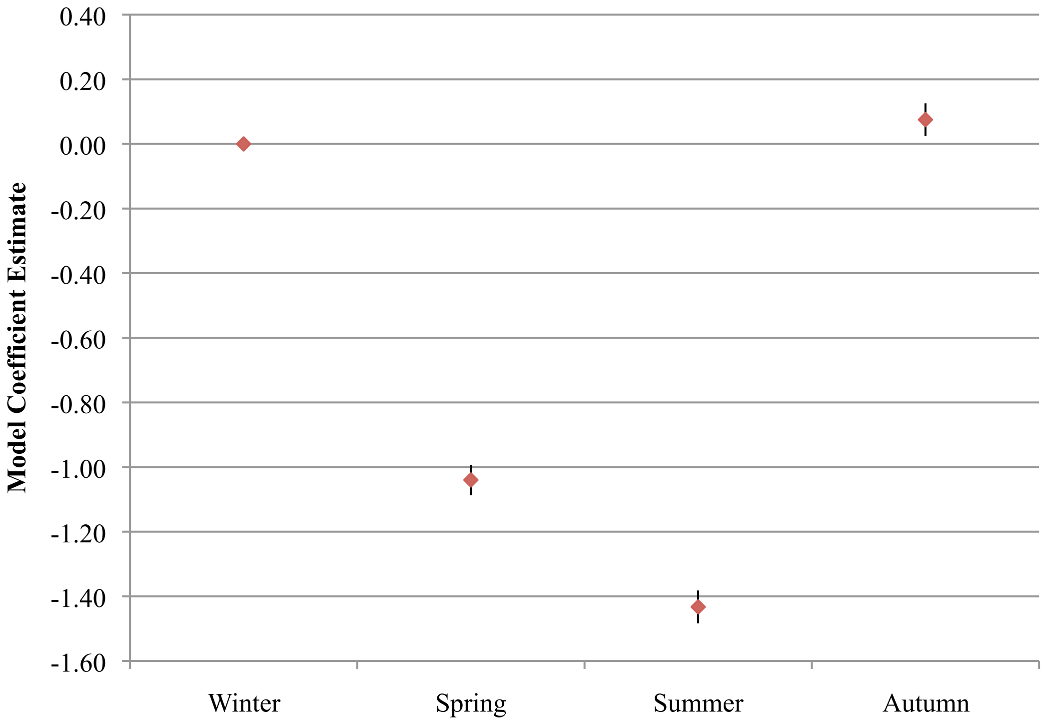


Figure 2.17: GEE coefficient estimates (and standard errors) for *Phalacrocorax spp.* observed by season at the Fall of Warness.

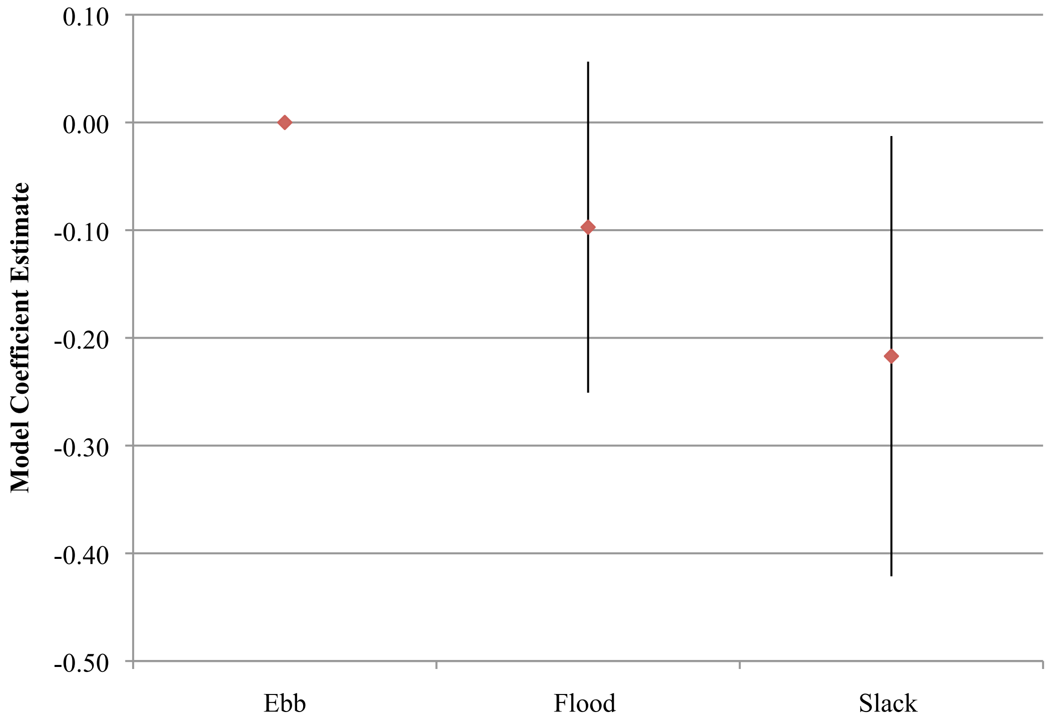


Figure 2.18: GEE coefficient estimates (and standard errors) for *Phalacrocorax spp.* observed by tidal state at the Fall of Warness.

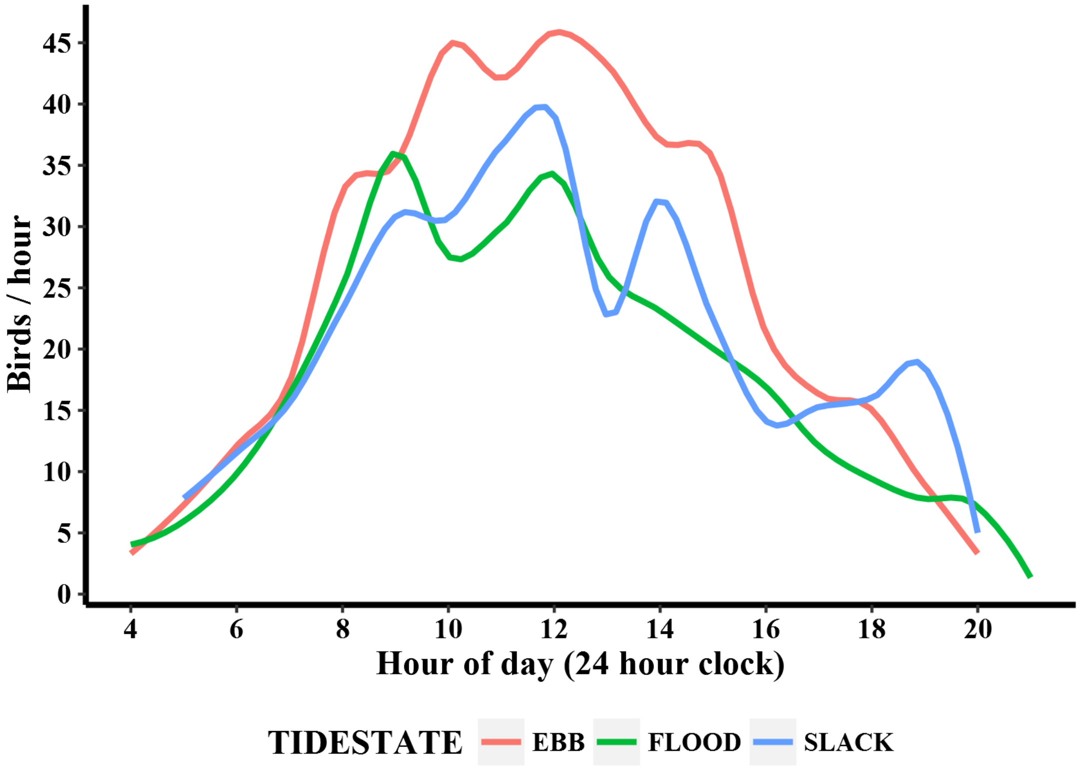


Figure 2.19: Mean number of *Phalacrocorax spp.* observed per hour, throughout the day at the Fall of Warness by ebb, flood and slack tidal states.

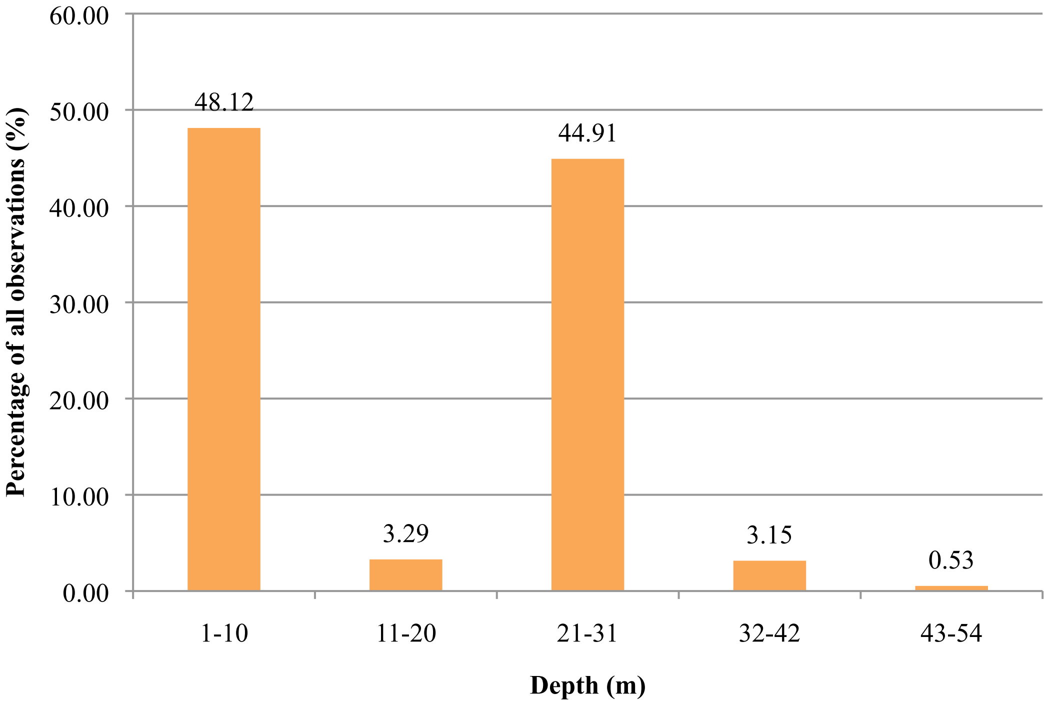


Figure 2.20: The proportion of observations *Phalacrocorax spp.* were observed at different depths, at the Fall of Warness.

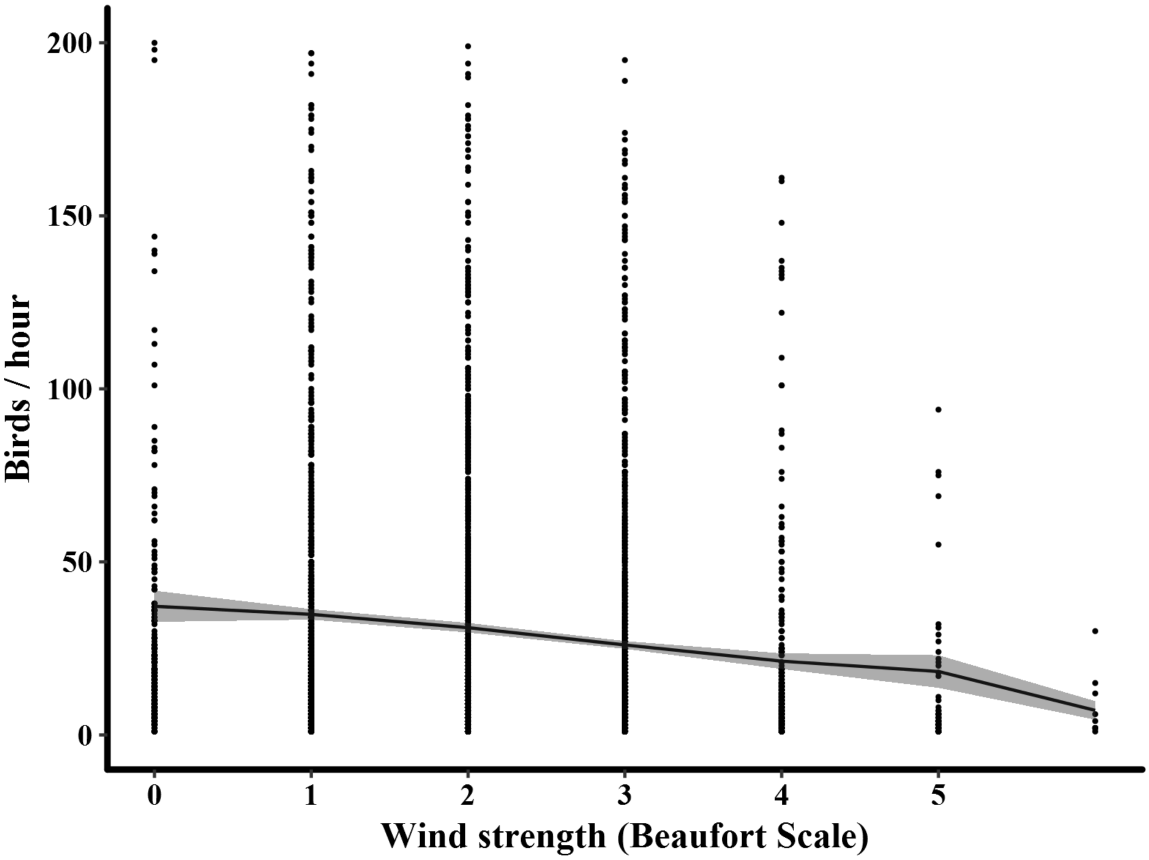


Figure 2.21: Mean number of *Phalacrocorax spp.* observed per hour during different wind strengths, using the Beaufort Scale, at the Fall of Warness, with raw data overlaid and standard error around the mean shown by the grey area. This figure was clipped to 95th percentile of the data.

2.3.5 Common Guillemot *Uria aalge*

A total of 54,771 common guillemots were counted during the observation period at the Fall of Warness. The top five GLMM models for common guillemot counts all included the variables for season, time of day, habitat type, wind direction, tide states and cloud cover (Table 2.7 and S. Table 1.5). The GEE results indicated that the influence of season, habitat type, wind direction and tide state was significant. The GEE model estimated over dispersion (scale parameter) to be 173 and correlation between observations on the same day and grid squares to be 0.453.

Common guillemot numbers varied across the seasons with greater numbers being seen in the spring and summer months. Similarly, the mean hourly common guillemot encounter rate also varied throughout the day. While 53% of all common guillemot observations occurred between 06:00 and 09:00, the morning peak encounter rate was 53 birds per hour at 08:00. The peak occurring between 16:00 and 17:00 is due to an outlier of 1500 birds in a single observation period (Fig. 2.23).

With respect to the marine environment, 84% of all common guillemots observed were recorded in pelagic areas (Fig. 2.24), and 91% in water of 20m or deeper (Fig. 2.25). Responses to tidal state indicate that greater numbers of common guillemots were observed during flood tides, and fewer during slack tides (Fig. 2.26). Models suggest that wind direction was more important than strength and figure 2.25 shows greater numbers of common guillemots occurred during southerly or variable winds.

Table 2.7: Parameter estimates, standard errors, probability values for GEE investigating common guillemot counts as a function of season, time of day, habitat type, wind direction, tidal state, precipitation and cloud cover.

	Estimate	Std. error	Wald	Pr (>|W|)	Signif.
(Intercept)	-0.4524	0.43828	1.07	0.302	
Spring	1.97919	0.15543	162.14	<2e-16	***
Summer	1.94014	0.13549	205.05	<2e-16	***
Autumn	-0.12489	0.10986	1.29	0.256	
Time of Day	-0.01099	0.01932	0.32	0.57	
Pelagic	1.38857	0.14841	87.54	<2e-16	***
Wind Dir None	-0.67984	0.28525	5.68	0.017	*
Wind Dir North	-0.32021	0.21244	2.27	0.132	
Wind Dir South	0.3448	0.21848	2.49	0.115	
Wind Dir Variable	0.52023	0.31009	2.81	0.093	.
Wind Dir West	-0.61842	0.24004	6.64	0.01	**
Flood	0.2966	0.16072	3.41	0.065	.
Slack	-0.36078	0.15707	5.28	0.022	*
Precipitation	0.03005	0.2187	0.02	0.891	
Cloud Cover	0.00342	0.00213	2.58	0.108	

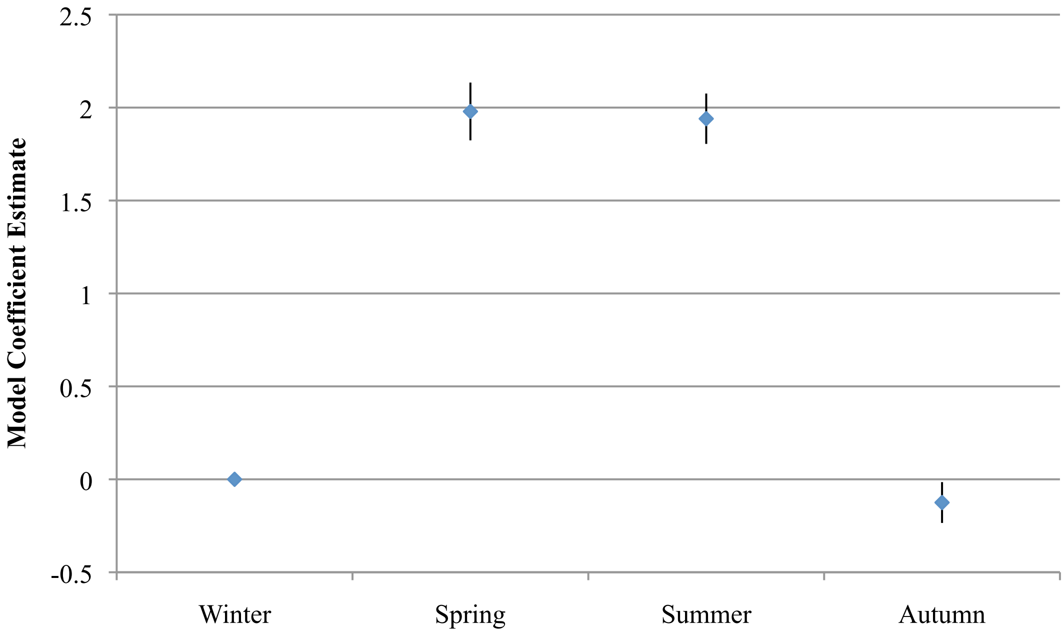


Figure 2.22: GEE coefficient estimates (and standard errors) for common guillemots observed by season at the Fall of Warness.

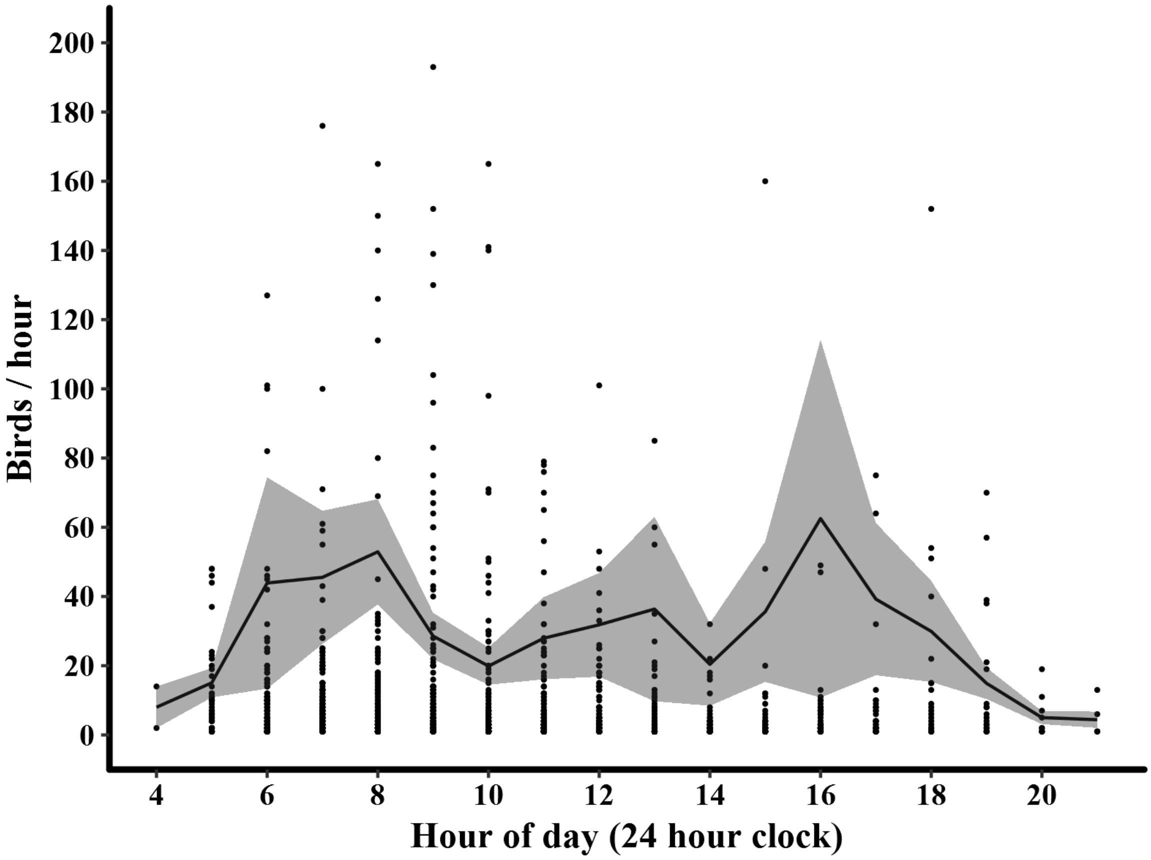


Figure 2.23: Mean number of common guillemots observed per hour, throughout the day at the Fall of Warness, with raw data overlaid and standard error around the mean shown by the grey area. This figure was clipped to 95th percentile of the data.

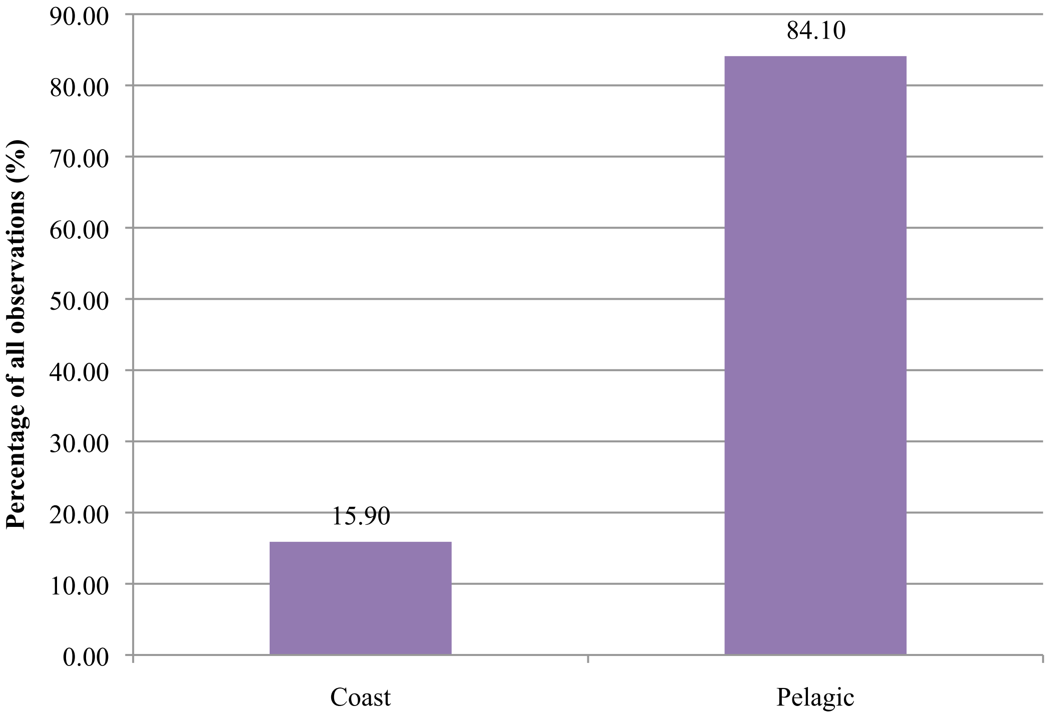


Figure 2.24: The proportion of observations common guillemots observed in different pelagic and coastal habitats at the Fall of Warness.

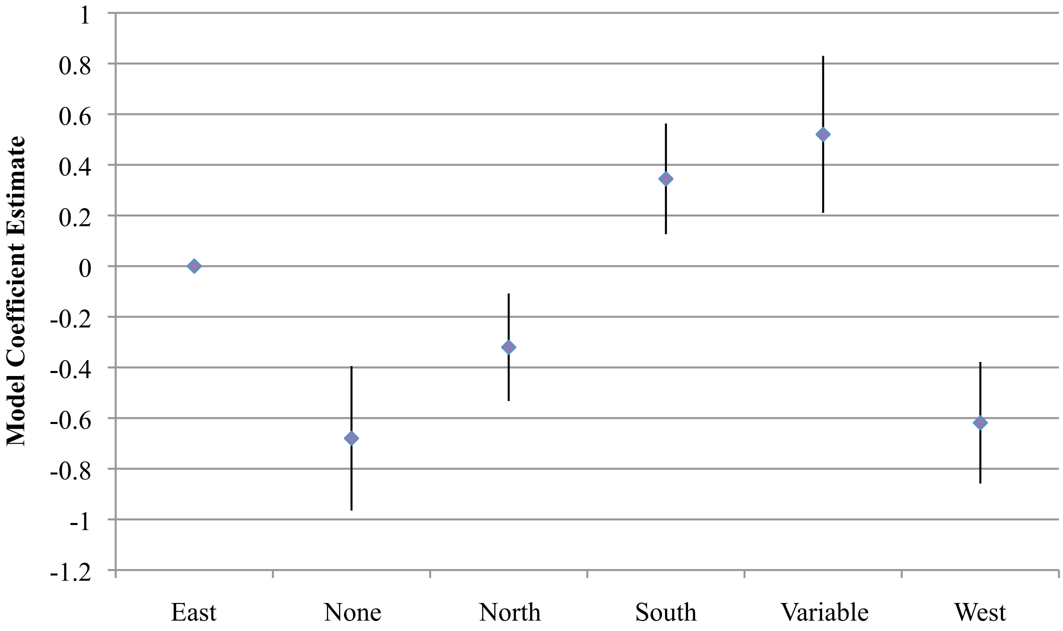


Figure 2.25: GEE coefficient estimates (and standard errors) for common guillemots observed with different wind directions at the Fall of Warness.

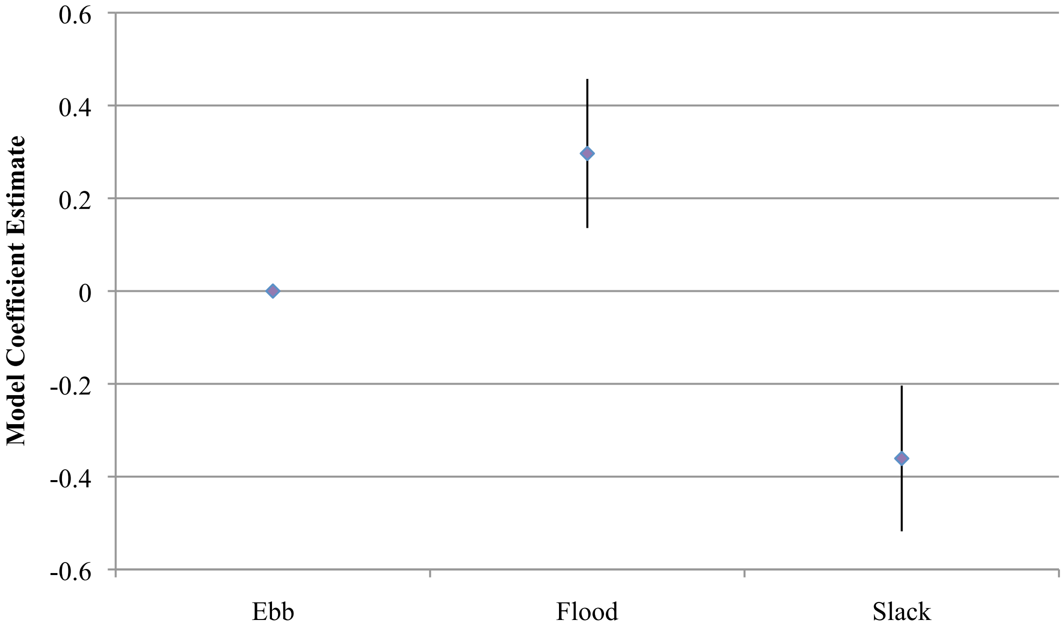


Figure 2.26: GEE coefficient estimates (and standard errors) for common guillemots observed by tidal state, at the Fall of Warness.

2.3.6 Razorbill *Alca torda*

Razorbills were the least numerous seabirds observed regularly in this study with only 1,699 counted during the observation period. The top five GLMM models for razorbill counts included the variables for season, time of day, habitat type, wind direction, tide states and cloud cover, with precipitation and depth selected for, in two and three of the top five models (refer to Table 2.8 and S. Table 1.6). The GEE model estimated over dispersion (scale parameter) to be 7.7 and correlation between observations on the same day and grid squares to be 0.395.

The GEE model predicts that higher numbers razorbills would be observed during the spring months, compared with summer, autumn and winter months (Fig. 2.27). Figure 2.28 also suggests temporal variation in encounter rate, with a peak occurring in the morning and declining throughout the day, although this was not selected in the GEE model (Table 2.8). It is possible this result is skewed due to the low number of razorbills observed. Razorbills were more frequently observed in the pelagic environment than coastal grid squares (Fig. 2.29). They were also observed more frequently during flooding tides, than ebb or slack tides (Fig. 2.31). The observations of this species were also predicted to be influenced by wind direction. The GEE model predicted that more razorbills would likely be observed when there was no wind, and fewer razorbills during westerly winds (Fig. 2.30).

Table 2.8: Parameter estimates, standard errors, probability values for GEE investigating razorbills counts as a function of season, time of day, habitat type, wind direction, tidal state, precipitation and cloud cover.

	Estimate	**Std. error**	**Wald**	**Pr (>|W|)**	**Signif.**
(Intercept)	-2.7454	1.0299	7.11	0.00768	**
Spring	0.9363	0.4055	5.33	0.02093	*
Summer	0.0372	0.4537	0.01	0.93463	
Autumn	-0.3689	0.4816	0.59	0.44362	
Time of Day	0.0659	0.0535	1.52	0.21826	
Pelagic	1.2334	0.203	36.91	1.20E-09	***
Wind Dir None	1.4526	0.4412	10.84	0.00099	***
Wind Dir North	-0.7294	0.3462	4.44	0.03515	*
Wind Dir South	-0.7111	0.3338	4.54	0.03314	*
Wind Dir Variable	-0.4395	0.5374	0.67	0.41345	
Wind Dir West	-1.12	0.4758	5.54	0.01858	*
Flood	1.0937	0.2847	14.76	0.00012	***
Slack	0.3681	0.2857	1.66	0.19759	
Precipitation	0.3396	0.3088	1.21	0.27154	
Cloud Cover	0.0209	0.0048	18.99	1.30E-05	***

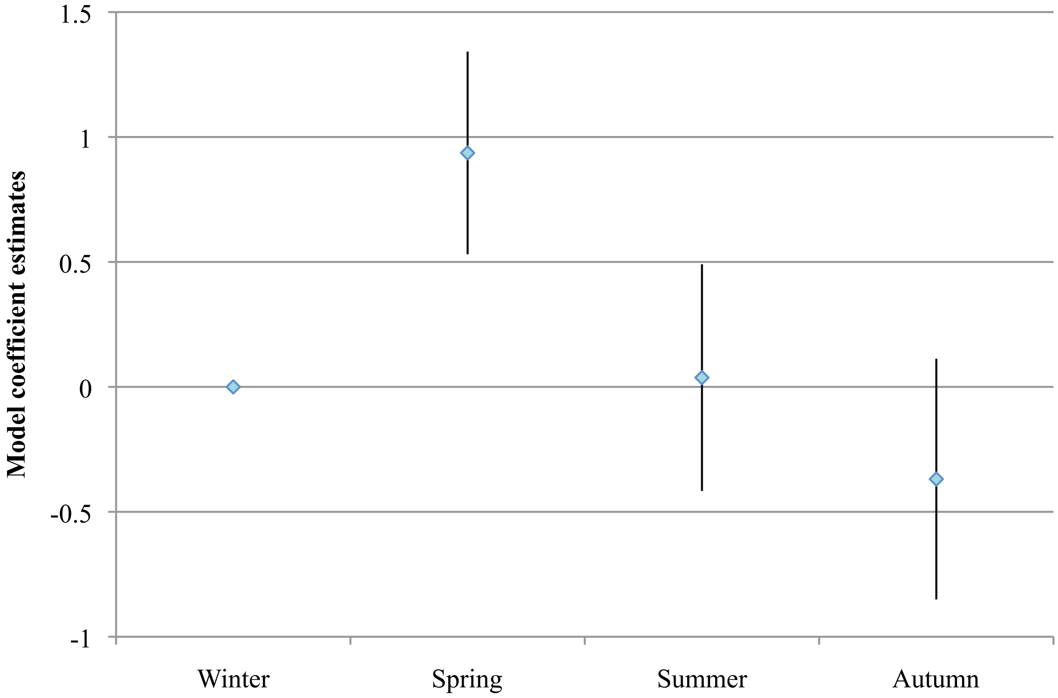


Figure 2.27: GEE coefficient estimates (and standard errors) for razorbills observed by season at the Fall of Warness.

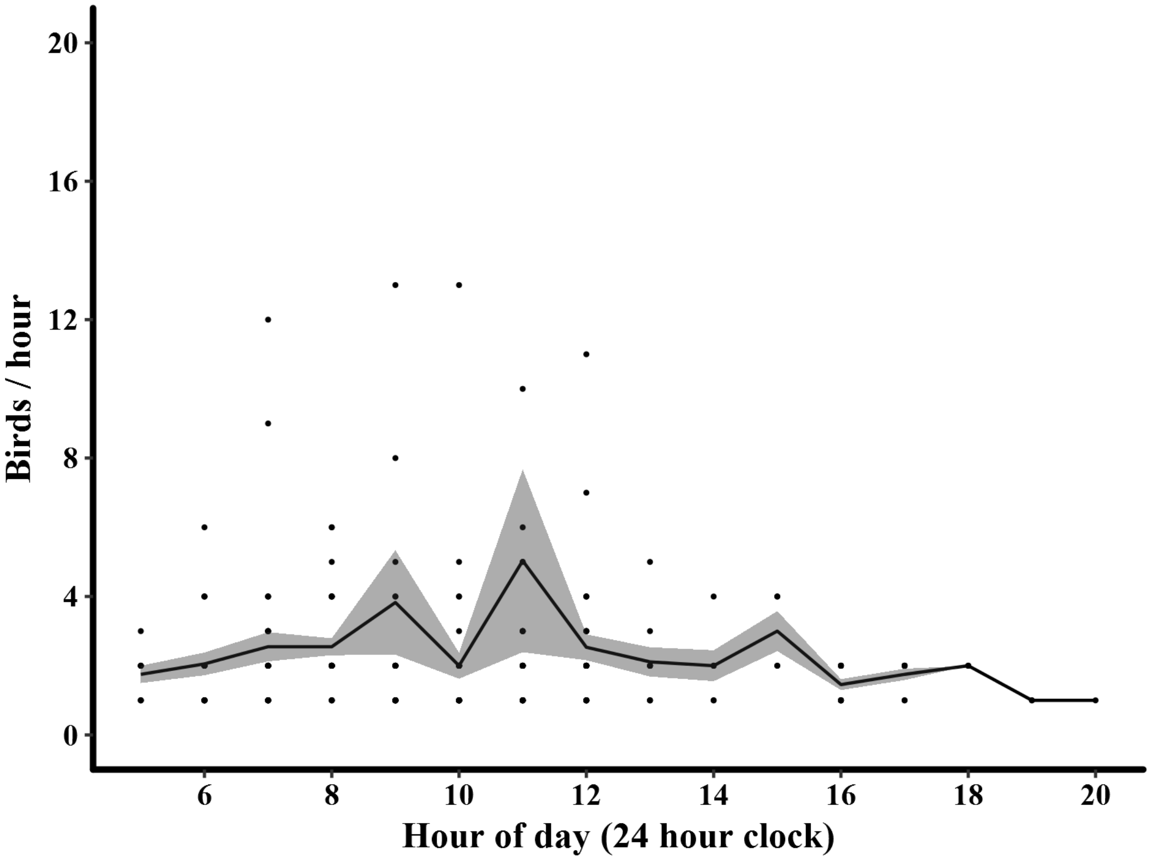


Figure 2.28: Mean number of razorbills observed per hour, throughout the day at the Fall of Warness, with raw data overlaid and standard error around the mean shown by the grey area. This figure was clipped to 95th percentile of the data.

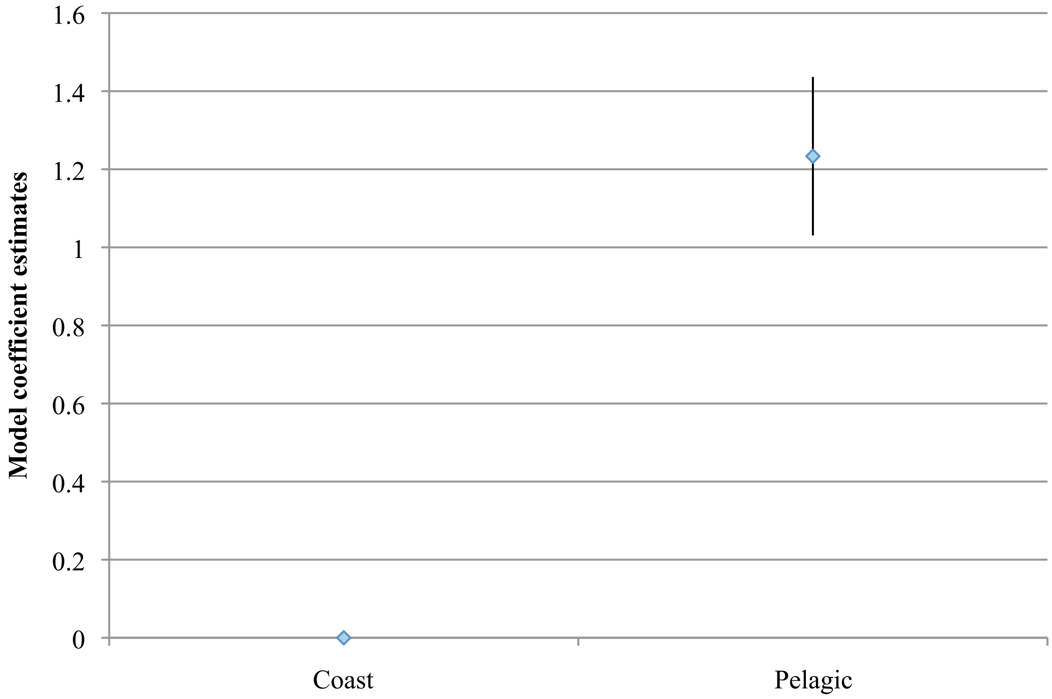


Figure 2.29: GEE coefficient estimates (and standard errors) for razorbills observed in coastal and pelagic habitats, at the Fall of Warness.

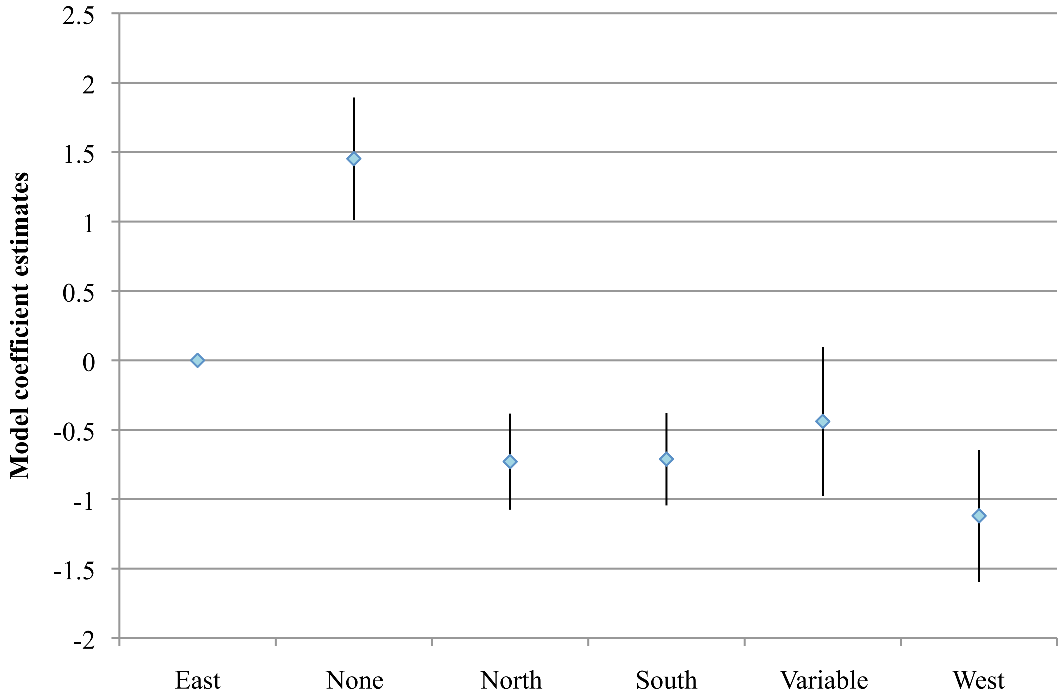


Figure 2.30: GEE coefficient estimates (and standard errors) for razorbills observed with different wind directions at the Fall of Warness.

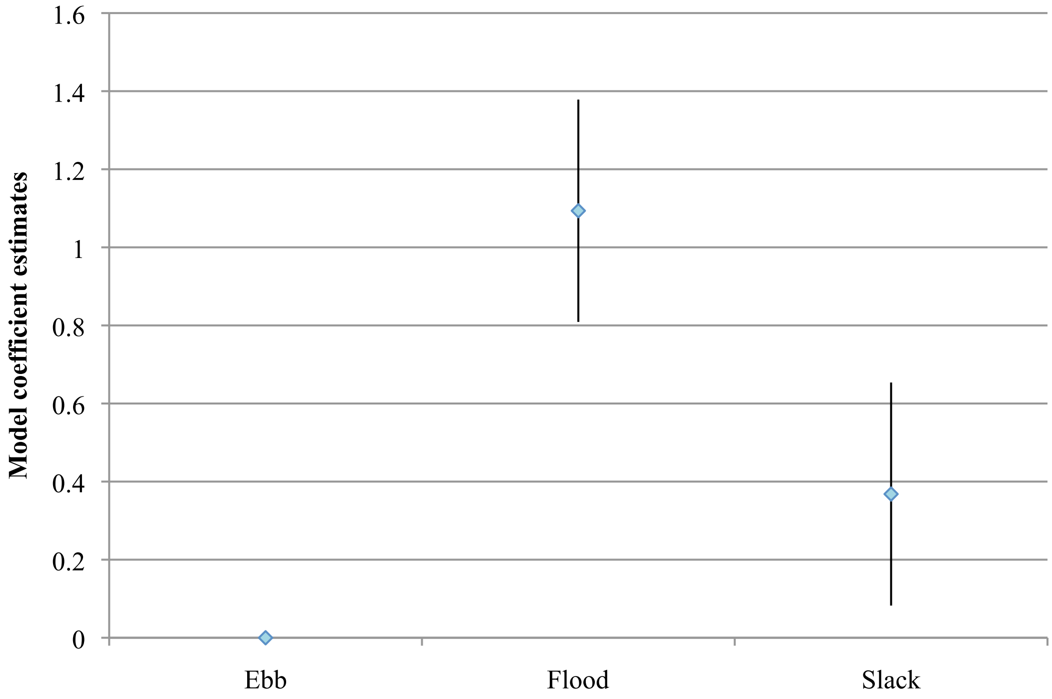


Figure 2.31: GEE coefficient estimates (and standard errors) for razorbills observed by tidal state, at the Fall of Warness.

2.3.7 Black Guillemot *Cepphus grylle*

A total of 78,071 black guillemots were observed at the Fall of Warness site. The top GLMM models indicate that black guillemot numbers are a function of season, time of day interacting with tidal state, depth, habitat type, wind strength, precipitation and cloud cover (S.Table 1.7). The GEE model coefficient estimates (and standard errors) suggest that all these variables were significant, excluding precipitation (Table 2.9). The GEE model estimated over dispersion (scale parameter) to be 7.38 and correlation between observations on the same day and grid squares to be 0.531.

Black guillemots were observed more during the spring and summer months (Fig. 2.32), with 35% of all observations occurring in spring and 45% in summer. Figure 2.33 highlights a clear decreasing temporal trend in black guillemot encounter rate throughout the day (mean number of birds per hour), with peak encounters occurring between 04:00 and 08:00. Unlike the other auk species, black guillemots were observed more frequently in coastal grid squares than in pelagic grid squares (Fig. 2.35). This was only partially reflected by water depth: with 40% of all observations occurring in water between 1 and 10m deep and 42% in water between 21 and 31m deep (Fig. 2.36).

The GEE model coefficients suggest that fewer black guillemots were seen during flooding tides (-0.445 ± 9.079SE), compared with ebb and slack tides (0.03±0.096) (Fig. 2.37). Furthermore, tidal state interacted with time of day showing the mean number of birds encountered per hour during flooding tides is less between 04:00 and 09:00, compared with ebbing and slack tides (the ebb mean peak at 05:00 = 57 black guillemots/hour, the flood mean = 37 black guillemots/hour, and the slack mean = 50 black guillemots/hour; Fig. 2.34). However, after 14:00 black guillemot encounter rate during flooding tides was slightly higher than the encounter rate during ebbing or slack tides. Finally, the mean hourly encounter rate for black guillemots decreased with increasing wind strength (Fig. 2.38).

Table 2.9: Parameter estimates, standard errors, probability values for GEE investigating black guillemot counts as a function of season, time of day*tidal state, depth, habitat type, wind strength, precipitation and cloud cover.

	Estimate	**Std. error**	**Wald**	**Pr (>|W|)**	**Signif.**
(Intercept)	3.021314	0.085964	1235.27	<2.00E-16	***
Spring	0.474125	0.047848	98.19	<2.00E-16	***
Summer	0.117568	0.04888	5.79	0.01616	*
Autumn	-0.166884	0.057867	8.32	0.00393	**
Time of Day	-0.119139	0.005556	459.84	<2.00E-16	***
Flood	-0.445485	0.079263	31.59	1.90E-08	***
Slack	0.030022	0.096358	0.1	0.75537	
Depth	-0.012933	0.001417	83.29	<2.00E-16	***
Pelagic	-0.274787	0.038852	50.02	1.50E-12	***
Wind Strength	-0.09639	0.014489	44.26	2.90E-11	***
Precipitation	0.029342	0.028613	1.05	0.30513	
Cloud Cover	0.001464	0.000399	13.44	0.00025	***
Time of Day*Flood	0.038906	0.007624	26.04	3.30E-07	***
Time of Day*Slack	-0.009054	0.009494	0.91	0.34026	

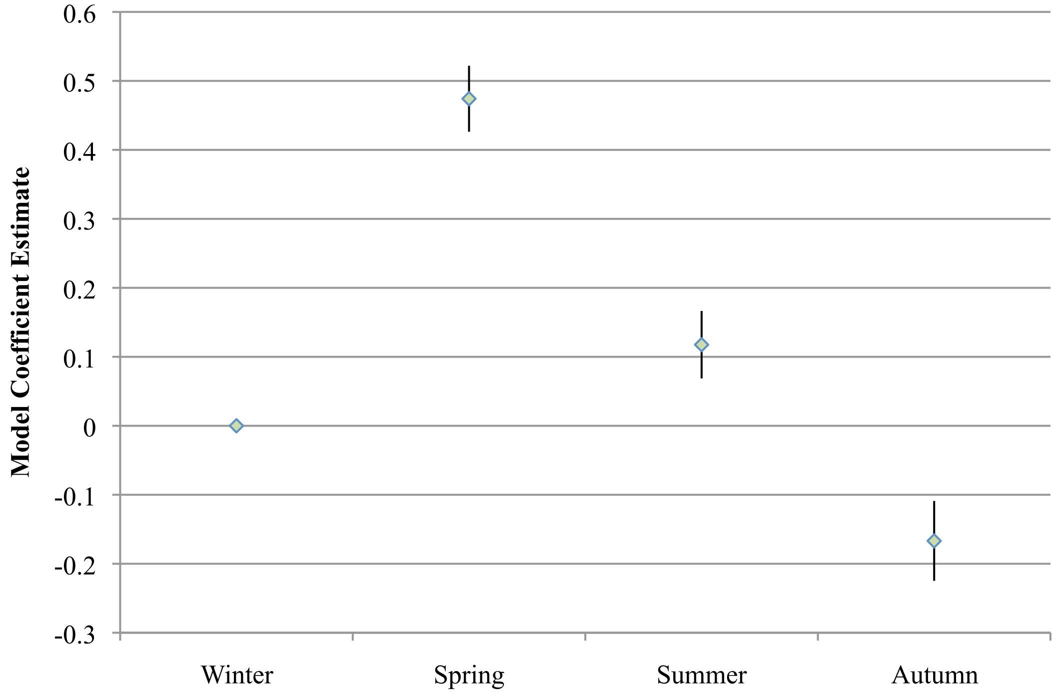


Figure 2.32: GEE coefficient estimates (and standard errors) for black guillemots observed by season at the Fall of Warness.

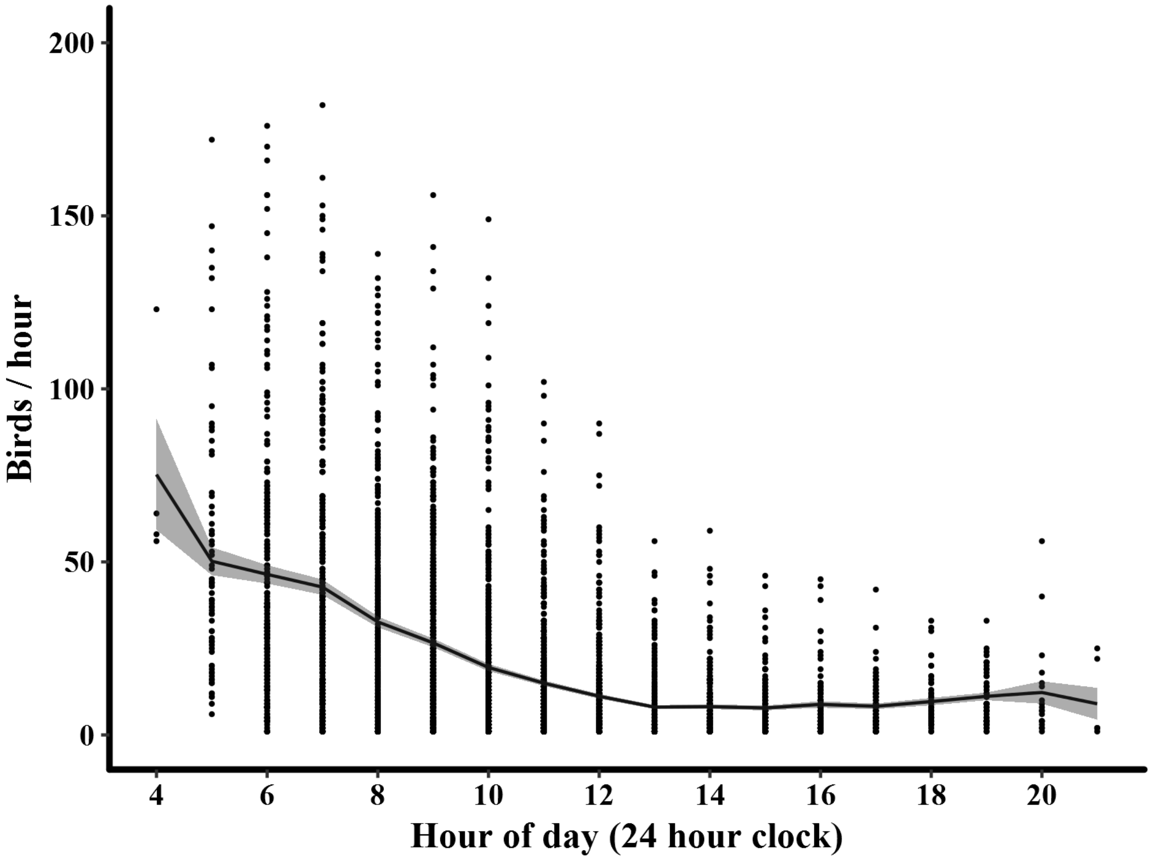


Figure 2.33: Mean number of black guillemots observed per hour, throughout the day at the Fall of Warness, with raw data overlaid and standard error around the mean shown by the grey area. This figure was clipped to 95th percentile of the data.

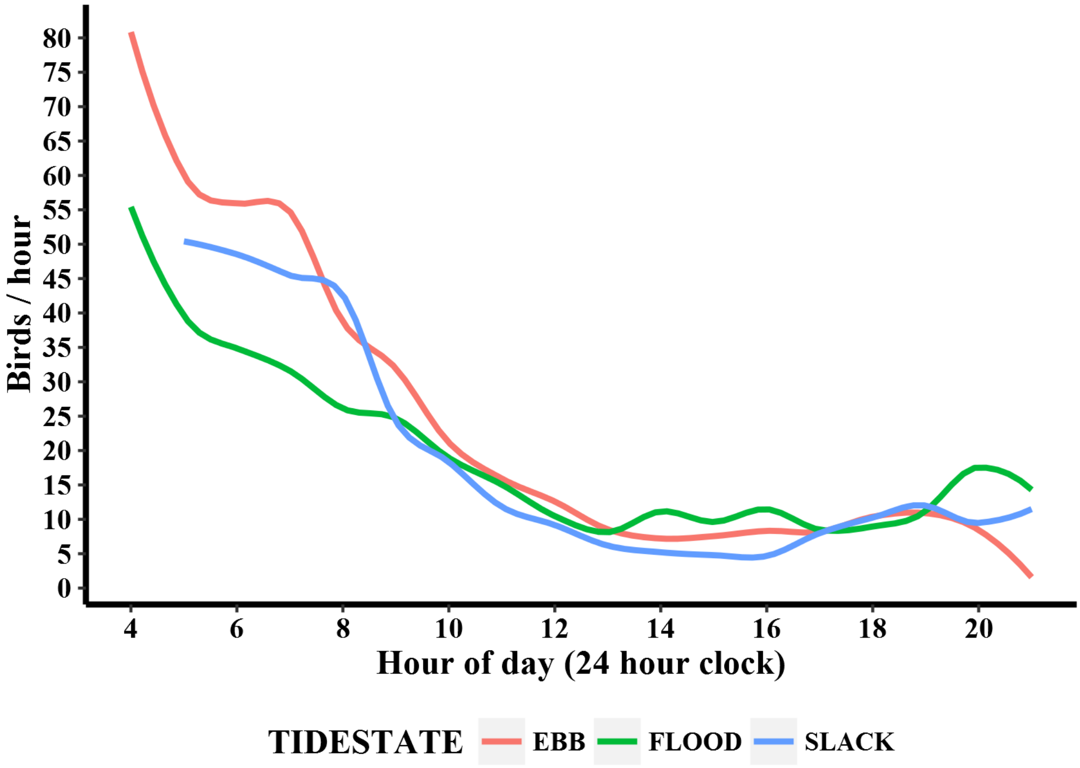


Figure 2.34: Mean number of black guillemots observed per hour, throughout the day at the Fall of Warness by ebb, flood and slack tidal states.

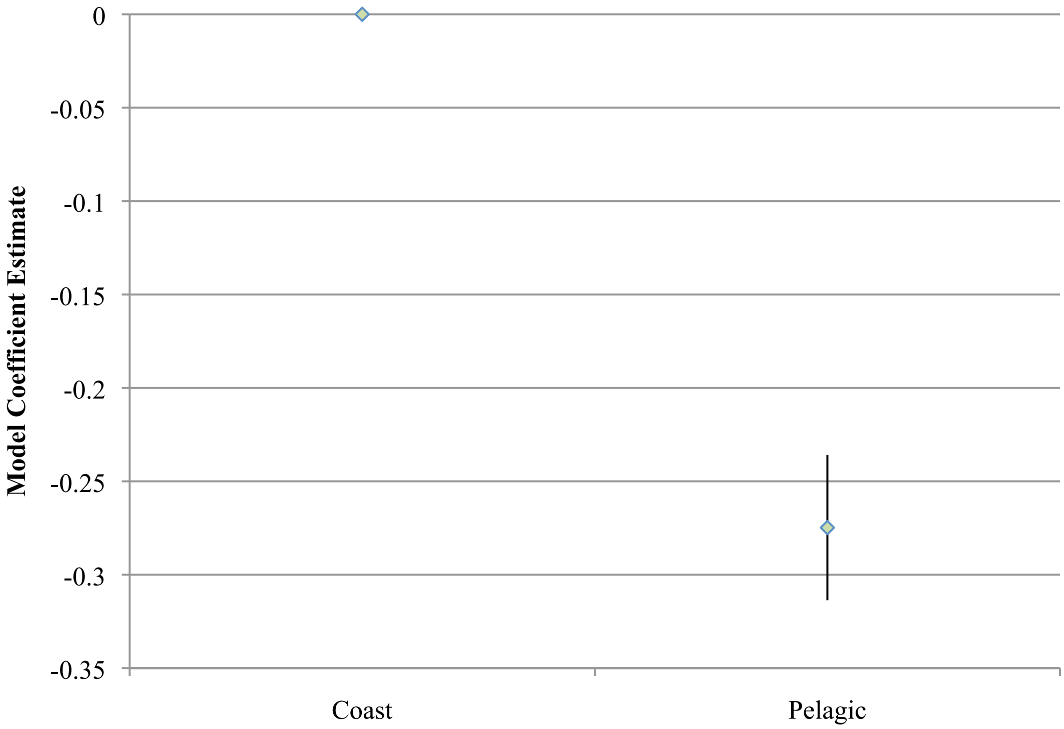


Figure 2.35: GEE coefficient estimates (and standard errors) for black guillemots observed in coastal and pelagic habitats, at the Fall of Warness.

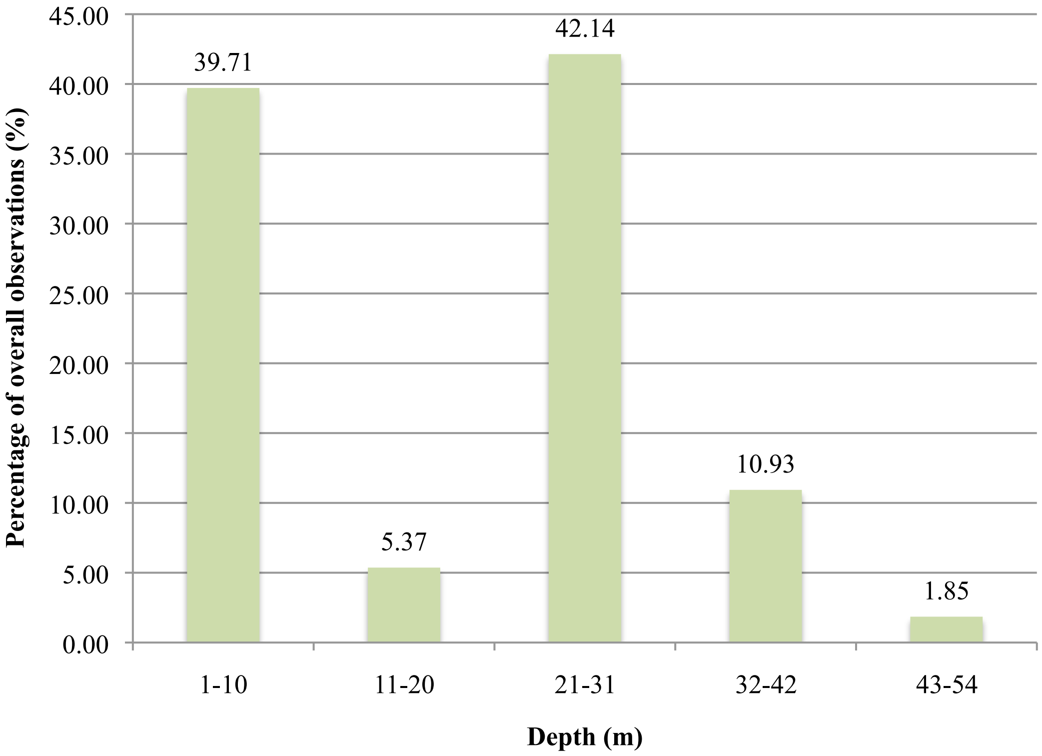


Figure 2.36: The proportion of observations black guillemots were observed at different depths, at the Fall of Warness.

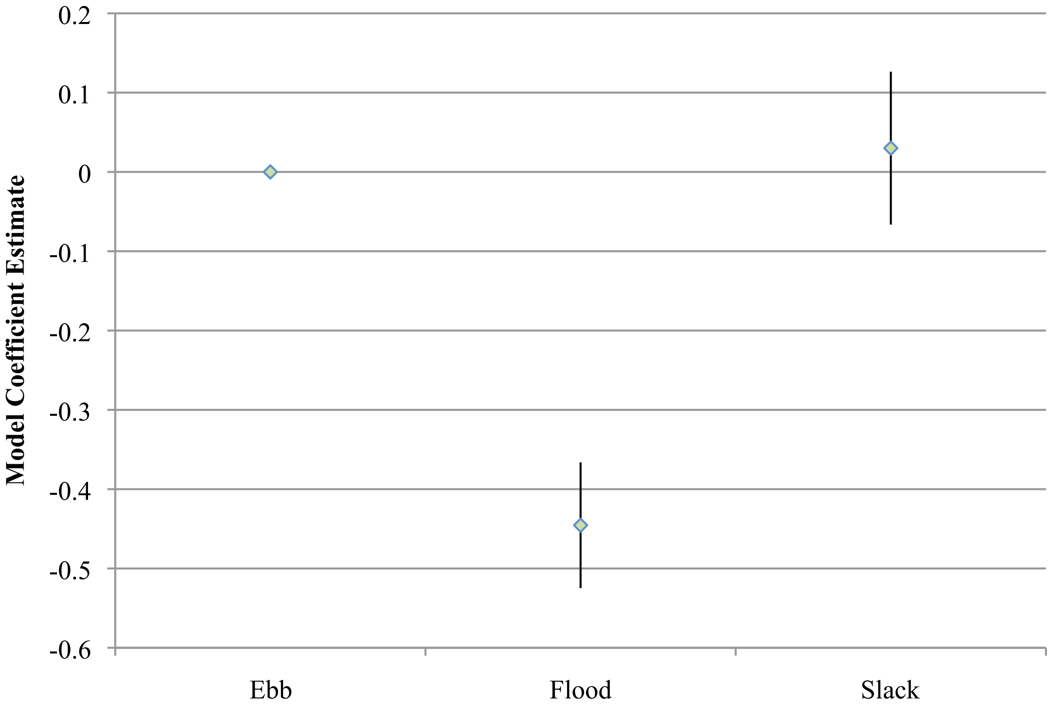


Figure 2.37: GEE coefficient estimates (and standard errors) for black guillemots observed by tidal state, at the Fall of Warness.

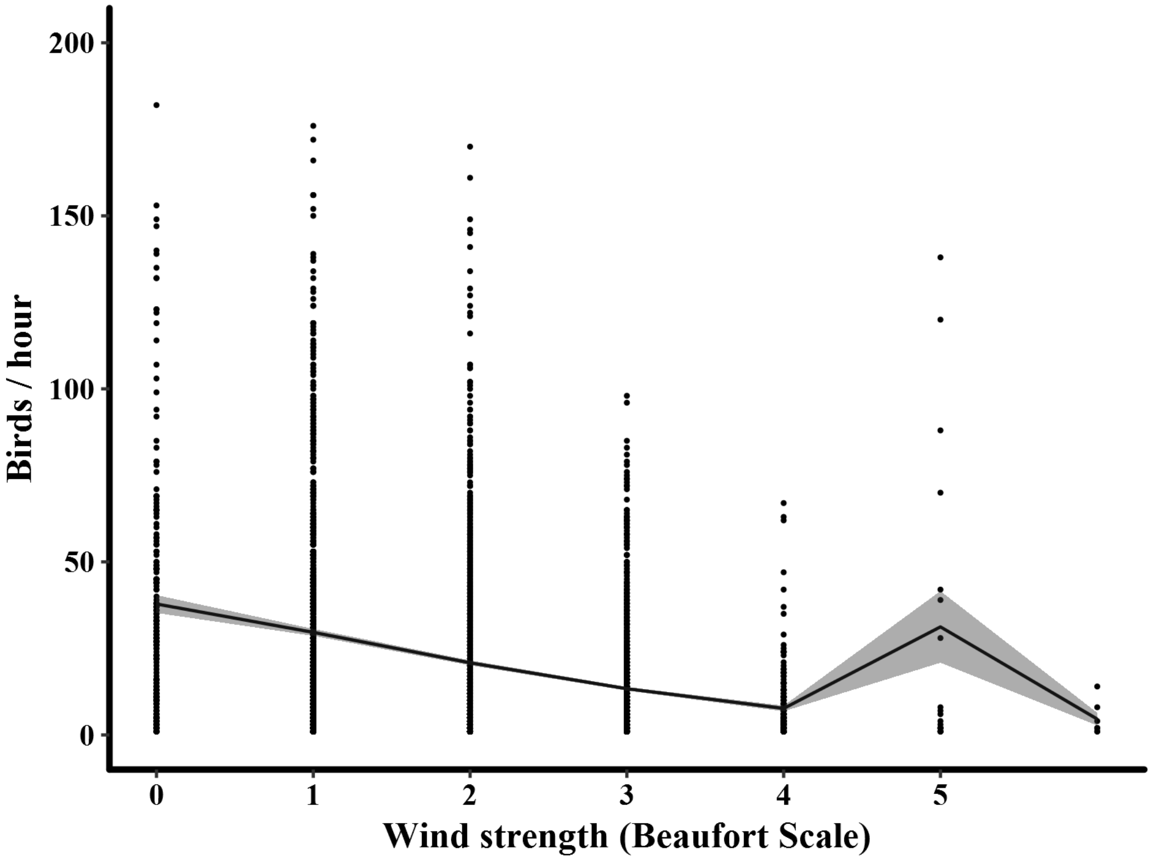


Figure 2.38: Mean number of black guillemots observed per hour during different wind strengths, using the Beaufort Scale, at the Fall of Warness, with raw data overlaid and standard error around the mean shown by the grey area. This figure was clipped to 95th percentile of the data.

2.3.8 Atlantic Puffin *Fratercula arctica*

A total of 7,882 puffins were observed at the Fall of Warness site. The top GLMM models indicate that puffin numbers are a function of season, time of day, tidal state, habitat type, depth and cloud cover (S. Table 1.8). The GEE model coefficient estimates (and standard errors) suggest that all these variables were significant, excluding tide state and cloud cover (Table 2.10). The GEE model estimated over dispersion (scale parameter) to be 1.47 and correlation between observations on the same day and grid squares to be 0.476.

Puffins were observed more frequently during the spring and summer months (Fig. 2.39) and mean numbers of puffins encountered per hour increased throughout the day peaking between 18:00 and 20:00, with 7-8 puffin/hour.

Puffins, like razorbills and common guillemots, were observed more in pelagic compared with coastal areas (Fig. 2.42), and 44.24% of all puffin observations were in water between 21 and 31 m deep. However, 37.74% of all puffin observations were in water of between 1 and 10 m deep (Fig. 2.43).

Table 2.10: Parameter estimates, standard errors, probability values for GEE investigating puffin counts as a function of season, time of day, tidal state, habitat type and cloud cover.

	Estimate	**Std. error**	**Wald**	**Pr (>|W|)**	**Signif.**
(Intercept)	0.438367	0.08944	24.02	9.50E-07	***
Spring	0.74193	0.05815	162.79	<2.00E-16	***
Summer	0.790093	0.058671	181.35	<2.00E-16	***
Autumn	0.180963	0.113705	2.53	0.11	
Time of Day	0.02141	0.00395	29.38	5.90E-08	***
Flood	0.019126	0.033163	0.33	0.56	
Slack	0.020291	0.042516	0.23	0.63	
Depth	-0.022173	0.001776	155.92	<2.00E-16	***
Pelagic	0.183599	0.039742	21.34	3.80E-06	***
Cloud Cover	-0.000315	0.00049	0.41	0.52	

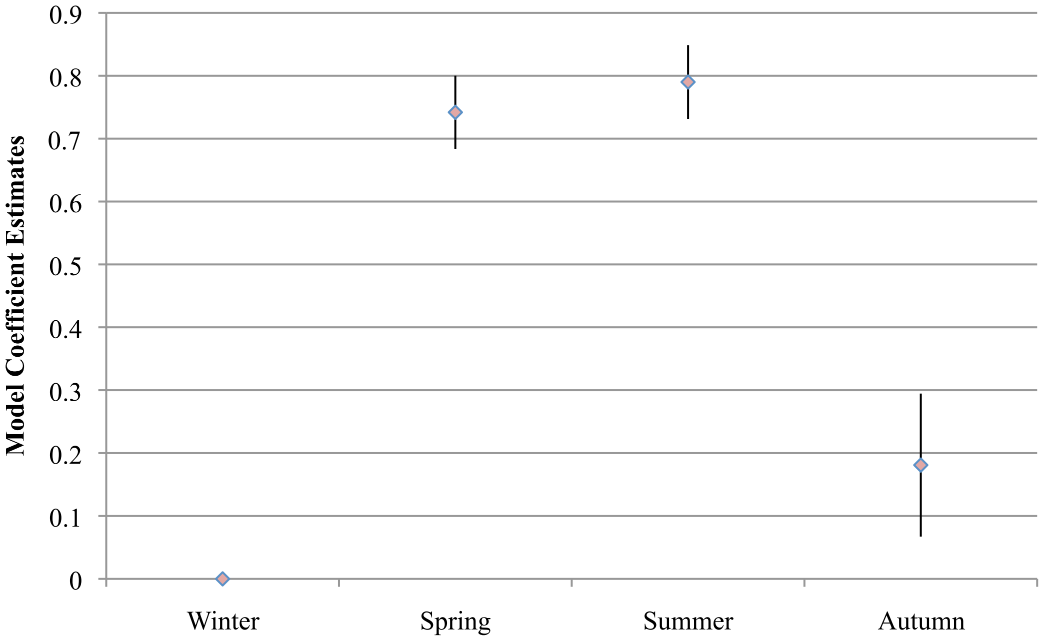


Figure 2.39: GEE coefficient estimates (and standard errors) for Atlantic puffins observed by season at the Fall of Warness.

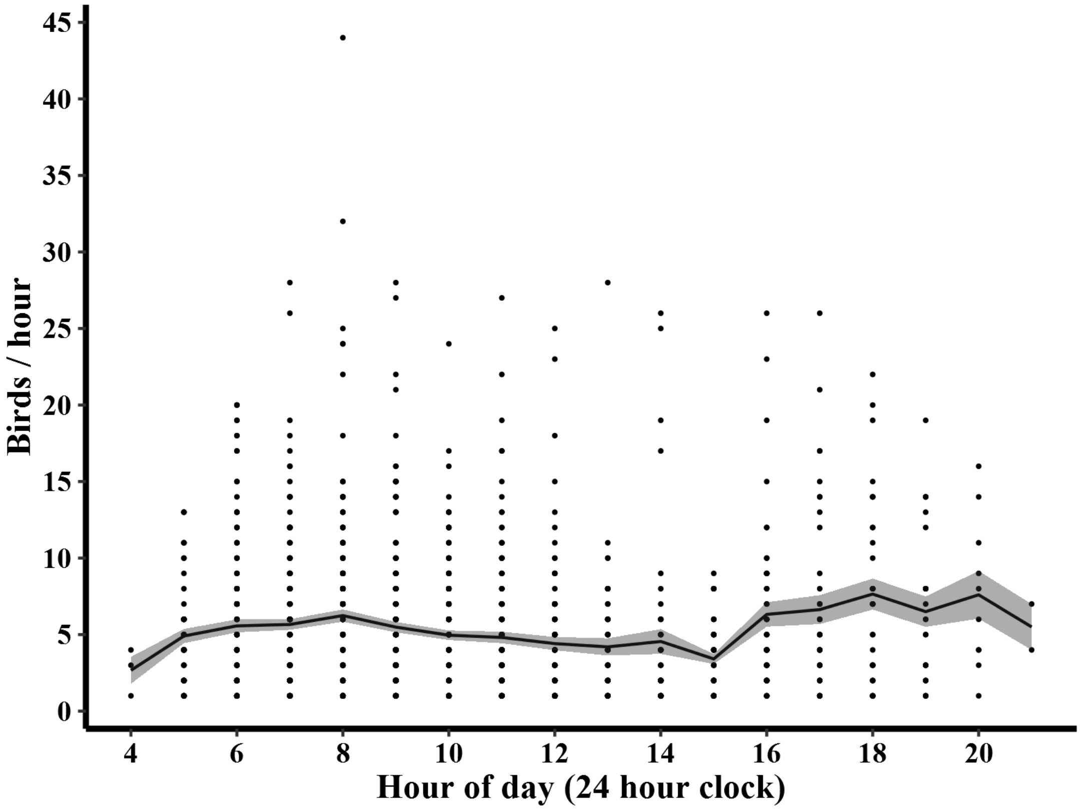


Figure 2.40: Mean number of Atlantic puffins observed per hour, throughout the day at the Fall of Warness, using the Beaufort Scale, with raw data overlaid and standard error around the mean shown by the grey area.

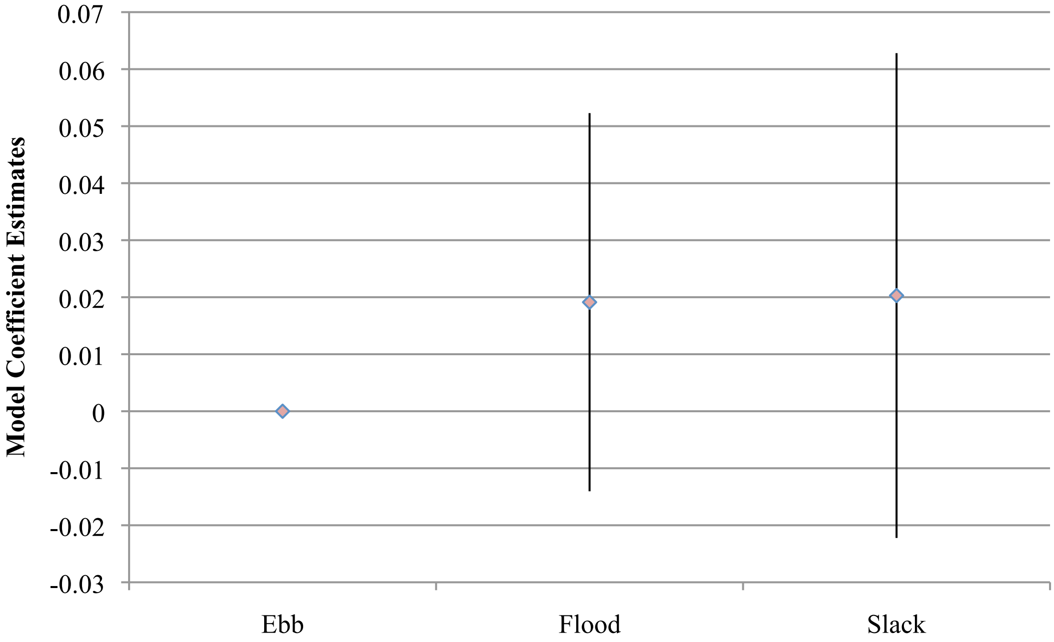


Figure 2.41: GEE coefficient estimates (and standard errors) for Atlantic puffins observed by tidal state, at the Fall of Warness.

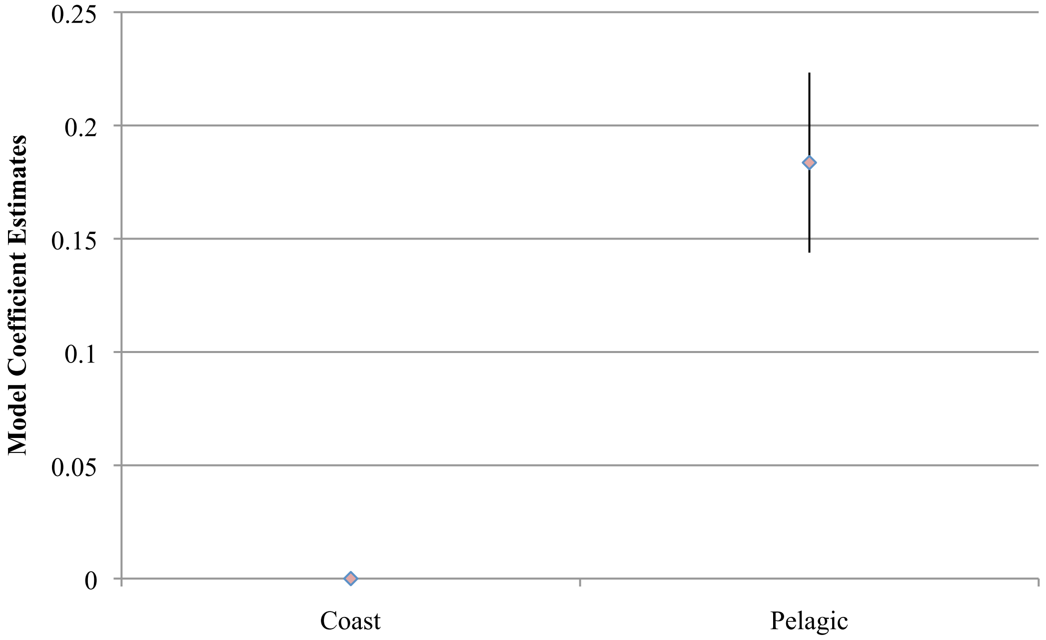


Figure 2.42: GEE coefficient estimates (and standard errors) for Atlantic puffins observed in coastal and pelagic habitats, at the Fall of Warness.

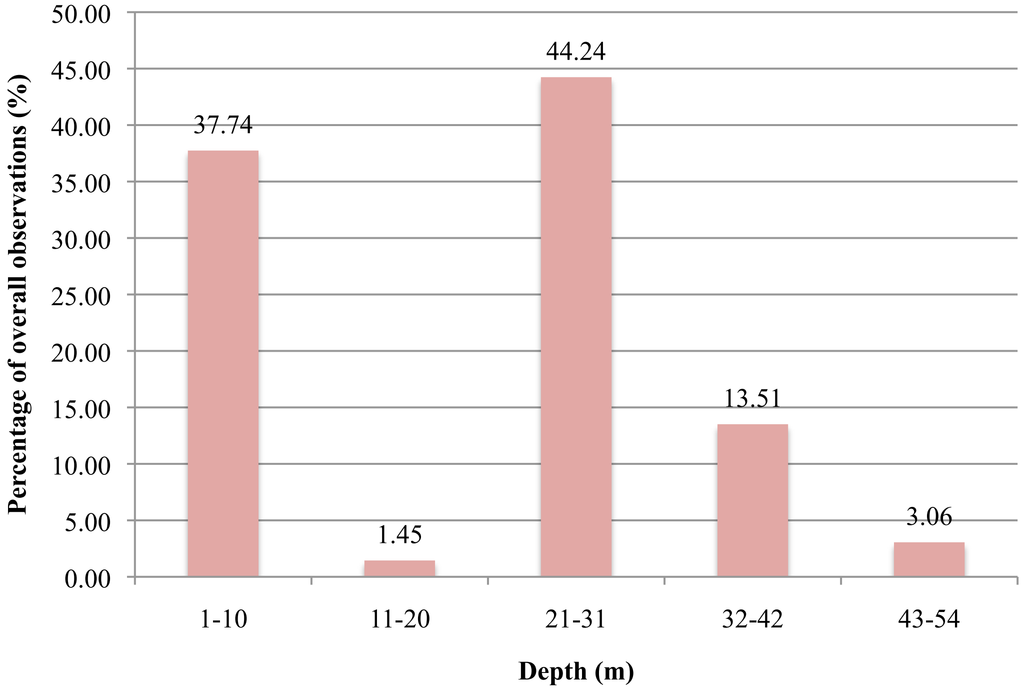


Figure 2.43: The proportion of observations Atlantic puffins were observed at different depths, at the Fall of Warness.

2.4 Discussion

This study provides a detailed baseline on site use and spatial and temporal distribution of the most commonly occurring bird species at the Fall of Warness EMEC tidal-stream test site. With this information we can identify where and when particular species are more likely to encounter test devices or related deployment activity. Consequently, it is then possible to consider potential interactions and any negative impacts that renewable energy devices could have on seabirds and their resource environment.

The spatial and temporal patterns of activity of marine birds at the Fall of Warness test site generally reflected the breeding and wintering habits that are typical for each species. Exploitation of the area varied among species according to seasons with more species present in spring and/or summer (seven of the eight species) and only cormorant spp. being absent during this portion of the year, preferring autumn and winter. This suggests different species are more vulnerable at different times of the year and would suggest that autumn and winter are the seasons when fewest species are likely to encounter devices. Similarly, time of day when activity was greatest (as determined by encounter rates per hour) varied among species with only three species demonstrating no obvious temporal trend. Of the remaining five species, early morning and/or early evening were preferred by three; eider the mid-morning and *Phalacrocorax spp.* from mid-morning to early afternoon.

Depth preference and habitat type are important in this study due to the nature of the tidal test devices. The tidal test berths are situated in a narrow channel between the Westray Firth and Stronsay Firth just west of the island of Eday in an area 2km across and approximately 4km in length, encompassing mainly pelagic grid squares. Therefore, the auk species, which generally preferred pelagic areas are more at risk from device interference or interactions. Similarly, with the tidal test devices set at depths ranging from 12m to 50m, the species, which more commonly use those depths, are more likely to be impacted. These deeper areas (21-30m) were important to the red-throated divers, gannets, common guillemots and puffins, however, it was not possible to explore foraging behaviour in these areas from the existing dataset. Interestingly, of these, the common guillemots and puffins primarily used the pelagic grid squares, which would put them at greater interaction risk. However, results for predominately coastal species such as eider were found to be consistent with literature that states that eiders are benthic feeders foraging close to shore and in water up to 4m deep (Owen et al. 1986). Therefore eider could be considered a much lower risk species for interacting with the devices at this test site.

Tidal state only influenced the distribution of four species with contrasting responses. Common guillemots and razorbills favoured flood tides while cormorants and black guillemots were encountered less frequently in this tidal state and cormorants showed a preference for ebb tides.

Weather variables were selected for for a number of species. While it is possible that birds behave differently under different weather conditions (such as wind influencing flight behaviours), it is also likely that these variables affected the observers' ability to record. For example, wind direction being selected for in the razorbills models may be because westerly winds affect the sea-state in this area, causing "choppiness" which in turn could reduce the observers' ability to observe and/or identify razorbills. Most species also showed a decline with increasing wind strength, which is also likely linked to a deteriation in sea state and ability for the observers to detect animals on the sea surface.

The results and outcomes of this study highlight the aspects in which particular species are more at risk from interactions with wet renewable energy systems such as tidal-stream devices. Knowing the likely situations in which seabirds are at greater risk of interactions with test devices can directly inform assessment of potential impacts from a particular development and aid decisions regarding location, season or time of operation of tide energy systems, helping to minimise risk to seabirds. However, it should be noted that as with any observational study the results may not necessarily applicable to periods were no observations have been carried out, for example usage of the test site during the night is still unknown.

2.4.1 Recommendations

Part of the variation in the results of this study demonstrates some of the inherent concerns incorporated in shore-based observational data.

Firstly, data were limited to diving species and therefore it was not possible to consider the use of this area by surface feeders, such as terns or kittiwakes, nor potential impacts to these species. Data were also limited to those interacting with the surface of the water. I strongly recommend that all species are included as part of the data collection as there may be a future need to explore potential impacts on species that are not currently considered to be particularly vulnerable to tidal developments (Furness et al. 2012), e.g. indirect impacts to surface feeders, which otherwise cannot be explored.

Second, there is a decreased probability of detecting wildlife with an increase in distance from the observation point (Bibby et al. 2000; Buckland et al. 2001). Given the scale of the test site there will a detection bias due with birds going undetected further away from the vantage point. However, it is currently not possible to include this within analyses, as it cannot be disentangled from the ecological gradient that radiates from the shoreline (and observer). The grid system used also prevents the inclusion of detection probabilities as the spatial accuracy of the observations is much coarser and it is possible that the usages of the most distant grid squares (from the observer) are being underestimated. The grid system also prevents any analysis of clustering or group size, which may also affect detection probabilities. It is therefore recommended that boat-based surveys undertake line-transects randomly across the test site to calibrate these land-based vantage point observations. These surveys should be carried out according to standardized methodologies (e.g. Buckland et al. 2001 and Camphuysen et al. 2004). Calibration through another observation platform such as a boat will enable areas further away from the vantage point to be surveyed without any distance detection bias and therefore give some indication of how large a bias it is for this dataset. It was also identified during discussion of methods, that while posts were used on the Eday coastline to help define the grid, it would useful to calibrate this with a map to ensure consistency between areas observed and recorded. Similarly the -1 row did not have a clear boundary and therefore the area grid cells in this row may have varied with visibility conditions.

Thirdly, in some cases clear patterns were observable in species' spatial distribution but were not significant in the models. This could be a product of observer positioning or the counts may have been too low for the model to determine an effect.

Fourth, the effects of wind and cloud conditions on visibility in ocean environments could have affected how well observers detected or differentiated seabirds. The observers also noted that the location and bearing of the vantage point meant that certain times of day were more likely to be subjected to poor glare conditions, which resulted in some time periods being sampled less frequently. The ideal solution for this would be to incorporate these weather variables into a detection correction instead of the approach that was used. It would also be appropriate to account for variation between observers (even of the same experience and skill) in analyses, by including 'Observer ID' as an additional parameter in the models. In addition, including a defined number of categories for both visibility and precipitation variables, would enable inclusion of visibility within future analyses, and may also enable more informative modeling of precipitation. Similarly it was not possible to fully account for observer effort as the exact timings of each scan were not recorded, so it had to be assumed that all observations followed the protocol.

Finally, there is currently, only very limited data available on the habitat types and depth within the Fall of Warness monitoring site (Finn 2009) and these data are not available for every grid square. Therefore it was not possible to consider, for example, the seabed substrate or slope, which may influence which species forage where within the site. This information and other habitat variables may be useful in future analyses. Similarly, this analyses did not include concurrent tidal data, however given the speeds of current flow at this site, and other areas identified for development, it would be useful to explore the effects of current on both the bird usage but also detection of birds in a mobile environment. For example, density estimate calculation may be influenced by birds resting on fast flowing water.

2.5 Conclusion

This study provides a comprehensive model upon which to formulate further studies of this test site and other proposed renewable energy sites in order to understand potential impacts to animals using the marine environment. This chapter has identified a number of methodological issues, such as the use of a grid and limiting species data are collected on, and made some recommendations for how data collection at the test sites can be improved. These recommendations should enable inaccuracies within the dataset, as a result of methodological biases, to be more clearly identified and enable these data to be utilised with a greater understanding the limitations. These steps should also then improve the ability to predict interactions between marine bird species and test devices or related deployment activity, improving future ecological assessments and development of the tidal test site going forward.

Chapter 3: Monitoring marine bird use of a wave-energy test site: an Orkney case study

This chapter has been published as a commissioned report to Scottish Natural Heritage as ‘Robbins, A. 2012. Analysis of Bird and Marine Mammal Data for Billia Croo Wave Test Site, Orkney. *Scottish Natural Heritage Commissioned Report No. 592’*.

This chapter comprises data owned and collected for EMEC as part of their wildlife monitoring programme at Billia Croo. I have undertaken all the analyses of these data presented within this chapter. Marine mammal data included in the original analysis have been omitted and some of the text and figures has been amended to fit within the context of this thesis.

Supplementary materials for this chapter, including tables and distribution maps are in Appendix 2.

Abstract

Marine renewable energy arrays, such as wave and tidal energy, are planned for near-shore areas potentially used by marine birds. Therefore, it is important to utilise a robust approach to data collection on the distribution, abundance and species composition of seabirds using potential development sites, in order to determine if these birds interact with marine renewable energy devices, and whether such interactions would be detrimental to the birds and their populations.

In this chapter I explored the nature of any relationships between the more frequently observed bird species' site usage and environmental variables at the European Marine Energy Centre (EMEC) wave-energy test site at Billia Croo, Orkney. The spatial and temporal distributions of fourteen bird species were monitored from 2009 to 2011 following protocols established by EMEC. In contrast to the previous chapter, I investigated the influence of environmental variables including wind, tide and glare modelled using GAMMs.

Spatial and temporal patterns of the Billia Croo wave-energy test site use varied among species. Many species showed slight differences in the locations of 'hotspots' and the extent to which they used the wave test site. Observed temporal variations, such as clear and specific diurnal patterns of site use were observed in fulmar, gannet, Arctic tern, black guillemot and puffin. The influence of environmental conditions was limited to tidal state, glare extent, wind strength and direction and responses varied among species.

This study provides the first comprehensive baseline information on the spatial and temporal distribution of the most frequently occurring marine bird species at the Billia Croo wave-energy test site. We can use this information to understand and identify where and when particular species are more likely to encounter test devices or related deployment activity. This could inform decisions regarding time of deployment or operation of wave energy converters so as to minimise impacts to seabirds.

Improvements to data collection protocols can also be identified to improve future analyses of similar data sets.

3.1 Introduction

The marine renewable energy industry is undergoing rapid growth in Scotland due to Scottish Government targets of meeting 100% of energy needs through renewable sources by 2020. However, the wet renewable sectors, wave and tidal-stream energy are still in their infancy. There are currently few marine renewable energy devices (MREDs) deployed in waters around the UK, apart from test sites such as the European Marine Energy Test Centre (EMEC) in Orkney. Scottish islands, such as the Orkney archipelago are renowned for harsh conditions and EMEC is one of the first centres of its kind to provide a purpose-built, accredited, open-sea testing facility to developers of MREDs.

The Billia Croo test site is located on the western edge of the Orkney mainland, is subjected to the powerful forces of the North Atlantic Ocean, and has one of the highest energy potentials in Europe with an average significant wave height of 2-3 metres, but reaching extremes of up to 17m. The site consists of five cabled berths in up to 70m water depth (four at 50m and one deeper), which are located approximately 2km offshore and 0.5km apart. There is an additional near shore berth closer to the substation for shallow water projects (EMEC 2016c).

Wave-energy developments are planned for areas that are potentially used by, and important to marine birds. As such they could interfere with foraging and at-sea activities of these birds or how they utilise the environment or even restrict or preclude seabirds' access to necessary resources. It is therefore essential to obtain robust data on marine bird species in potential development areas and how they use the marine environment so it can be determined if they interact with, or are otherwise affected by, the proposed devices. This will allow us to begin to understand whether such interactions could be detrimental to seabird populations.

Many seabirds inhabit and use the marine environment of the Orkney Islands, (Mitchell et al. 2004; Forrester et al. 2007) and the location of Billia Croo makes it ideal to conduct land-based observations of seabird species and their patterns of activity. By doing so, it is possible to improve the general understanding of species composition in these high-energy environments, ascertain potential interactions with wave energy converters and how these interactions they may affect the birds. Until now, there has been very limited analysis on the wildlife monitoring dataset collected at the Billia Croo test site.

This chapter aimed to obtain a baseline on the spatial and temporal distribution, abundance and species composition of seabirds that use the EMEC Billia Croo wave-energy test site; identify those most commonly occurring; and explore relationships (if any) between the more frequently observed bird species' site usage patterns and habitat and environmental variables such as water depth, wind strength or direction, tidal state and cloud cover. This study will set the foundation and provide a better understanding of appropriate methodologies, wildlife monitoring protocols and data management that can be used in future studies determining the level and type of interaction between seabirds and tidal energy systems.

3.2 Methods

3.2.1 Data collection

Following protocols established by EMEC, land based seabird observations take place from a look-out shelter on Black Craig, Billia Croo (58˚58.746'N 03˚21.499'W), approximately 110m above sea level. These data were collected on behalf of EMEC by two experienced surveyors. Marine bird observations commenced at Billia Croo on 15th March 2009 and data used here are up to 15th March 2011. A four-hour watch format, 5 days per week was used, timetabled to cover different tidal states and times of day (see Table 3.1 for total hours per month). Only observations within 5km of the vantage point were used. The rota was designed to ensure relatively even coverage across daylight and tidal states. Where watches were unable to be carried out due to weather they were typically undertaken at the next weekend or following opportunity.

The observers scanned the site using a pair of fix-mounted 25x power binoculars (Big eyes) to detect birds on the surface of the water. These were mounted on a tripod with horizontal and declination angle boards to estimate geographical locations of animal sightings. The 'Big eyes' were housed inside a coastguard lookout station overlooking the Billia Croo test site. The observation scans were carried out in a consistent manner from left to right at a series of distances from land ensuring the whole study area was covered. The study area was fully covered with two sweeps using the 'Big eyes' (a near and a far sweep) and an ad-hoc sweep of the near-shore area using binoculars. The observer aligned the 'Big eyes' at the appropriate declination and horizontal angles for the 'far' area (1400-5000m from the shore) and sweep from left to right. Once complete, the observer then re-aligned to the appropriate the declination and horizontal angles to sweep the mid area (800- 1400m). Once the mid area sweep is complete, the observer would use binoculars to sweep the near area (shore -800m) (refer to figure 3.1b). Scans were estimated to take approximately 20 minutes, with a 10 minute period of rest between scans to reduce observer fatigue and allow for data entry. Two scans were carried out per hour, with 8 scans per watch/40 per week (further details of the timetable are outlined in the data collection protocol; SMRU Ltd, 2008).

Birds were only recorded as sightings when they were on the water or hovering directly above. Groups of animals were defined as all animals within approximately 100m of each other. If groups of animals were recorded then the location was based on the centre of the group. Observers recorded bird behaviour as:

Diving from flight	One or more birds diving underwater from a hovering or flying position.
Diving from water	One or more birds diving underwater from a position on the water surface.
Swimming at surface	The birds are making progress at the surface.
Stationary at the surface	The birds are stationary at the surface.

3.2.2 Data Preprocessing

These data required significant preparation prior to analysis, which included:

- Alteration of misspelled species names and categorical variables.
- Sorting of environmental and observation data including matching the correct environmental observations with sightings.
- Separating bird records where mixed species had been seen in a group and the numbers of individual species were recorded only in a comment/text string.
- Re-calculation of latitude and longitude for observations, using the recorded declination and horizontal angles.
- Calculation of additional variables such as “time from low tide” and “time lapse”.

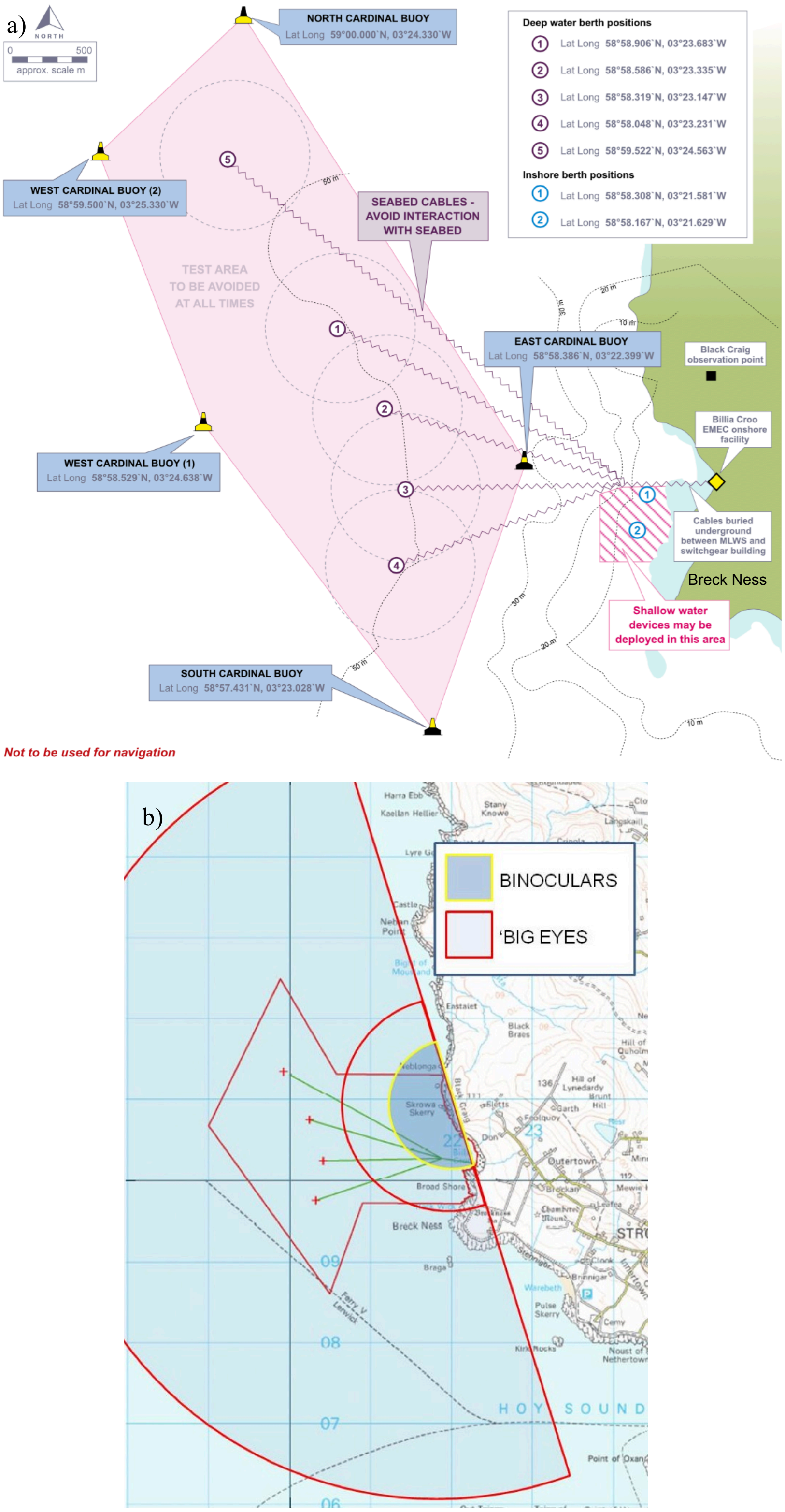


Figure 3.1 a) Map of the EMEC Billia Croo wave-energy site, showing the berth positions and Black Craig observation point (Adapted from EMEC Wave Test Site Awareness Chart: http://www.emec.org.uk/facilities/wave-test-site/#) **b)** Map of the EMEC Billia Croo wave-energy site showing the study area, extending to approximately 5km offshore. The area is divided in 3 sub areas (Near, Mid, and Far) that were scanned by the 'Big eyes' (Mid and Far sub areas) and binoculars (Near sub area). SMRU Ltd, 2008

Table 3.1: The total number of hours of survey that marine birds were observed in between 15th March 2009 and 15th March 2011.

Month	2009	2010	2011
January		55	79
February		72	57
March	55	90	44
April	107	85	
May	90	86	
June	93	79	
July	101	80	
August	81	86	
September	76	77	
October	55	83	
November	64	81	
December	48	67	
Total	**770**	**941**	**180**

3.2.3 Data Analysis

All data analyses were completed using the statistical data package R, with relevant packages (R Development Core Team, 2011; see below in modeling section 3.2.3.3). For all bird species with sufficient data, I used the species count data as the response variable to investigate the influence of the different habitat and environmental conditions (Table 3.2). In addition, I mapped seasonal and behavioural distributions of bird species (and groups of species) in ArcGIS version 9.3 for Desktop. Due to the commercial sensitivity of the test site devices, specific dates when these devices were present or operational were not available at the time of analysis and could not be included.

3.2.3.2 Co-linearity

Similar variables were tested for co-linearity using Variance Inflation Factors (VIF). Sea state, swell height and wind strength were all found to have co-linearity. Wind strength was found to be the most representative variable and was subsequently used in the analysis. Wind strength was used as a proxy for sea state, the method for recording both was using the Beaufort scale, an empirical measurement, which is used to describe wind intensity based on observed sea conditions. Although, noting that the numerical values between the two differ due to the lag effect between the wind strength increasing and the sea state increasing (Met Office, 2016).

Table 3.2 Environment variables monitored at Billia Croo wave-energy test site.

Environmental Variable	**Description**
Wind Direction	This was subdivided into the following five categories "North", "South", "East", "West" and "None". Due to the latter two categories this variable was not treated as a circular variable.
Wind Strength	This was defined by observations using the Beaufort scale descriptions.
Sea State	This was defined also by observations using the Beaufort scale descriptions.
Cloud Cover	This was recorded as a percentage.
Weather	Weather was recorded as "Fair", "Rain" or "Snow".
Glare Extent	This was subdivided into ordinal categories "None", "Slight", "Moderate" and "Severe".
Tidal state[1]	This was subdivided into "High Slack", "Ebb", 'Low Slack" "Flood".
Time from Low Tide	This was the time calculated (in decimal hours) from Low Tide.
Julian Day	Is the ordinal date for the day the observation occurred (with the year omitted).
Season[1]	The months were grouped in to "Winter" (December, January and February); "Spring" (March, April and May); "Summer" (June, July and August) and "Autumn" (September, October and November).
Time of Day	This was recorded as the hour in which the observation occurred, i.e. 10:30 was "10" and 14:15 was "14" using GMT. This covariate was fitted as a smooth.
Daylapse	The number of days from the start of the data collection, where the first day is 1 and 380th is 380.
Observer ID	This was defined by the initials of the two observers or "NA" if unknown.

[1] Ordinal variables were treated as factors.

3.2.3.3 Modelling

I modelled marine bird abundance within the EMEC Billia Croo wave test site using an extension of generalized additive model ("GAM") techniques. These additive models can be used when data are non-linear, and a transformation of the data is inappropriate. They allow for non-linear relationships between response and explanatory variables through the use of smoothing models, which can be applied to non-linear data (Zuur et al. 2009). Generalized additive mixed models (hereafter "GAMM") were used with Negative Binomial errors (and adjusted theta) and log link functions. The dataset was zero-truncated and therefore the analysis was of presence only data. The model interpretation therefore represents the impact of environmental covariates on the absolute number of birds. To account for temporal auto-correlation, i.e. for counts that occurred on the same day and may be correlated, generalized estimation equations (hereafter "GEE") were used, which enabled estimation of robust standard errors. The AR1 correlation structure was applied to the daylapse variable. Continuous variables were modeled as splines (i.e. Lat/Long, Julian day, time of day and time from low tide) or linear terms (i.e. wind strength and cloud cover) and categorical variables were added as factors. Variable significance was calculated using GEE-based *p*-values.

The results reported include the GAMM model coefficient estimates, GEE-based *p*-values and standard errors. The plots presented within this report incorporate GAMM model coefficient estimates, the encounter rate of mean number of birds observed (per hour) or percentage of overall observations. The higher model coefficient values represent a greater number of predicted birds.

GAMM model validation was undertaken by plotting and reviewing the distribution of the selected models residuals and fitted values. However, residuals of the estimated models still showed patterns, which were more pronounced in those species with fewer observations. Consequently, the results within this report should be considered with caution. Due to this issue a higher significance level has been applied to the terms in the model, i.e. where p is <0.01.

3.3 Results

Abundance and the spatial and temporal distribution of seabirds observed between March 2009 and March 2011 at Billia Croo wave test site varied among the 14 species considered in this study. With the exception of Arctic skuas, several environmental variables, predominantly wind, tides and glare conditions, influenced all remaining species but responses varied among species. Model results are presented for 11 species and 1 species group, . along side the pertinent figures for each. Further model results and figures, including maps for each species are in Appendix 3

3.3.1 Common Eider *Somateria mollissima*

A total of 2,635 eiders were observed between March 2009 and March 2011 at Billia Croo wave test site. The distribution maps (S.Figs. 2.1 & 2.2) highlight a clear spatial pattern, with all sightings occurring in the southern part of Billia Croo test site, close to the shoreline around Breck Ness.

The best predictive model had a small amount of auto-correlation as $\rho = 0.06$ but showed significant seasonal variation in numbers ($p = <0.01$, Supplementary Table 2.1, Fig. 3.2). Eiders were most frequently observed around February (Julian Date 32), after which the numbers rapidly decreased until June (Julian Date 152; Fig. 3.2), after which numbers once again began to increase from July. Although time of day was not selected for, the mean number of birds encountered per hour suggests that feeding increased during the morning, with a peak mean of 5 birds at 10:00 (Fig. 3.5). The mean remained above 2 birds per hour for the duration of the day. Greater numbers of birds were observed resting, compared with feeding, and the mean number of resting birds encountered peaked at 15 birds in the evening (18:00).

Eider sightings varied under different tidal states ($p = 0.03$), and they were more frequently observed during ebbing tides and significantly less frequently in flooding tides ($p=<0.01$) (S.Tables 2.1 and 2.2; Fig. 3.3). In addition, the proportion of eiders observed resting and feeding was found to differ between tidal states (Fig. 3.6). Overall 2,346 eiders were observed resting and only 254 eiders were recorded feeding: 4% of eiders were observed feeding during ebbing tides, while only 1% were observed feeding during slack tide. Conversely, 34% of eiders were observed resting during low slack tide and 17% of eiders were observed resting during high slack tide and also flood tides (Fig. 3.6). Finally, when comparing among the different levels of glare (low, moderate and severe) significantly more eiders were observed under moderate glare conditions ($p=0.01$) (S.Tables 2.1 and 2.2; Fig 3.4).

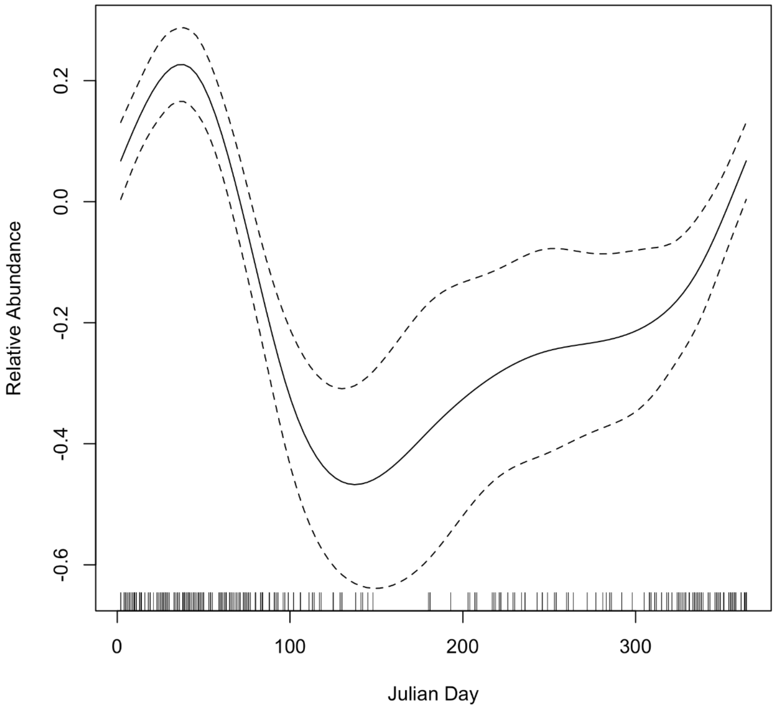


Figure 3.2: The estimated seasonal pattern of relative number of common eider observed. The solid line is the smoothing curve for Julian date and dotted lines are 95% confidence bands.

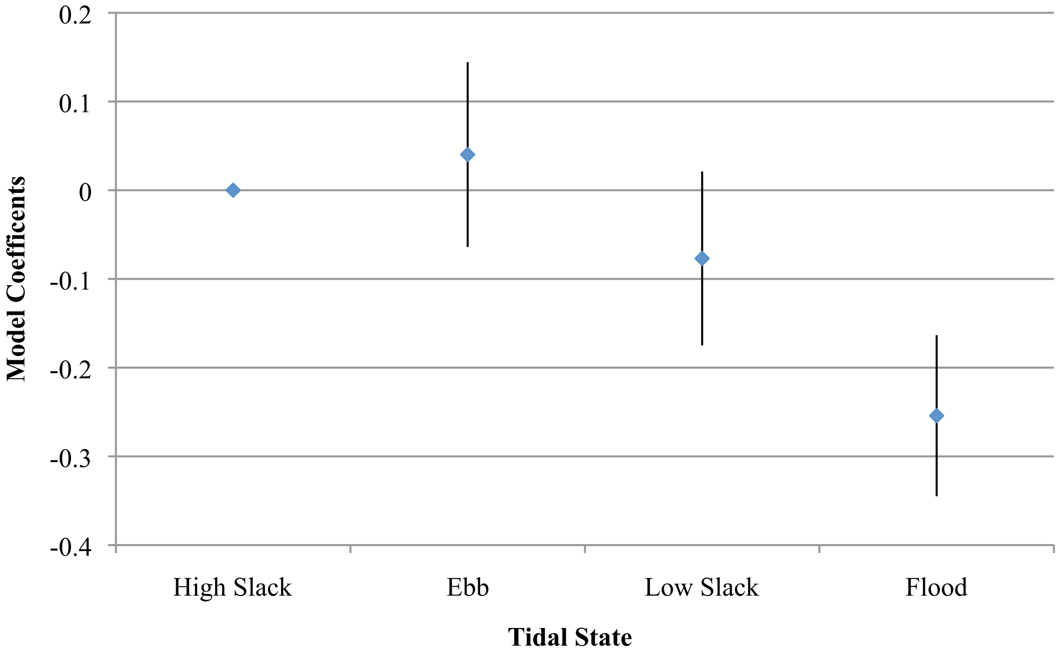


Figure 3.3: GAMM coefficient estimates (and standard errors) for common eiders observed by tidal state at Billia Croo.

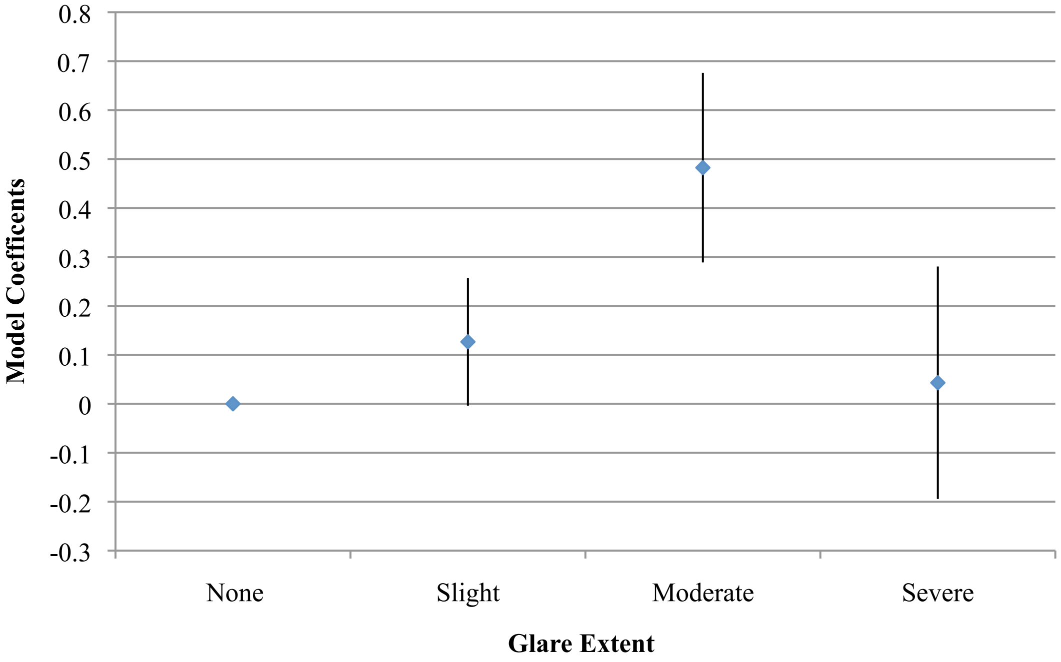


Figure 3.4: GAMM coefficient estimates (and standard errors) for common eiders observed by glare extent at Billia Croo.

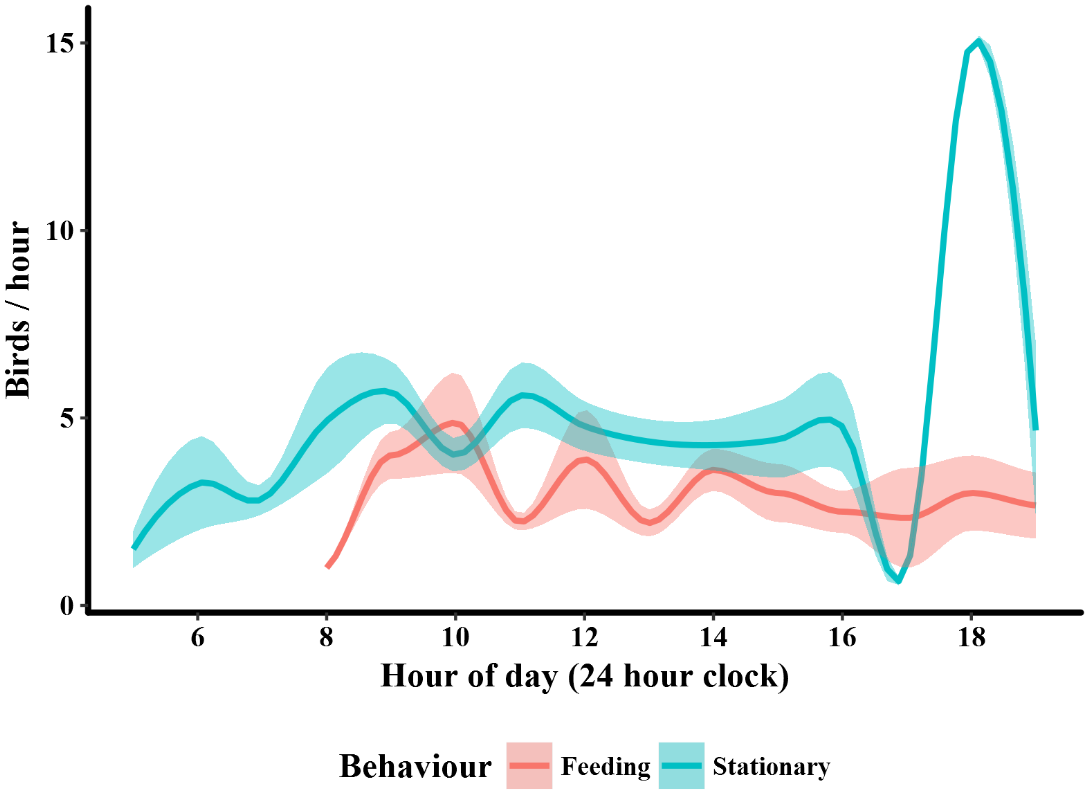


Figure 3.5: Mean number of feeding and resting common eider observed per hour, throughout the day at Billia Croo, with standard error around the mean shown by the shaded areas.

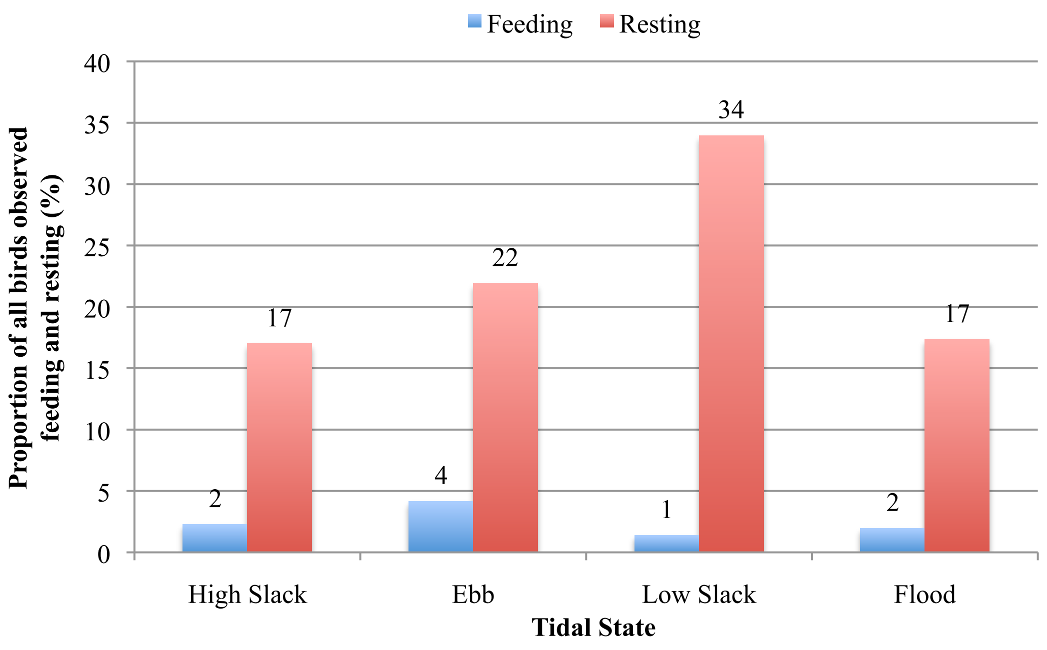


Figure 3.6: The proportion of observations of feeding and resting common eider observed at different tidal states, at Billia Croo.

3.3.2 Red-Throated Diver *Gavia stellata*

A total of 50 red-throated divers were observed at Billia Croo during the observation period, which were too few to perform robust models. Nevertheless, the distribution map (S.Fig. 2.3) shows red-throated divers were primarily sighted in the southern and western areas of the site, with the majority of birds observed within 1,000m of the coastline. The map also indicates a spatio-temporal change in distribution across this site, with summer (breeding season) sightings being further from the coastline. Red-throated divers also displayed seasonal differences in numbers of individuals encountered at Billia Croo (S.Fig. 2.6), with the greatest numbers of birds observed during spring and winter (42% and 34% of sightings, respectively). Only 4% of red-throated divers were observed in the autumn. Sightings also fluctuated according to time of day with mean numbers peaking at 6 hourly intervals: 5:00, 11:00 and 17:00 (Fig. 3.7).

Time from low tide was the only environmental variable that shows some influence over red-throated diver distributions. The highest mean number of birds occurred 1 hour after high tide (5 hours before low tide), while no birds were observed 1 hour before high tide (5 hours after low tide) (Fig. 3.8). The figure also highlights secondary peak of encounters 2 hours after low tide.

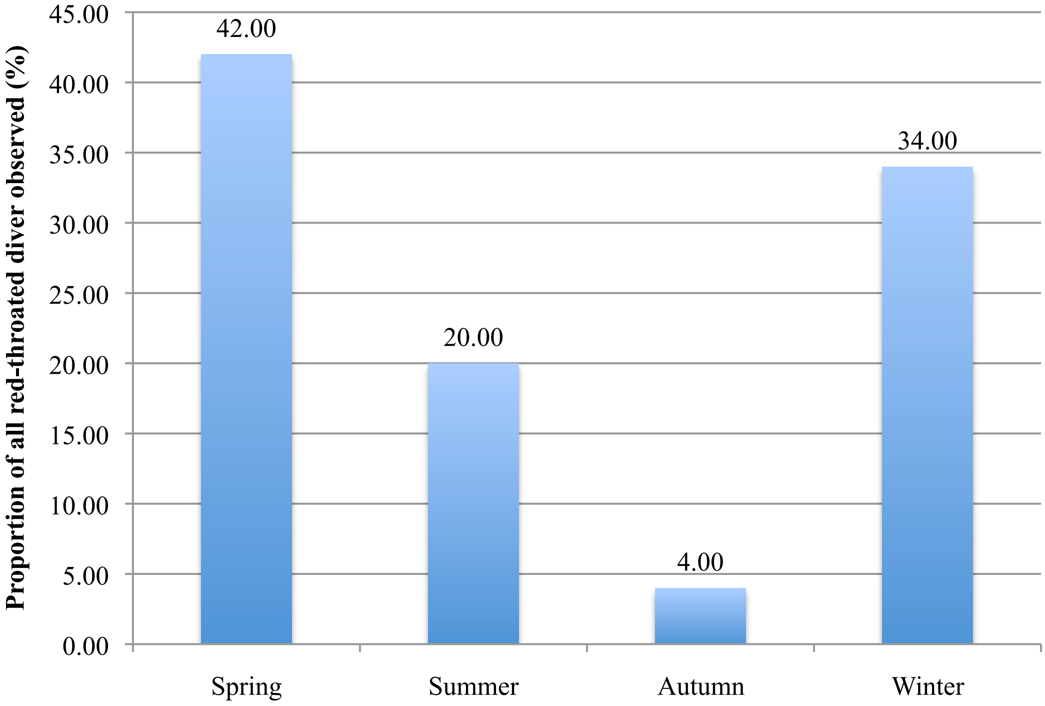


Figure 3.7: The proportion of red-throated diver sightings by season at Billia Croo.

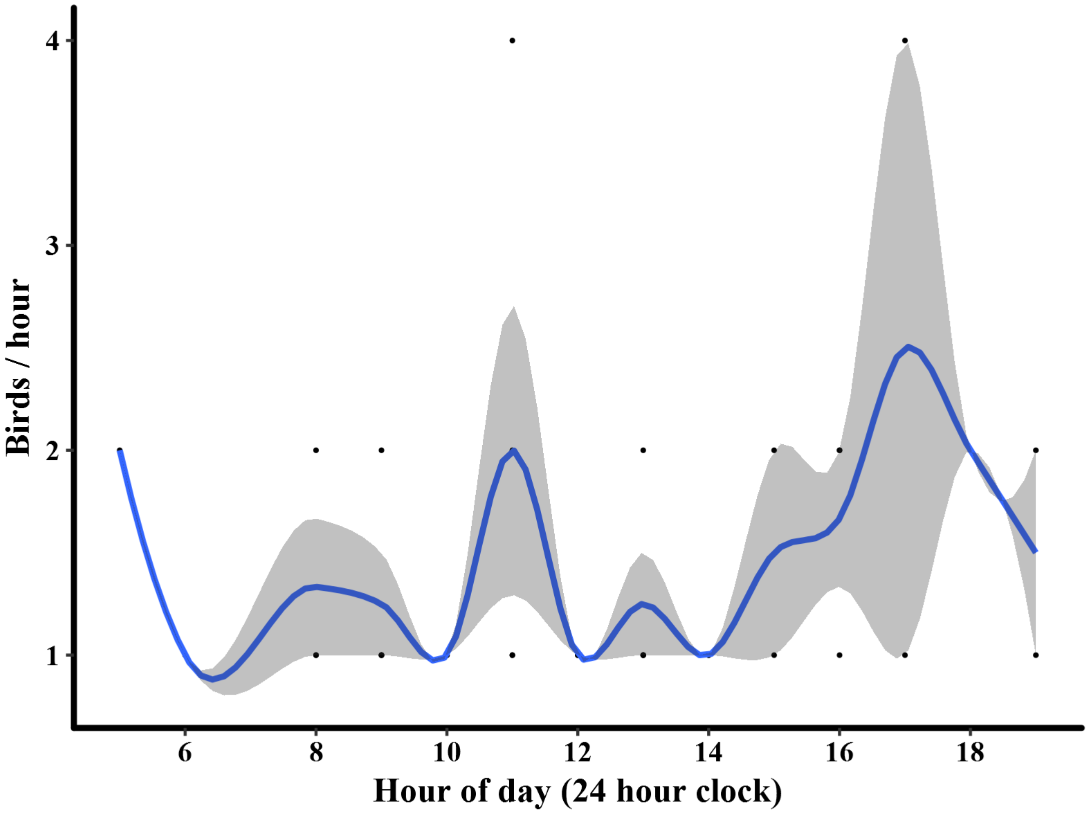


Figure 3.8: Mean number of red-throated divers observed per hour, throughout the day at Billia Croo, with raw data overlaid and standard error around the mean shown by the grey area.

3.3.3 Northern Fulmar *Fulmarus glacialis*

54,576 fulmars were recorded within 5km of the vantage point during the observation period and distributed across the Billia Croo test site, but with a concentration of records immediately to the west of the Black Craig observation point (S.Figs. 2.4 & 2.5) The selected model, which had some auto-correlation (ρ = 0.09), highlights this same concentration with standard error lines for higher abundances at the southern end of the site and a highly significant spatial smooth term (p=<0.01) (S.Tables 2.3 & 2.4; Fig. 3.9).

The greatest numbers of sightings were recorded in the western part of the site, adjacent to the coastline, while the highest numbers of fulmars per sighting were observed in the southern section. Peak fulmar sightings were in December, during the wintering period, after which numbers decreased until September (approx. Julian day 250; Fig 3.10). The two small increases in numbers around May and August coincide with the onset of laying and the fledging period respectively. Fulmar observations also demonstrated temporal variation throughout the day with a steady increase in the morning, to a peak around mid-day, followed by a steady decline until the evening (Fig. 3.11).

Fulmars use the site predominantly for resting/stationary behaviours: 97.76% of the fulmars for which behaviours were recorded were observed resting, with only 0.38% feeding and 1.86% travelling. Figure 3.12 explores the diurnal fluctuation in sightings further, by separating fulmars that are feeding and resting. The mean number of birds/hour observed resting follows the same diurnal pattern, with a 2-3 hourly cycle of peaks and troughs. However, the mean encounter rate for feeding fulmars shows two periods of increased encounters, the first at 08:00 (16 birds/hour) and the second at 16:00 (13.5 birds/hour).

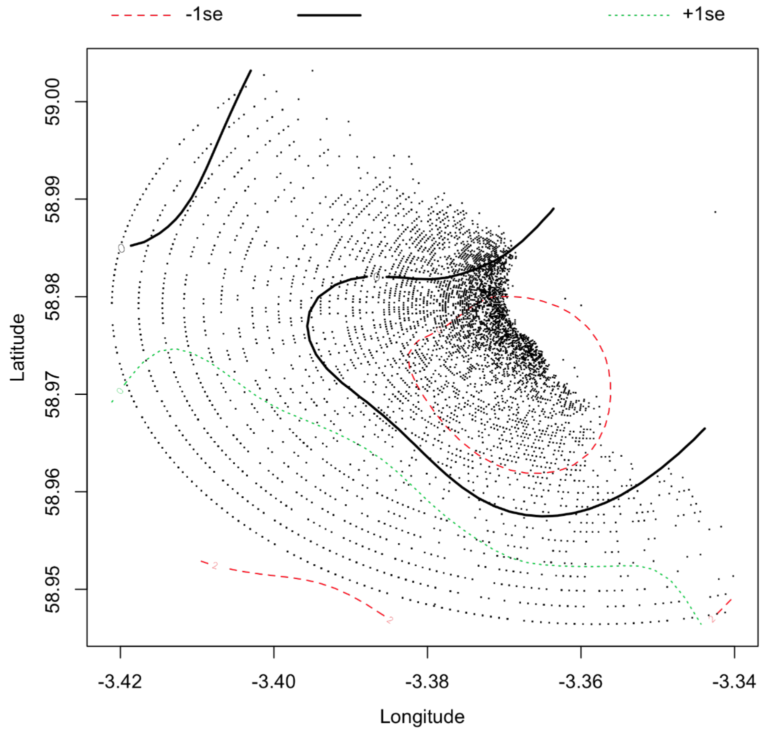


Figure 3.9: The estimated spatial pattern of relative number of northern fulmar observed. The solid line is the smoothing curve for 0, red dotted lines are -1 standard error from the smoothing curve and the green dotted lines are +1 standard error from the smoothing curve.

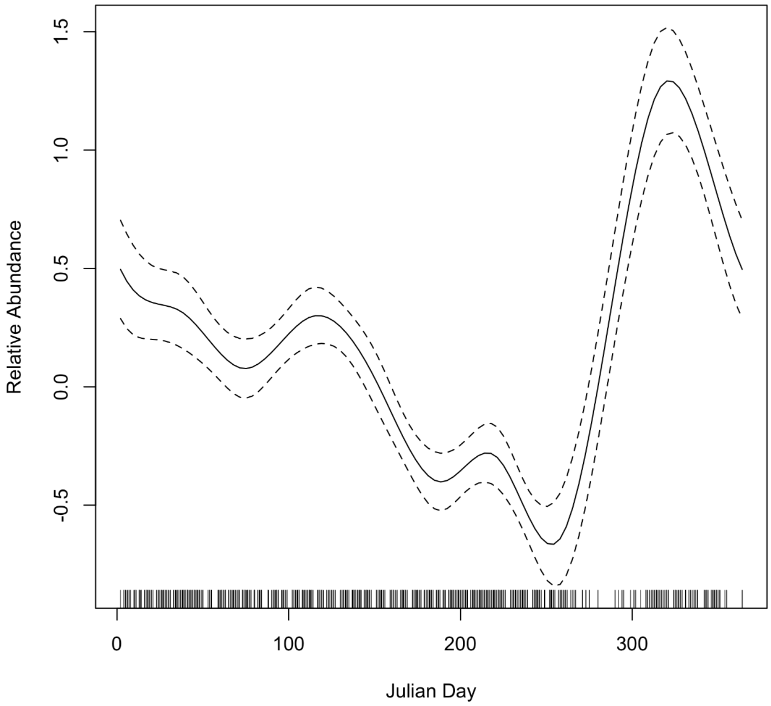


Figure 3.10: The estimated seasonal pattern of relative number of northern fulmar observed. The solid line is the smoothing curve for Julian day and dotted lines are 95% confidence bands.

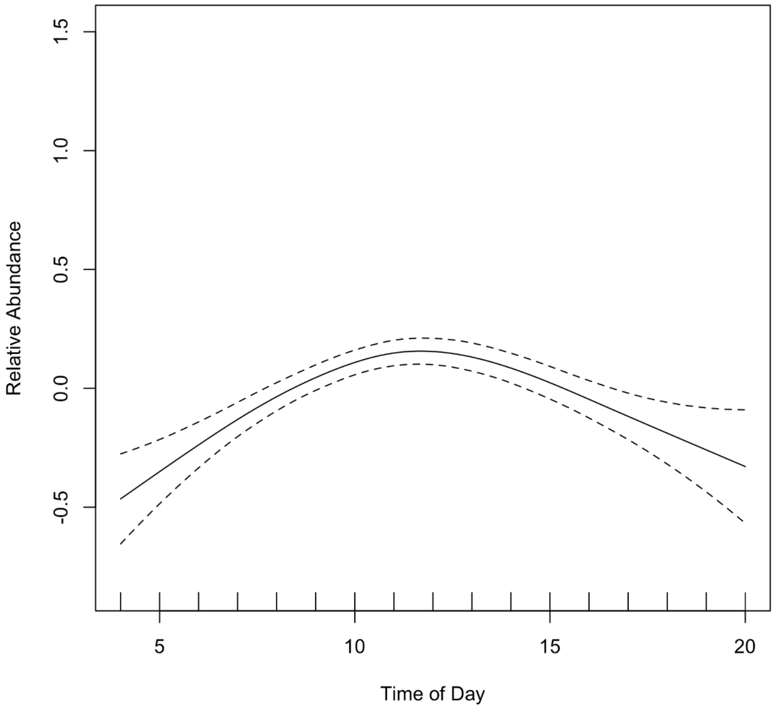


Figure 3.11: The estimated diurnal pattern of relative number of northern fulmar observed. The solid line is the smoothing curve for time of day and dotted lines are 95% confidence bands.

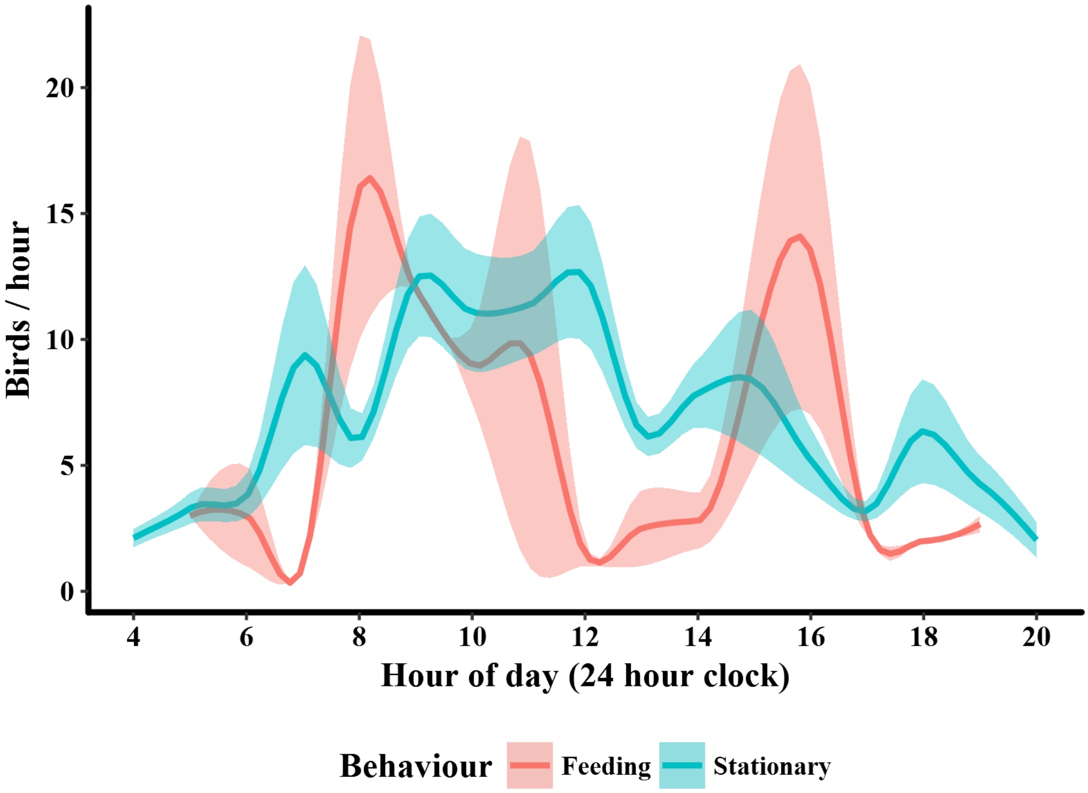


Figure 3.12: Mean number of feeding and resting northern fulmar observed per hour, throughout the day at Billia Croo, with standard error around the mean shown by the shaded areas.

3.3.4 Northern Gannet *Morus bassanus*

A total of 8,944 gannets were counted during the observation period, within 5km of the vantage point. The selected model, with auto-correlation as $\rho = 0.05$ shows a greater concentration of gannets off Breck Ness (S.Fig. 2.6) and that many of the observations of feeding birds were in this area (S.Fig. 2.7). The significant smooths of latitude and longitude ($p = <0.01$; Fig. 3.13) also predict higher number of birds in the southern part of the site.

Gannet numbers show a significant seasonal pattern ($p = <0.01$) increasing from the start of the year to a peak in sightings around May/June (Fig. 3.14, S.Table 2.5), there is a subsequent decrease after this period until a smaller peak in August. Gannets were also found to show a significant temporal pattern throughout the day ($p = <0.01$). Numbers encountered increased steadily throughout the day until 13:00-14:00, with an ensuing decline (Fig. 3.15). In 89% of observations where behaviour was recorded, gannets were documented as being stationary with only 10% of the observations noted as feeding (Fig. 3.17). In fact, with the exception of 13:00 and 19:00, mean encounter rate of resting gannets was always higher than those feeding. Glare extent was included within the model as it was found to contribute towards explaining variation in gannet numbers observed ($p = 0.04$; S.Tables 2.5 and 2.6; Fig. 3.16). The model predicted fewer birds observed under slight glare conditions and higher numbers under moderate and severe glare conditions, although the differences between these is not significant.

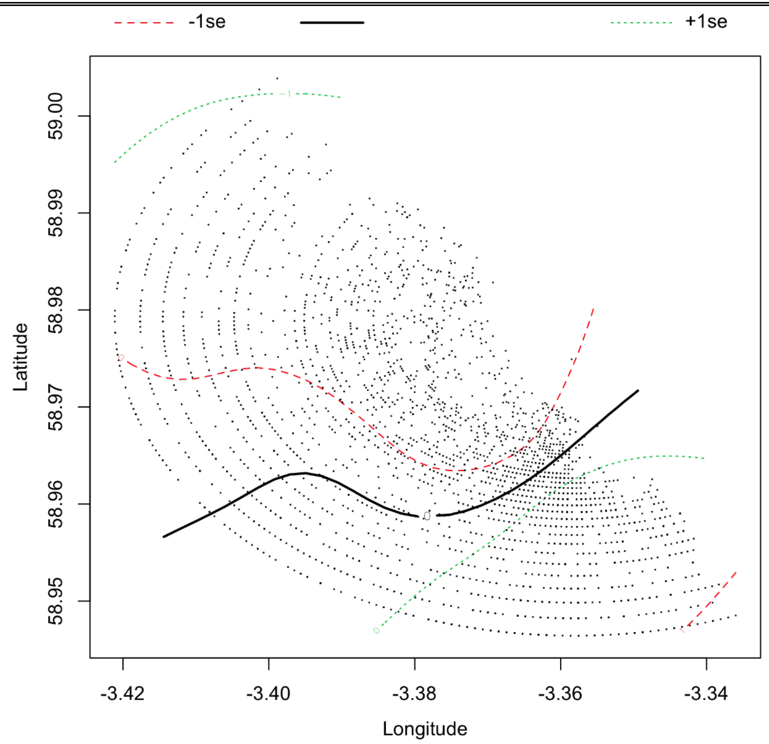


Figure 3.13: The estimated spatial pattern of relative number of northern gannets observed. The solid line is the smoothing curve for 0, red dotted lines are -1 standard error from the smoothing curve and the green dotted lines are +1 standard error from the smoothing curve.

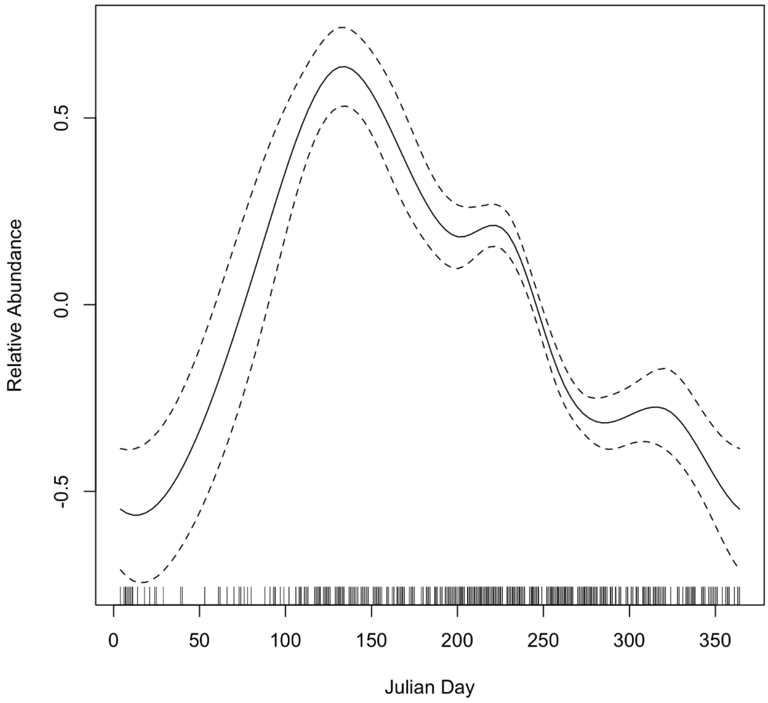


Figure 3.14: The estimated seasonal pattern of relative number of northern gannets observed. The solid line is the smoothing curve for Julian day and dotted lines are 95% confidence bands.

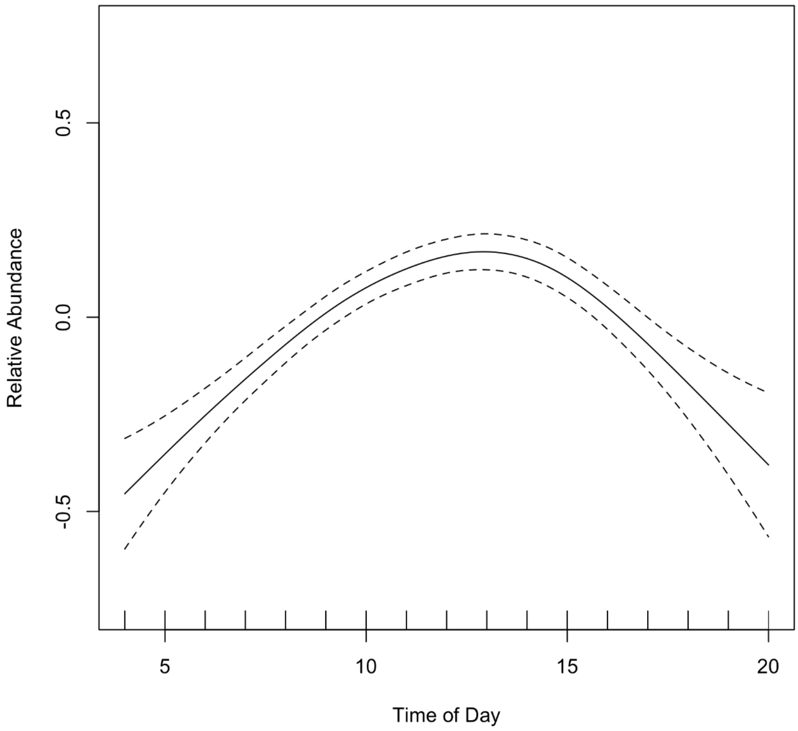


Figure 3.15: The estimated diurnal pattern of relative number of northern gannets observed. The solid line is the smoothing curve for time of day (hours) and dotted lines are 95% confidence bands.

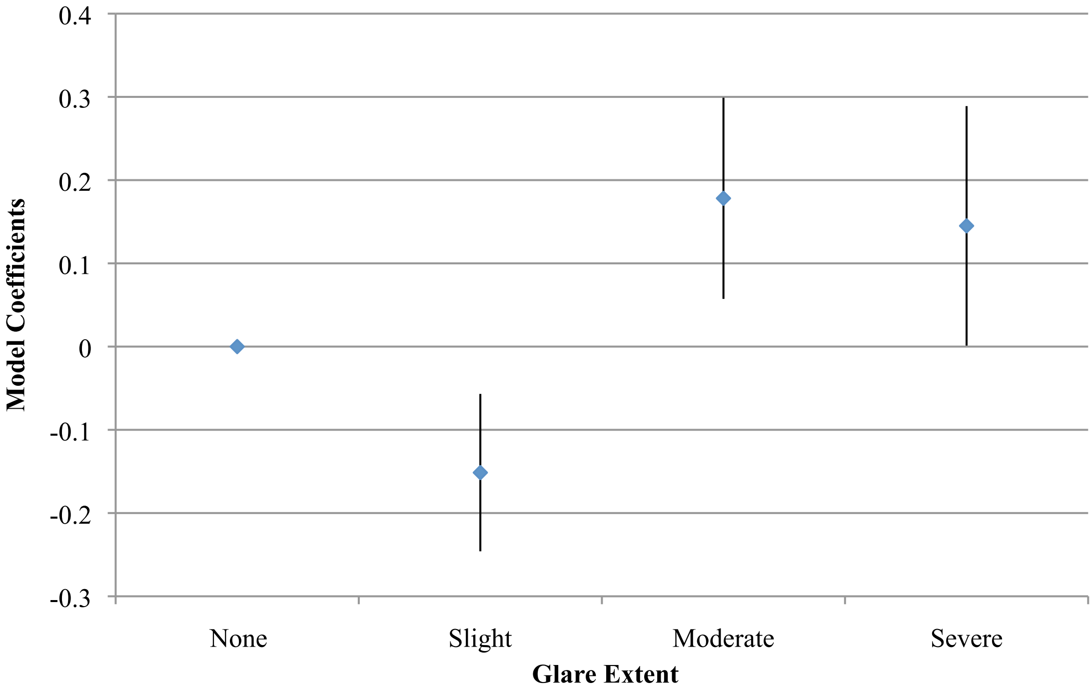


Figure 3.16: GAMM coefficient estimates (and standard errors) for northern gannets observed by glare extent at Billia Croo.

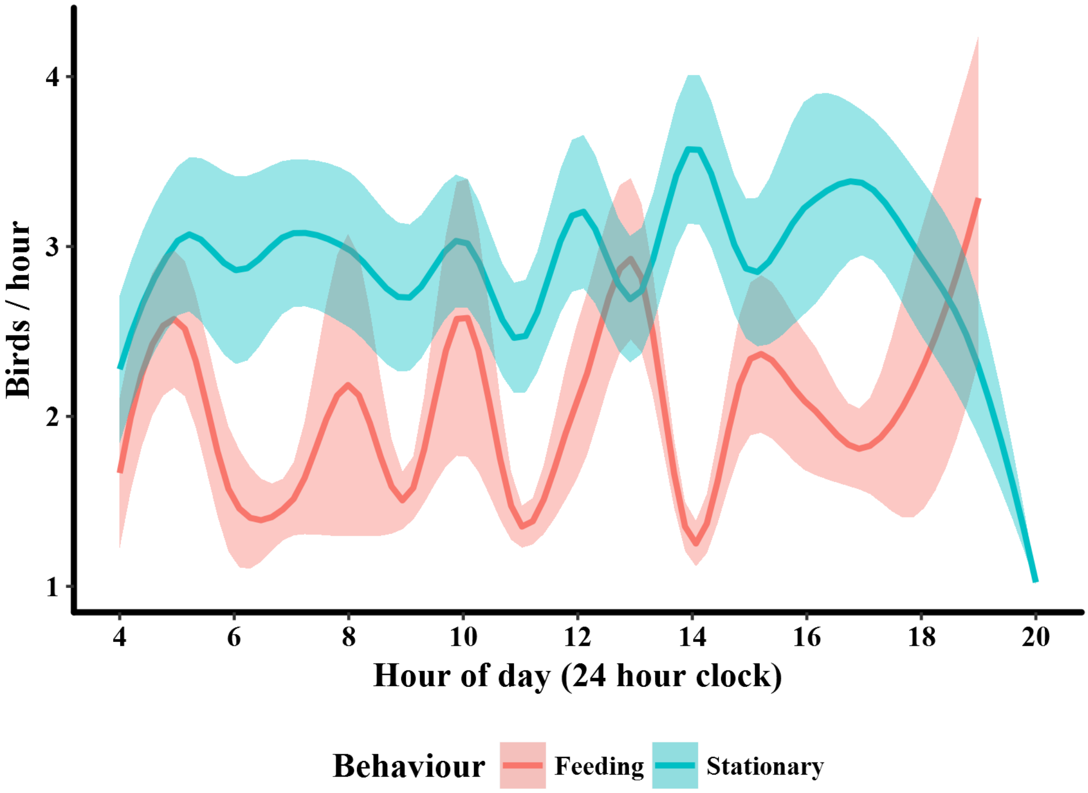


Figure 3.17: Mean number of feeding and resting northern gannets observed per hour, throughout the day at Billia Croo, with standard error around the mean shown by the shaded areas.

3.3.5 European Shag *Phalacrocorax aristotelis*

36,781 European shags were observed at Billia Croo during the period analysed. The selected GAMM, with auto-correlation; ρ = 0.12, contained the variables for latitude and longitude, Julian day, time from low tide, glare extent and observer ID (S.Tables 2.7 and 2.8).

As with gannets, the distribution maps show a clear spatial concentration of observations at the southern part of the site (S.Figs. 2.8 & 2.9). Most records were within 2,000m of the coastline between the Black Craig observation tower and Breck Ness. The highly significant smooth for latitude and longitude (*p* =<0.01) predicts that shag relative abundance increases south of the 0 smooth line and within a circled hotspot (58.97°N, 3.38°W; Fig. 3.18).

Shags, although a year-round resident (Forrester et al. 2007), were found to show seasonal variation with sightings steeply declining at the beginning of the year until early spring (i.e. March) when numbers increase (Fig. 3.19). A peak occurs around Julian day 150 (the beginning of June), followed by another decline until the end of July/beginning of

August (~ Julian day 200). After this, numbers steadily increase until the end of the year. The model predicted a gradual peak in relative abundance between 1 hour before low tide and 3 hours after (Fig. 3.20). However, figure 3.23 shows that the mean number of feeding shags peaked around 2 hours after high tide, with an additional mean peak period around low tide. The mean number of resting shags encountered shows a slight increase on the flooding tide. Figure 3.22 also highlights cyclic diurnal peaks in feeding activity (i.e. 08:00-09:00 and 12:00-13:00). Finally, while the mean number of resting shags encountered increases in the early morning, with a peak at 06:00, and decreases in the evening, the numbers encountered remained consistent through most of the day. Significantly fewer shags were detected in severe glare conditions (*p*=0.03) than other glare conditions (S.Tables 2.7 and 2.8; Fig. 3.21). Observer ID was also found to significantly affect the model with more shags observed by SW (*p*=<0.01).

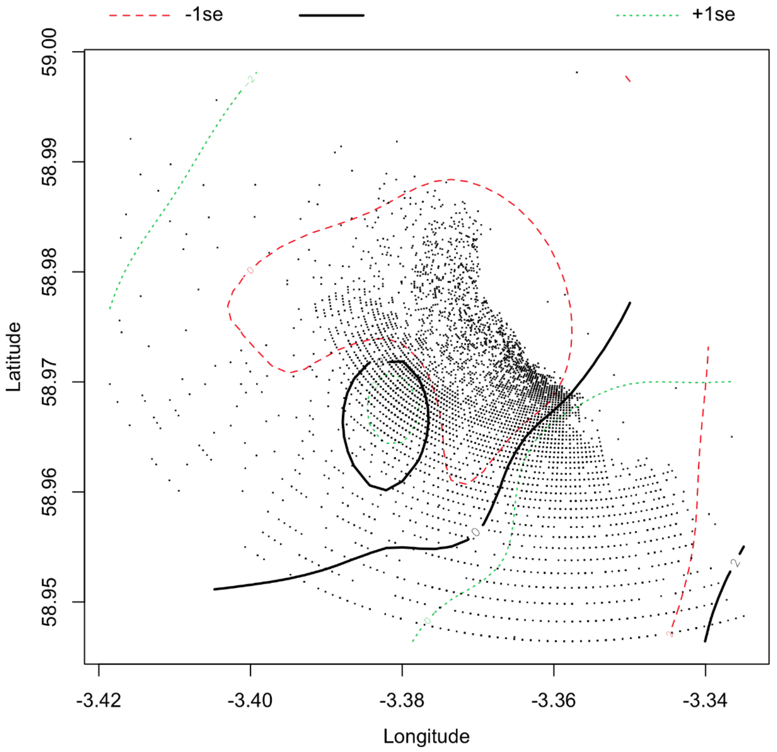


Figure 3.18: The estimated spatial pattern of relative number of European shag observed. The solid line is the smoothing curve for 0, red dotted lines are -1 standard error from the smoothing curve and the green dotted lines are +1 standard error from the smoothing curve.

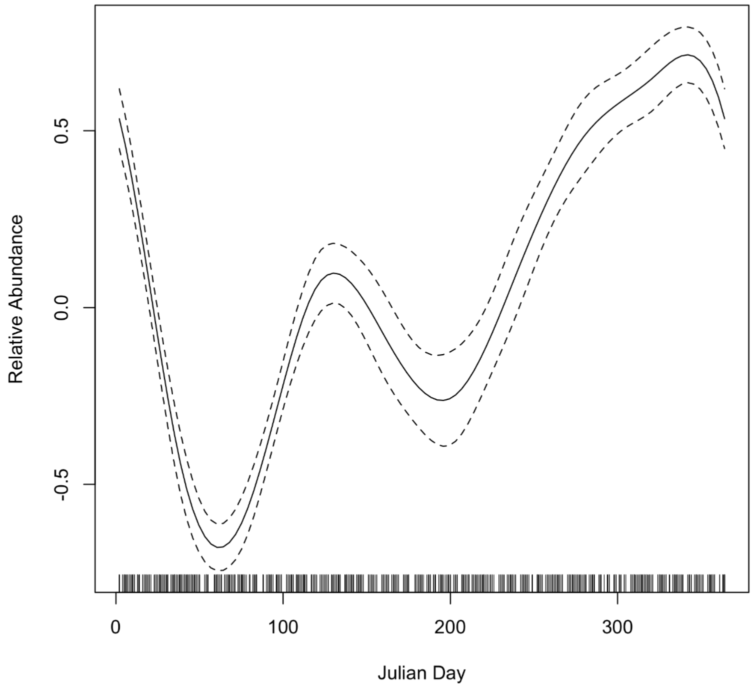


Figure 3.19: The estimated seasonal pattern of relative number of European shag observed. The solid line is the smoothing curve for Julian day and dotted lines are 95% confidence bands.

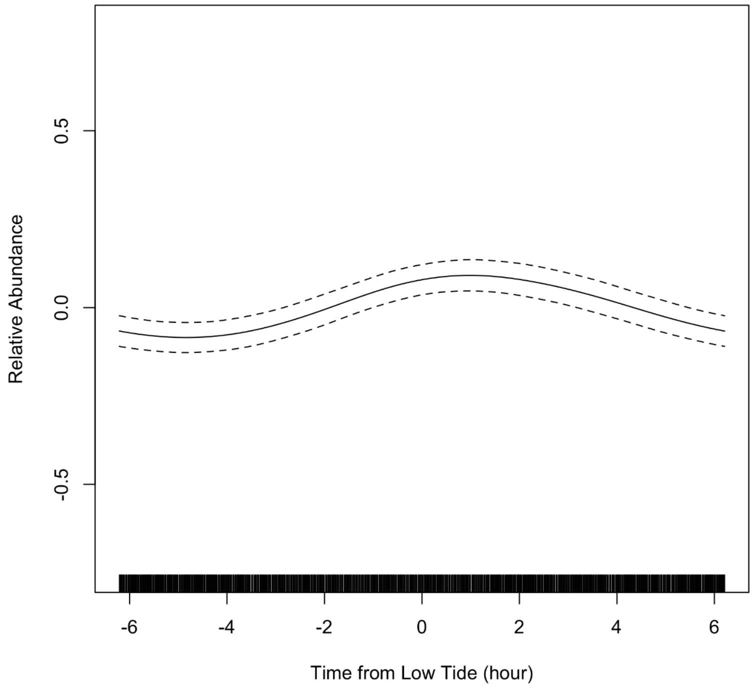


Figure 3.20: The estimated pattern of change in relative number of European shag observed across the semi-diurnal tidal cycle. The solid line is the smoothing curve for time from low tide (hours) and dotted lines are 95% confidence bands.

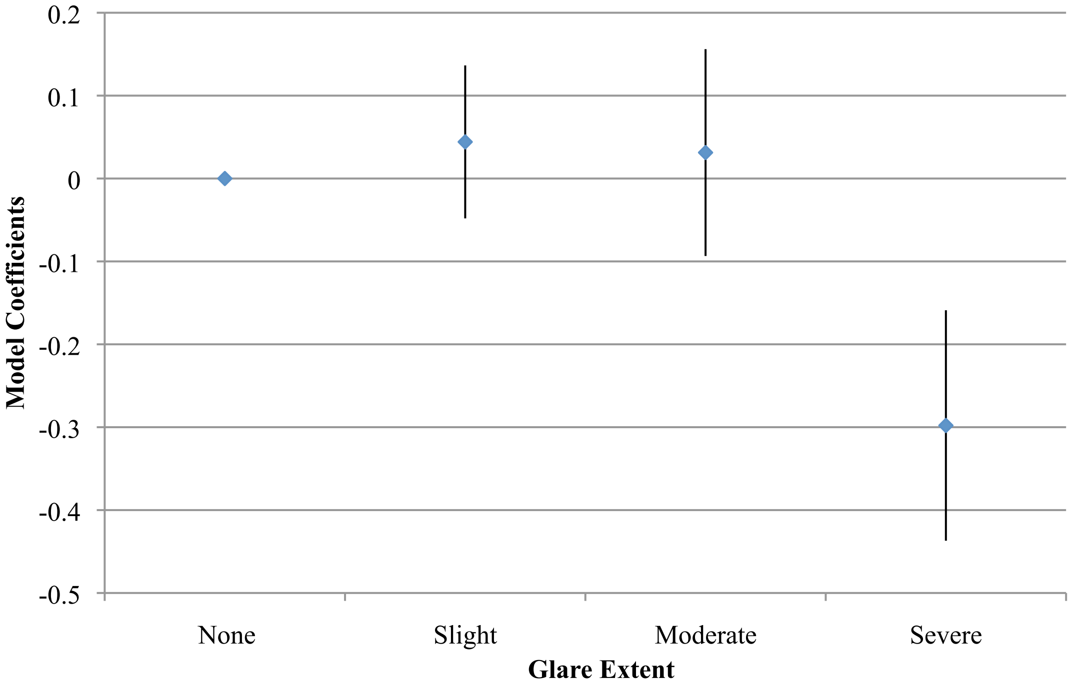


Figure 3.21: GAMM coefficient estimates (and standard errors) for European shags observed by glare extent at Billia Croo.

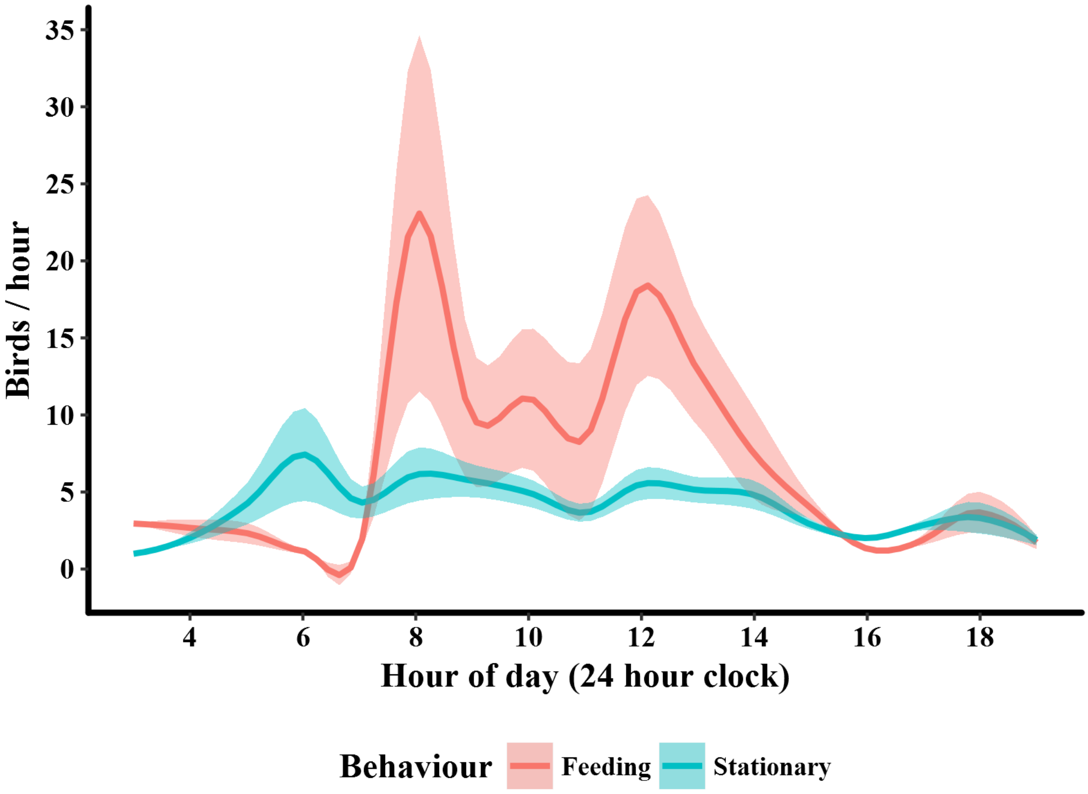


Figure 3.22: Mean number of feeding and resting European shags observed per hour, throughout the day at Billia Croo, with standard error shown by the shaded area.

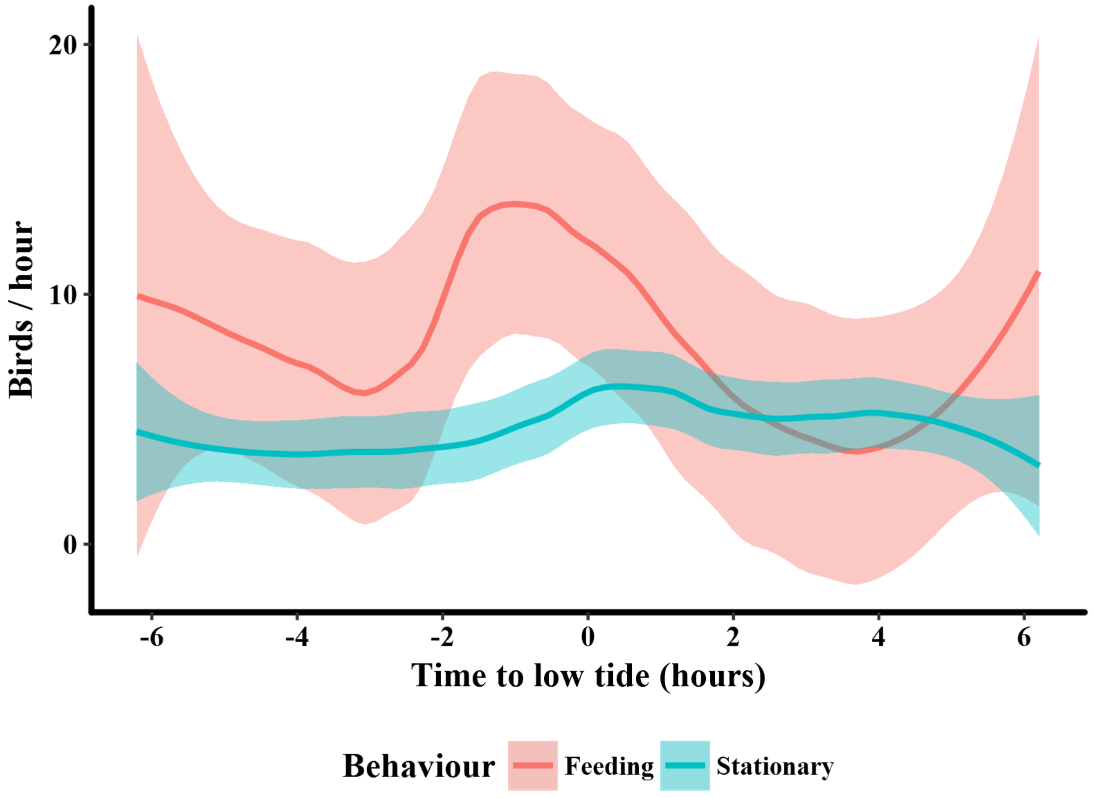


Figure 3.23: Mean number of feeding and resting European shags observed per hour, by time from low tide at Billia Croo, with standard error shown by the shaded area.

3.3.6 Great Skua *Stercorarius skua*

1,076 great skuas were observed at Billia Croo during the study period. The GAMM model, with a minimal amount of auto-correlation as ρ=<0.001, selected for latitude and longitude by mixed and single species flocks, glare extent and observer ID.

Spatial variation in the use of the Billia Croo site by great skuas differed when they were in a single species flock (*p*=0.09) vs. a mixed species flock (*p*=<0.01), and was only significant when they occurred in mixed species flocks. Figure 3.24 highlights hotspots of increased relative abundance, particularly around the central and southern parts of the site. This corresponds to the pattern seen in the distribution maps (S.Figs. 2.10 and 2.11) notably, where the feeding great skuas are observed, off Breck Ness and the southern area of the site.

Great skuas did show strong seasonal variation in abundance at Billia Croo (Fig. 3.26), however, due to an absence of any sightings during winter months, this was not selected for by the model. As the figure highlights, 68.59% of all sightings occurred during the summer, with 23.98% during the spring and the remaining proportion during the autumn. Glare extent did not significantly affect the numbers of great skuas observed, but a slight pattern can be seen (Fig. 3.25), which enabled a better fit of the preferred model. The number of great skuas observed decreased within increasing glare extent. Observer ID was also selected for (*p*=0.01).

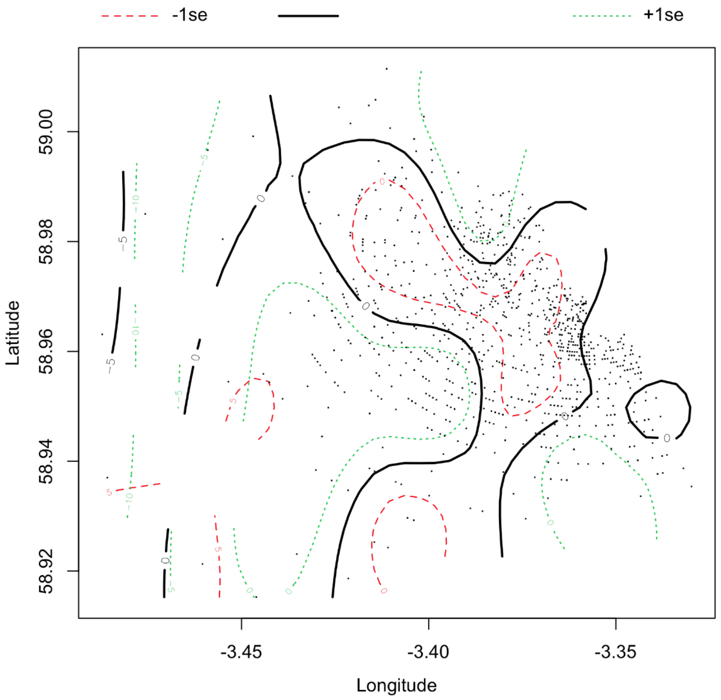


Figure 3.24: The estimated spatial pattern of relative number of great skua observed with other species. The solid line is the smoothing curve for 0, red dotted lines are -1 standard error from the smoothing curve and the green dotted lines are +1 standard error from the smoothing curve.

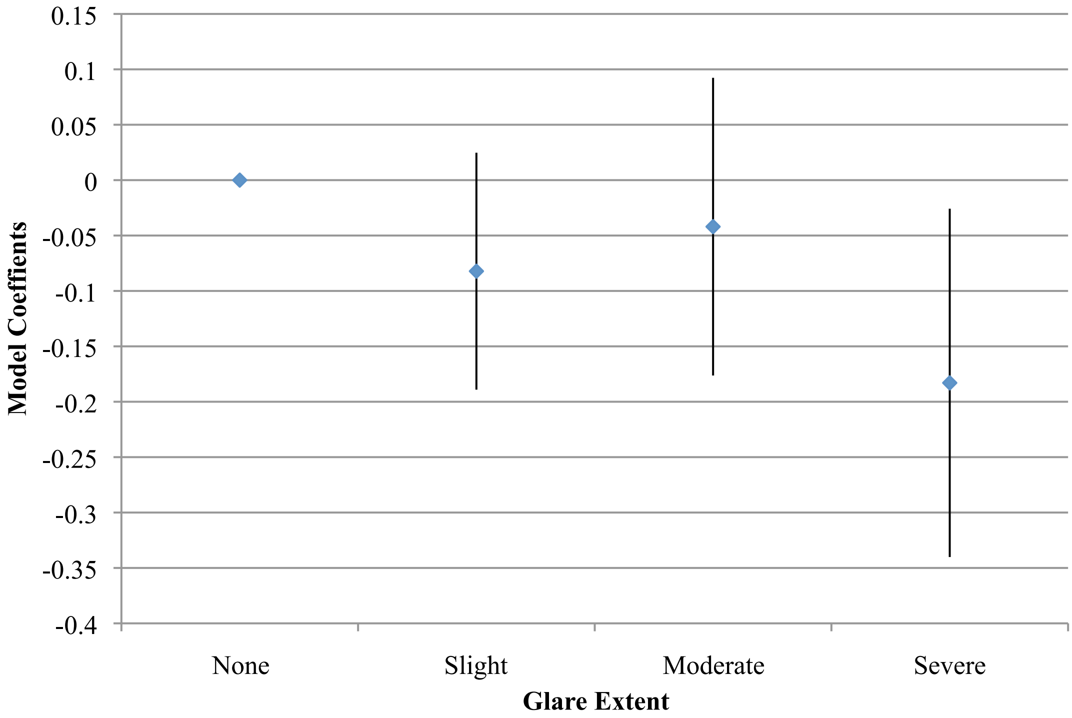


Figure 3.25: GAMM coefficient estimates (and standard errors) for great skua observed by glare extent at Billia Croo.

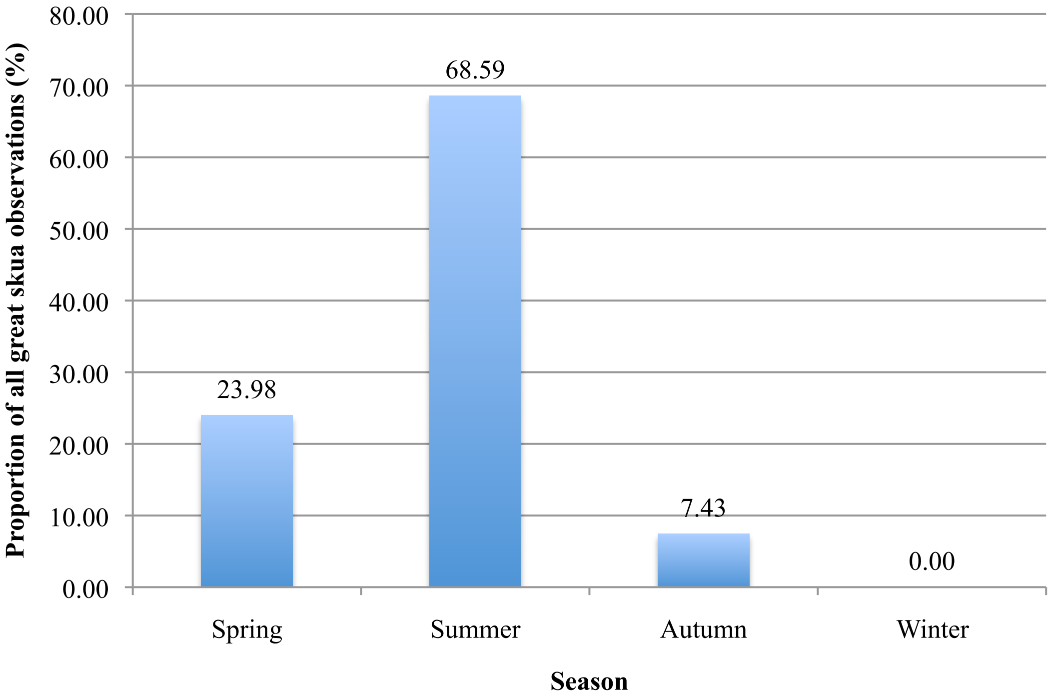


Figure 3.26: The proportion of great skua sightings by season, at Billia Croo.

3.3.7 Arctic Skua *S. parasiticus*

Only 76 Arctic skuas were observed at Billia Croo during the study period, which again was too few to model. These observations only occurred during Spring (59%) and Summer (41%). Supplementary figure 2.12, which plots these sightings, suggests a slightly different spatial usage to that of the great skua, with a greater proportion of the Arctic skua sightings to the west and northwest of the observation point.

3.3.8 Gulls *Larus spp.*

3,729 *Larus spp*. were observed at Billia Croo These included 833 common gulls (*L. canus*), 1,871 great black-backed gulls (*L. marinus*), 660 herring gulls (*L. argentatus*) and 365 that were unidentified to species (Fig. 3.31). The *Larus* gulls showed spatial and temporal variation in their use of the site, with the model (auto-correlation as $\rho = 0.06$) selecting for latitude and longitude ($p= 0.02$), Julian day (by species), glare extent and wind strength. The distribution maps (S.Figs. 2.13 & 2.14) show most of the sightings around Breck Ness and near the coastline along the southern part of the site.

The gull species observed showed different seasonal temporal patterns, all of which were significant (common gull, p=<0.01; great black-backed gull, p=<0.01; herring gull, p=<0.01, *Larus spp, p*=<0.01). Figure 3.28 highlights the different smooths for each species, all of which show peaks and troughs at different points. Relative abundance of common gulls increased towards the end of the year (after Julian day 300), while great black-blacked gulls and herring gulls both peaked in spring (after Julian day 100).

Gulls at Billia Croo, were predominantly observed resting at the site (Fig. 3.32 and S.Fig.2.14), with 47.06% of all observations (for which behaviours were recorded) being of resting great black-backed gulls. 4.66% of observations were of feeding *Larus* spp., unidentified to species. This probably reflects the inherent difficulty of identifying individuals within a gull feeding flock. Figure 3.33 highlights a diurnal pattern in the behaviour of gulls at Billia Croo, with a peak in the mean number of birds (17/hour) encountered feeding at 09:00. The mean number of resting birds showed an evening peak of 9 birds/hour at 17:00. Gull numbers were found to vary under certain environmental conditions. Glare extent was not found to be significant (p=0.27) but there was a clear pattern showing relative abundance decreasing with increasing severity of glare (Fig. 3.29) and the inclusion of the variable provided a better fit of the model. Wind strength significantly affect the numbers of gulls observed (p=<0.01). Lower mean numbers of birds were encountered in winds between Force 0 to 6, and numbers peaked during high winds of Force 7 (Fig. 3.30).

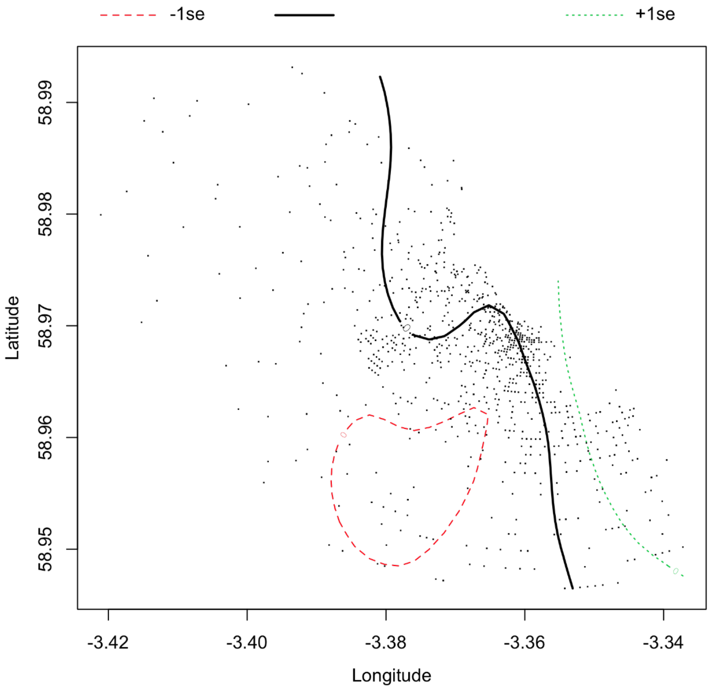


Figure 3.27: The estimated spatial pattern of relative number of *Larus spp.* observed. The solid line is the smoothing curve for 0, red dotted lines are -1 standard error from the smoothing curve and the green dotted lines are +1 standard error from the smoothing curve.

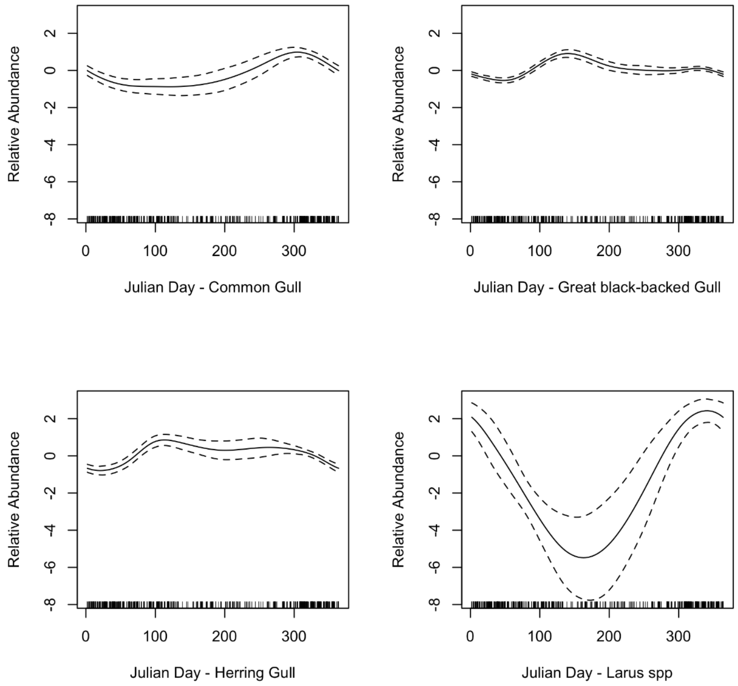


Figure 3.28: The estimated seasonal pattern of relative number of *Larus spp.* observed. The solid line is the smoothing curve for Julian day and dotted lines are 95% confidence bands.

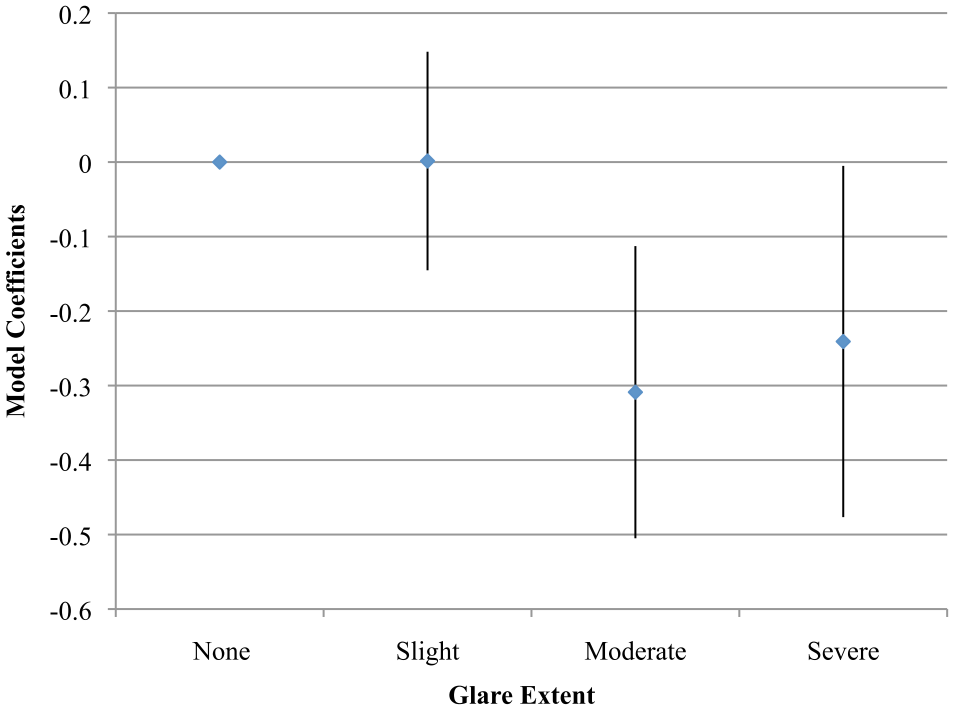


Figure 3.29: GAMM coefficient estimates (and standard errors) for *Larus spp*. observed by glare extent at Billia Croo.

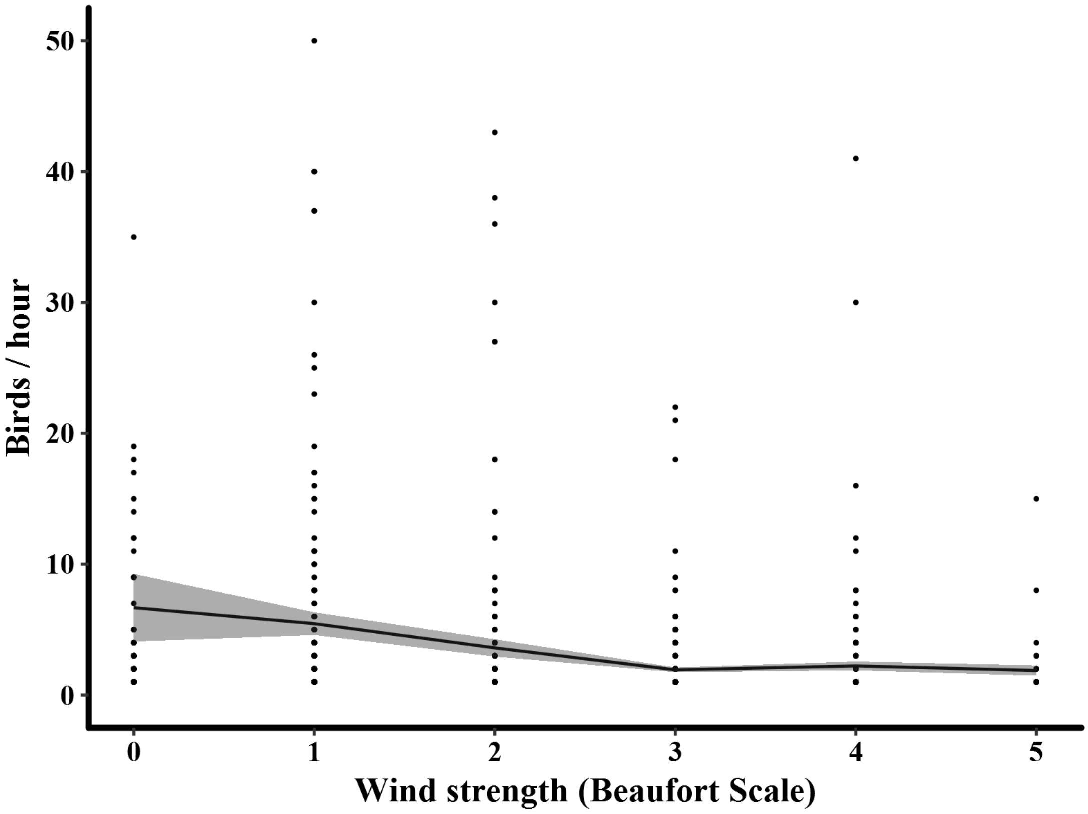


Figure 3.30: Mean number of *Larus spp.* encountered, by wind strength at Billia Croo, with raw data overlaid and standard error shown by the shaded area.

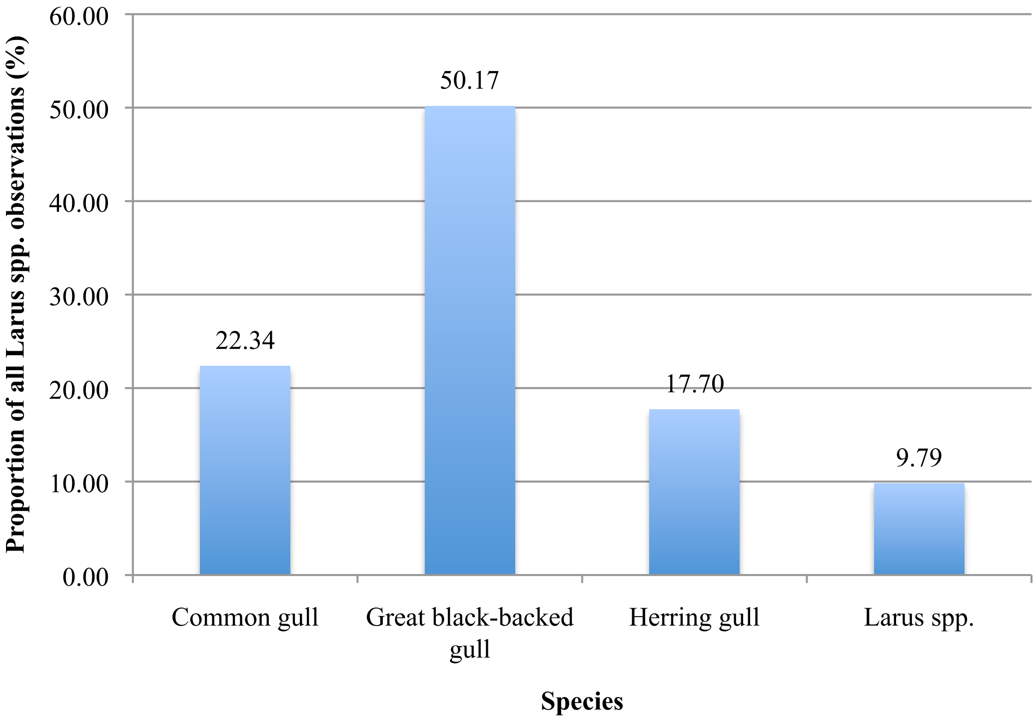


Figure 3.31: Proportion of *Larus spp.* sightings, by species at Billia Croo.

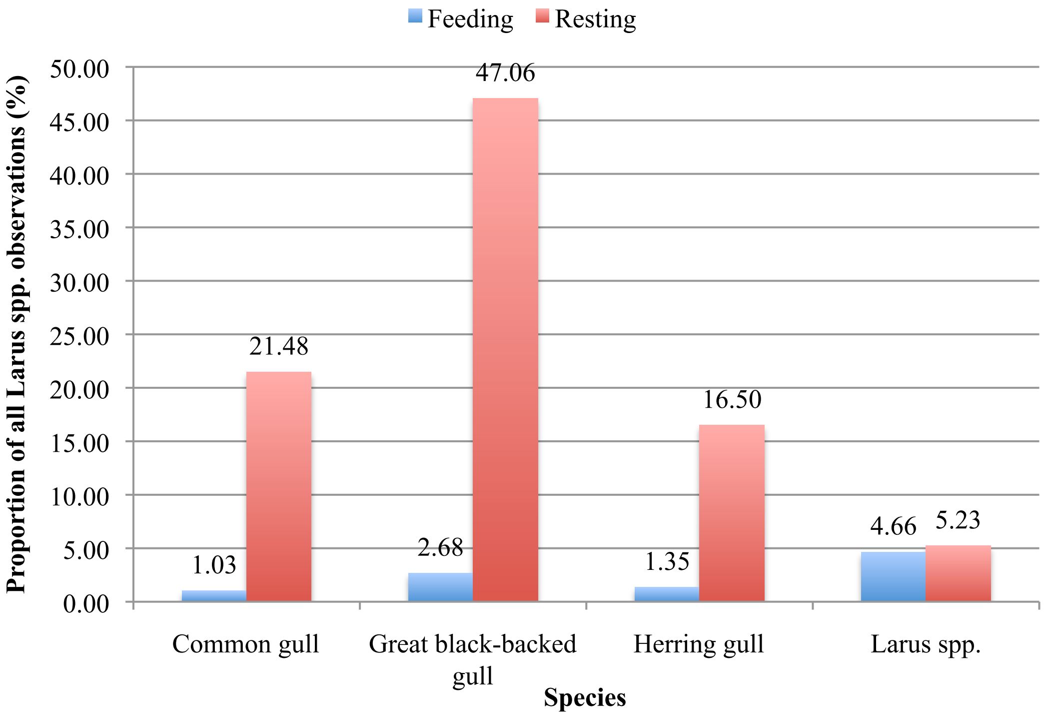


Figure 3.32: Proportions of all feeding and resting gull species, at Billia Croo.

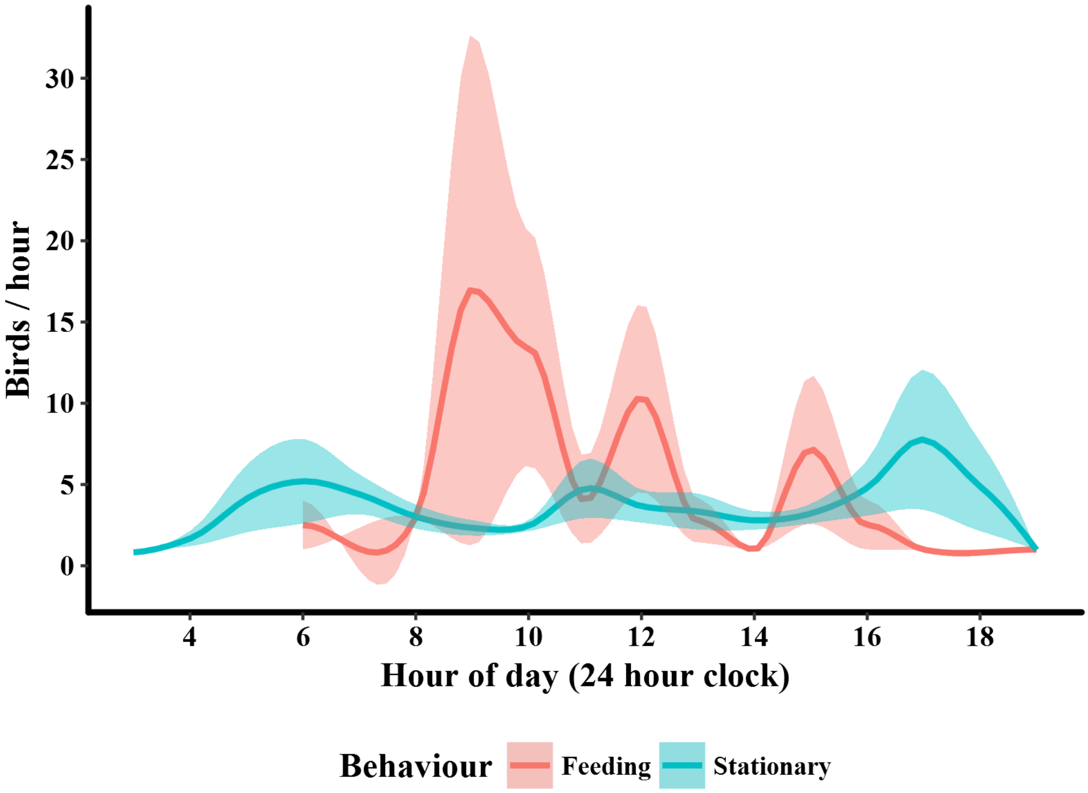


Figure 3.33: Mean number of feeding and resting *Larus spp.* observed per hour, throughout the day at Billia Croo, with standard error shown by the shaded area.

3.3.9 Black-Legged Kittiwake *Rissa tridactyla*

Over the study period 5,610 kittiwakes were observed at the Billia Croo test site. The selected model (autocorrelation ρ = 0.158) indicates spatial and temporal activity patterns, including variation over the season and with the diurnal tidal cycle, and influence of wind direction, glare extent and observer ID.

Kittiwakes were recorded across the Billia Croo site (Fig. 3.34 and S.Fig 2.15), however the model predicts greater numbers of kittiwakes observed further away from the observation point, suggesting a more pelagic use of the test site. The longitudinal and latitudinal smooth was highly significant (*p*=<0.01).

Kittiwakes showed significant seasonal variation, with the peak relative abundance during the spring months (when 60% of observations occurred; Fig. 3.35). Numbers subsequently decreased over the summer, when only 31% of the observations occurred. The remaining 9% of observations occurred during the autumn and winter months. Time from low tide was also important in the model, although with the higher threshold (of *p* <0.01) this is not significant (*p*=0.02) (S.Table 2.13 and Fig. 3.36). Sightings increased an hour before low tide, peaked an hour after low tide and decreased until 3 hours after high tide. While the mean number of birds observed resting and feeding both follow the same pattern of greater number around low tide over the same diurnal cycle, there are also peaks in foraging activity 4 hours before and after low tide (Fig. 3.37).

The model also selected the wind direction, glare and observer ID. With the higher *p* value threshold wind direction and glare were not significant but did allow for a better fitting model (wind direction, *p*=0.03; glare extent, *p*=0.03; and observer ID, *p*=<0.01; S.Table 2.14). The model predicted that fewer birds would be observed during moderate glare conditions, however it did predict a greater number of birds than those which were observed, would occur in severe glare conditions. Finally, the model estimated fewer birds in westerly winds, which, at Billia Croo, is the offshore wind direction.

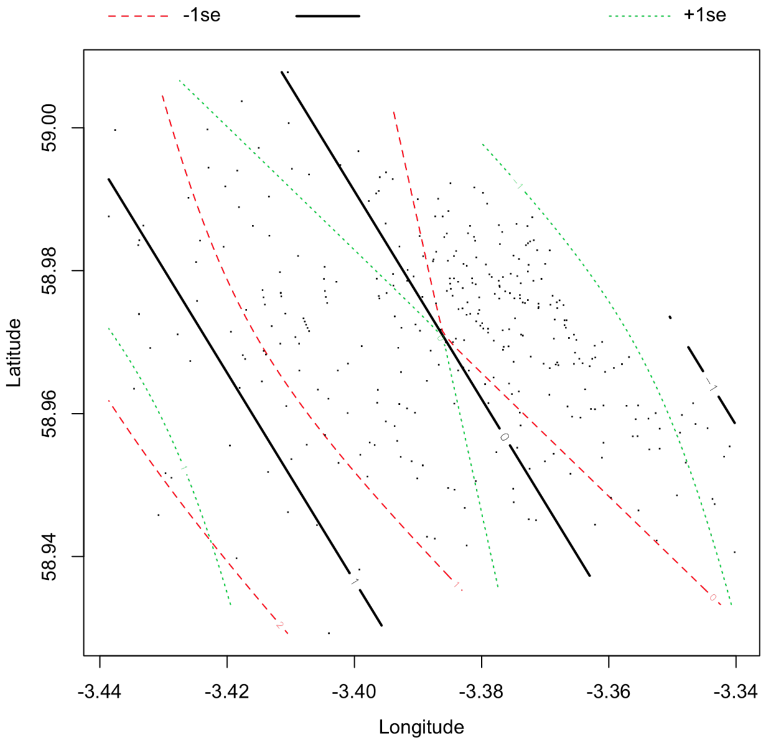


Figure 3.34: The estimated spatial pattern of relative number of black-legged kittiwakes observed. The solid line is the smoothing curve for 0, red dotted lines are -1 standard error from the smoothing curve and the green dotted lines are +1 standard error from the smoothing curve.

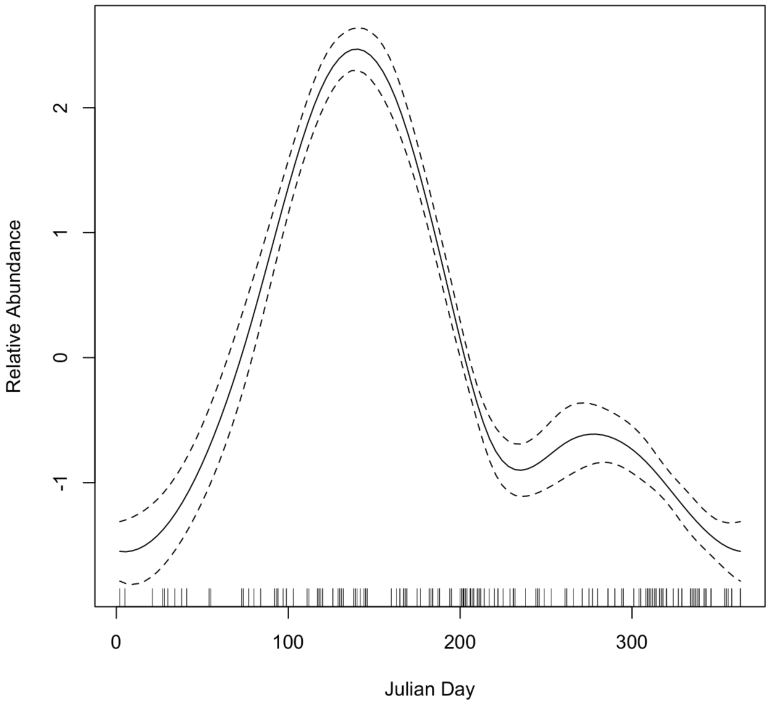


Figure 3.35: The estimated seasonal pattern of relative number of black-legged kittiwakes observed. The solid line is the smoothing curve for Julian day and dotted lines are 95% confidence bands.

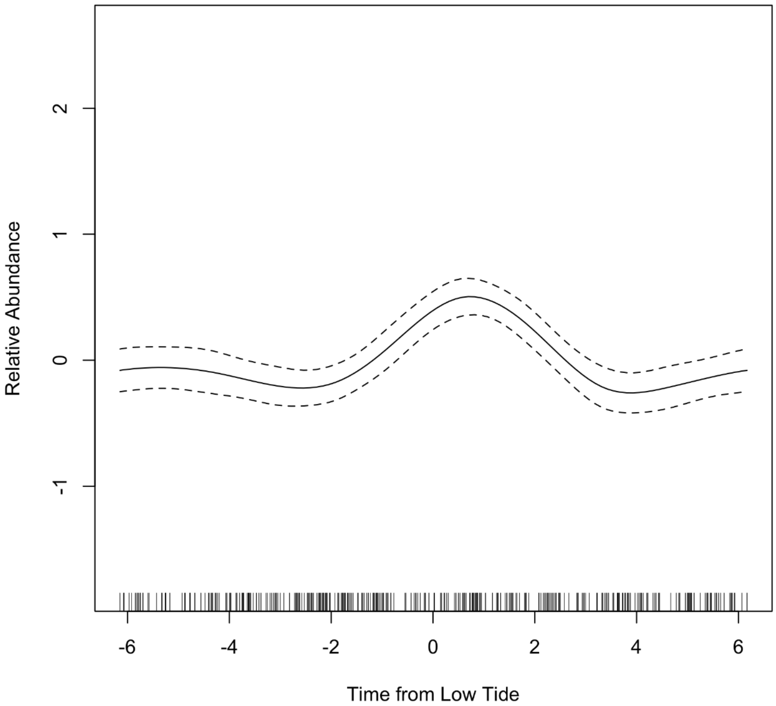


Figure 3.36: The estimated pattern of change in relative number of black-legged kittiwakes observed across the semi-diurnal tidal cycle. The solid line is the smoothing curve for time from low tide (hours) and dotted lines are 95% confidence bands.

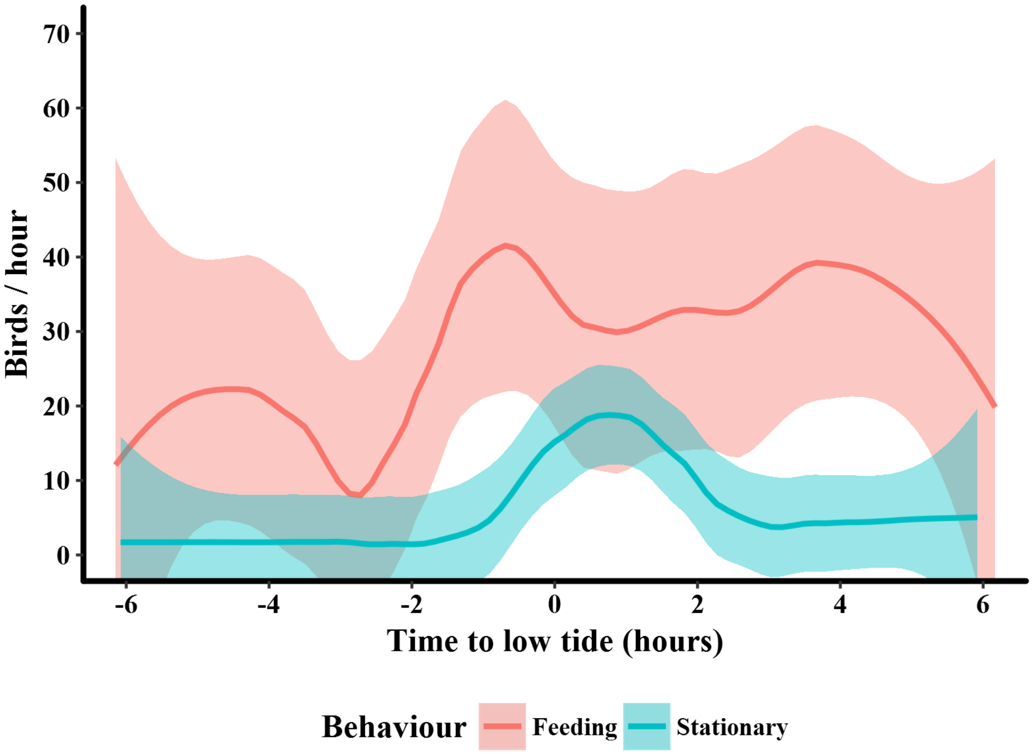


Figure 3.37: Mean number of feeding and resting black-legged kittiwakes observed per hour, by time from low tide at Billia Croo, with standard error shown by the shaded areas.

3.3.10 Arctic Tern *Sterna paradisaea*

A total of 2,315 Arctic terns were observed at Billia Croo scattered across the centre of the study site, within 2km of the coastline. However, no clear pattern has emerged (S.Fig. 2.16). 75% of the Billia Croo sightings occurred during the spring and the remaining 25% during the summer with a within season pattern (Fig. 3.38). The smooth clearly shows the decline from spring into summer, estimating a significantly greater number of terns in spring (*p*=<0.01). Terns also showed a highly significant diurnal pattern (*p*=<0.01) with a small peak in predicted numbers around 08:00, subsequent increase throughout the day until 15:00-16:00, after which numbers decreased (Fig. 3.39). Significantly greater numbers of Arctic terns were observed in southerly winds (*p*=<0.01), with fewer birds observed in offshore winds (westerly, Fig. 3.40). As with kittiwakes, fewer terns were predicted to be observed during slight glare extent compared with moderate or severe conditions (*p*=<0.01; Fig. 3.41).

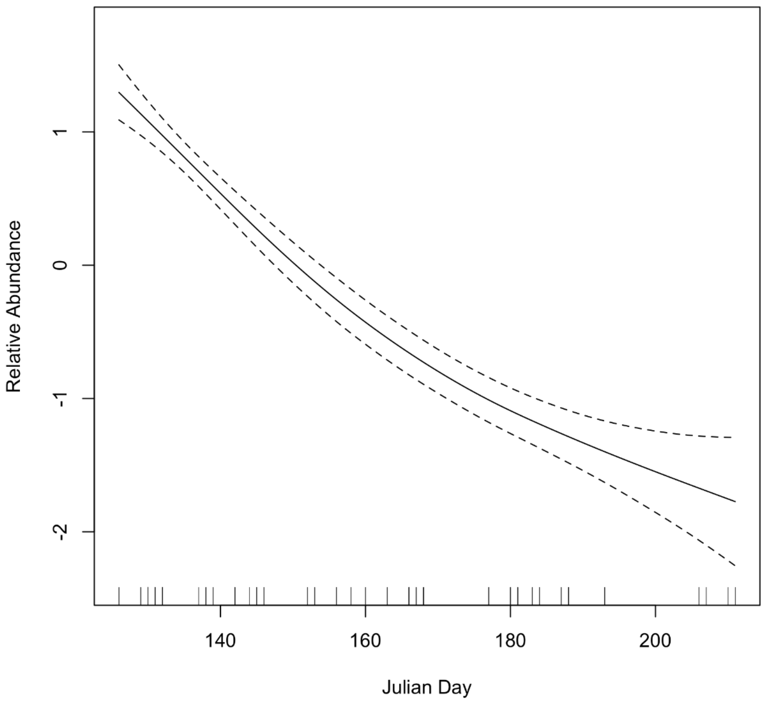


Figure 3.38: The estimated seasonal pattern of relative number of Arctic tern observed. The solid line is the smoothing curve for Julian day and dotted lines are 95% confidence bands.

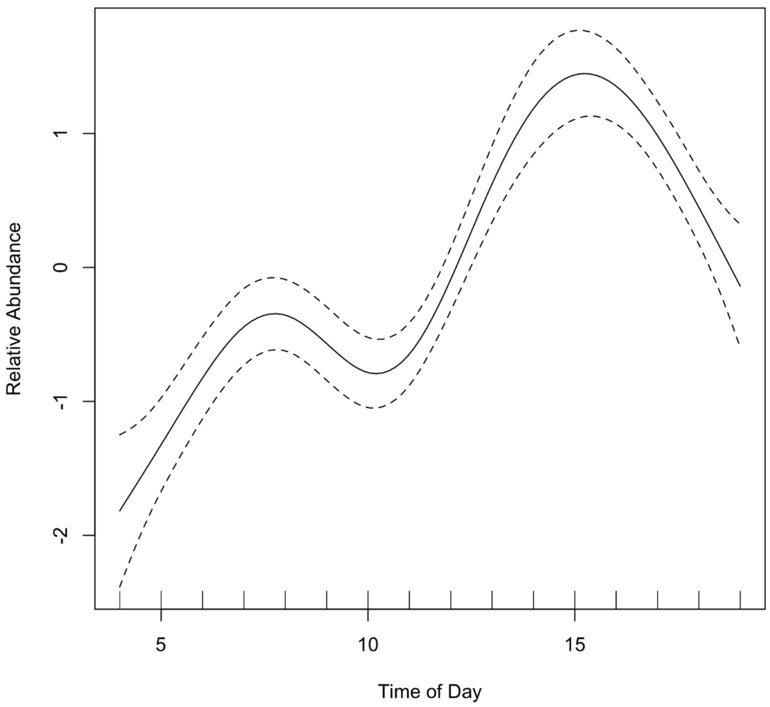


Figure 3.39: The estimated diurnal pattern of relative number of Arctic tern observed. The solid line is the smoothing curve for time of day (hours) and dotted lines are 95% confidence bands.

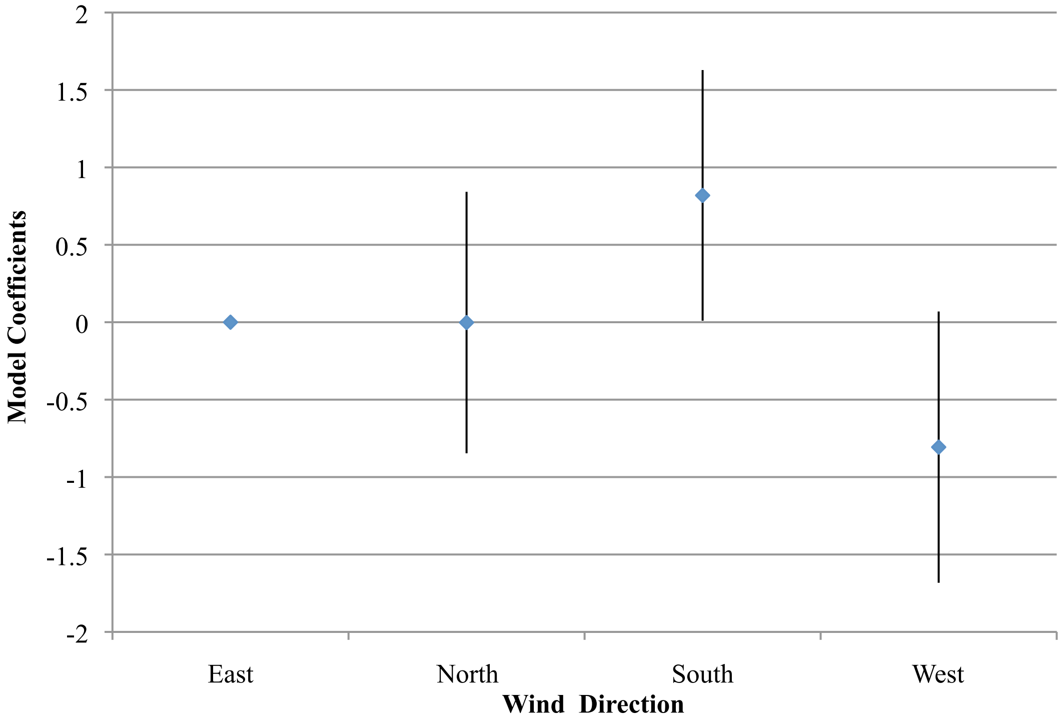


Figure 3.40: GAMM coefficient estimates (and standard errors) for Arctic tern observed by wind direction at Billia Croo.

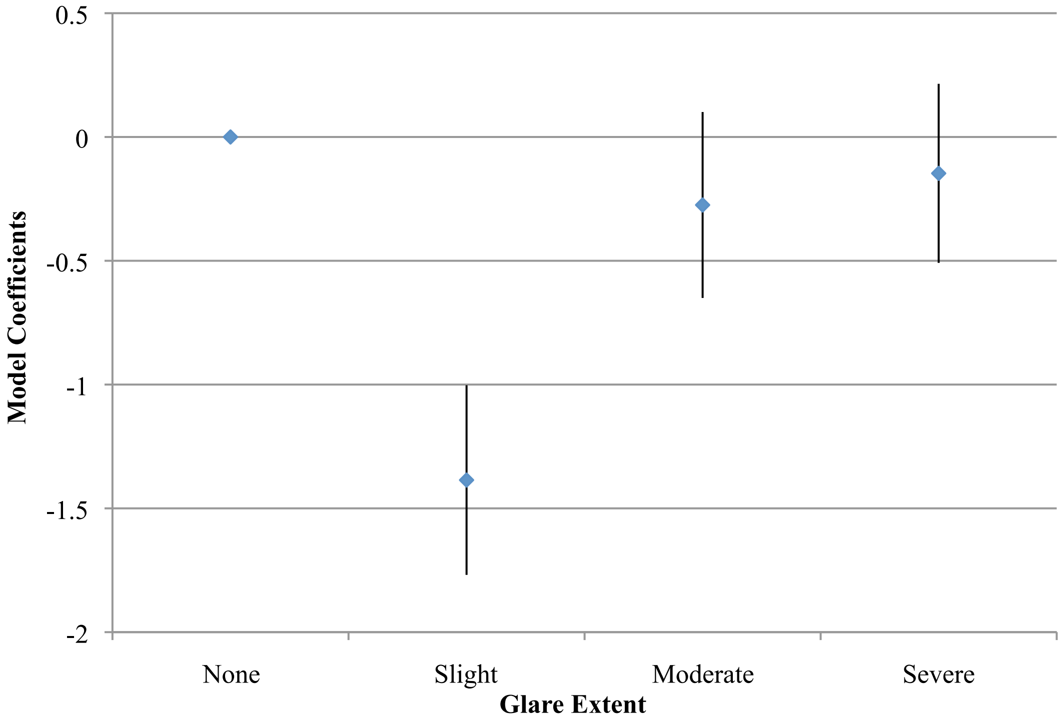


Figure 3.41: GAMM coefficient estimates (and standard errors) for Arctic tern observed by glare extent at Billia Croo.

3.3.11 Common Guillemot *Uria aalge*

A total of 10,433 common guillemots were counted during the observation period at Billia Croo. The preferred model for common guillemot counts did not include any spatial variables but was influenced by season (Julian day) and glare extent with low auto--correlation between observations of ρ = 0.046. While spatial patterns were not significant, supplementary figures 2.17 & 2.18 show a clear concentration of common guillemot sightings in the central part of the site, directly west and within 2km of the observation point.

Common guillemots showed a significant seasonal pattern (*p*=<0.01; Fig. 3.38), with numbers steeply increasing through the spring to a peak in June, after which numbers rapidly declined. Mean numbers of foraging common guillemots peaked between 07:00 and 09:00 (Fig. 3.44). Foraging was also influenced by tidal cycle with two clear peaks in the mean numbers encountered approximately one hour either side of low tide, although there is a drop in numbers observed at low tide (Fig. 3.45). There are also two smaller peaks in activity approximately one hour either side of high tide. By contrast, mean number of resting birds encountered was fairly consistent across the tidal cycle and throughout the day. Glare extent, while selected for in the model, was not highly significant (*p*=0.05). Nevertheless, figure 3.43 suggests a negative relationship between observed numbers of common guillemot and increasing levels of glare. Furthermore, when testing differences in levels of glare, there were significantly fewer common guillemots observed under severe glare conditions (S.Table 2.18).

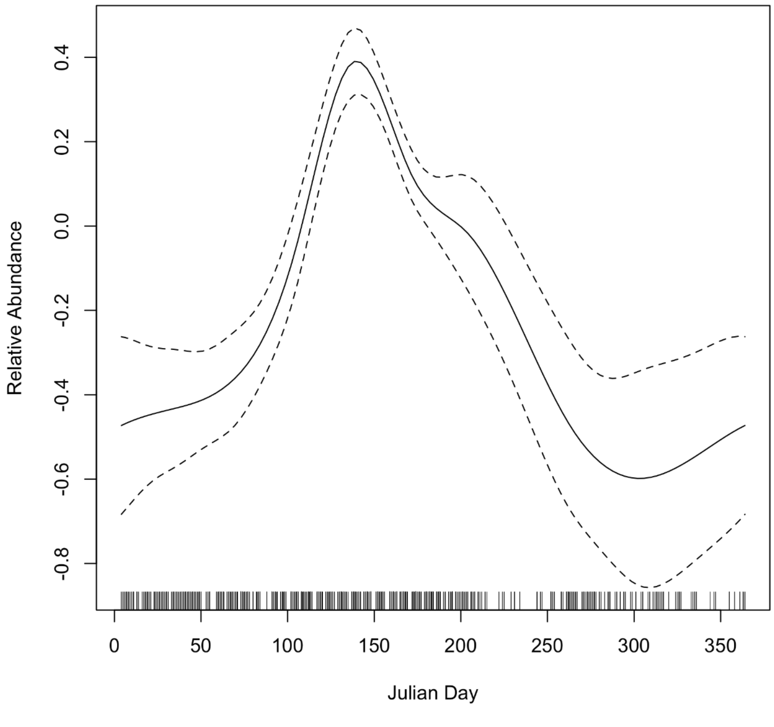


Figure 3.42: The estimated seasonal pattern of relative number of common guillemots observed. The solid line is the smoothing curve for Julian day and dotted lines are 95% confidence bands.

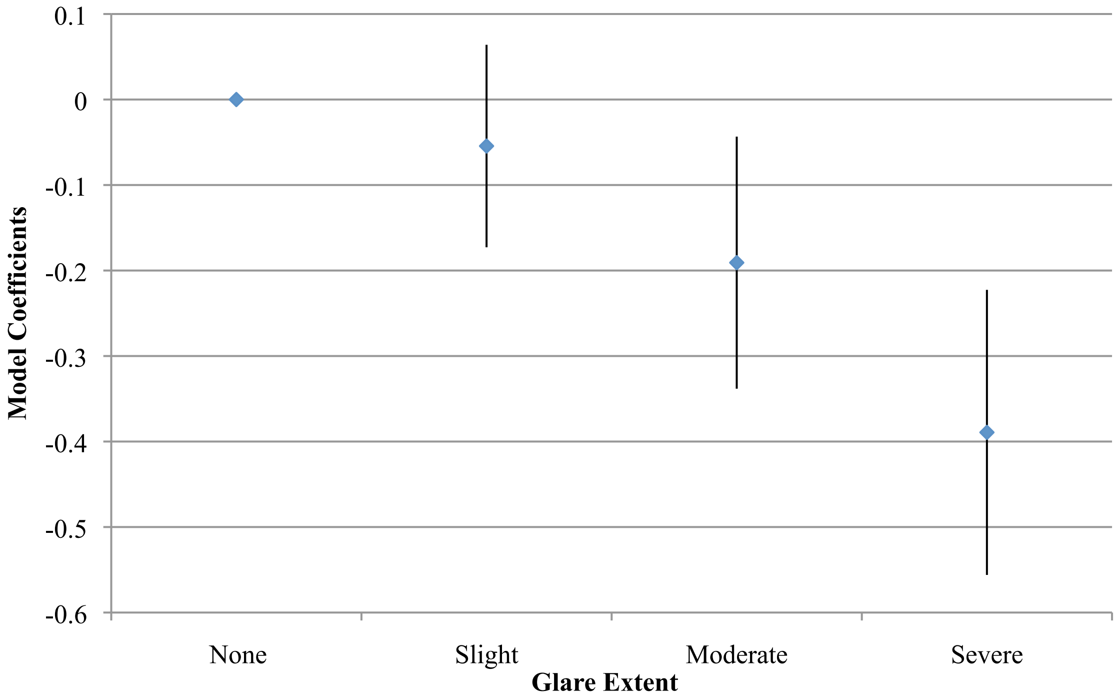


Figure 3.43: GAMM coefficient estimates (and standard errors) for common guillemots observed by glare extent at Billia Croo.

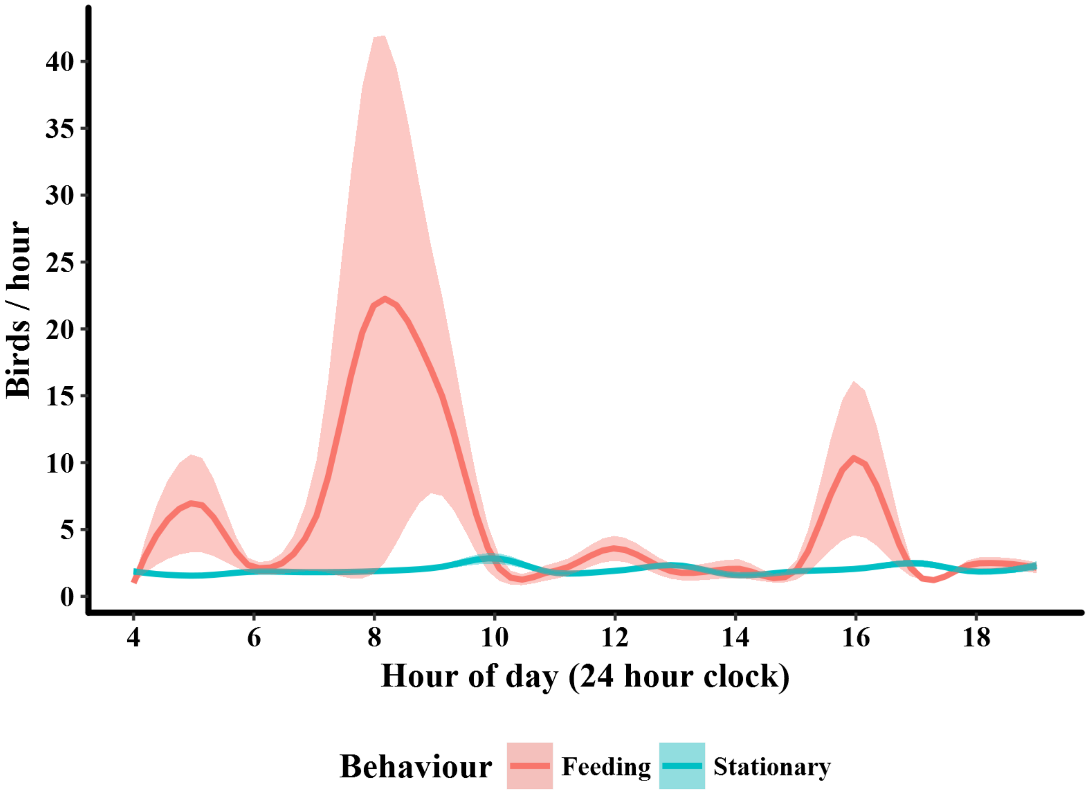


Figure 3.44: Mean number of feeding and resting common guillemots observed per hour, throughout the day at Billia Croo, with standard error shown by the shaded areas.

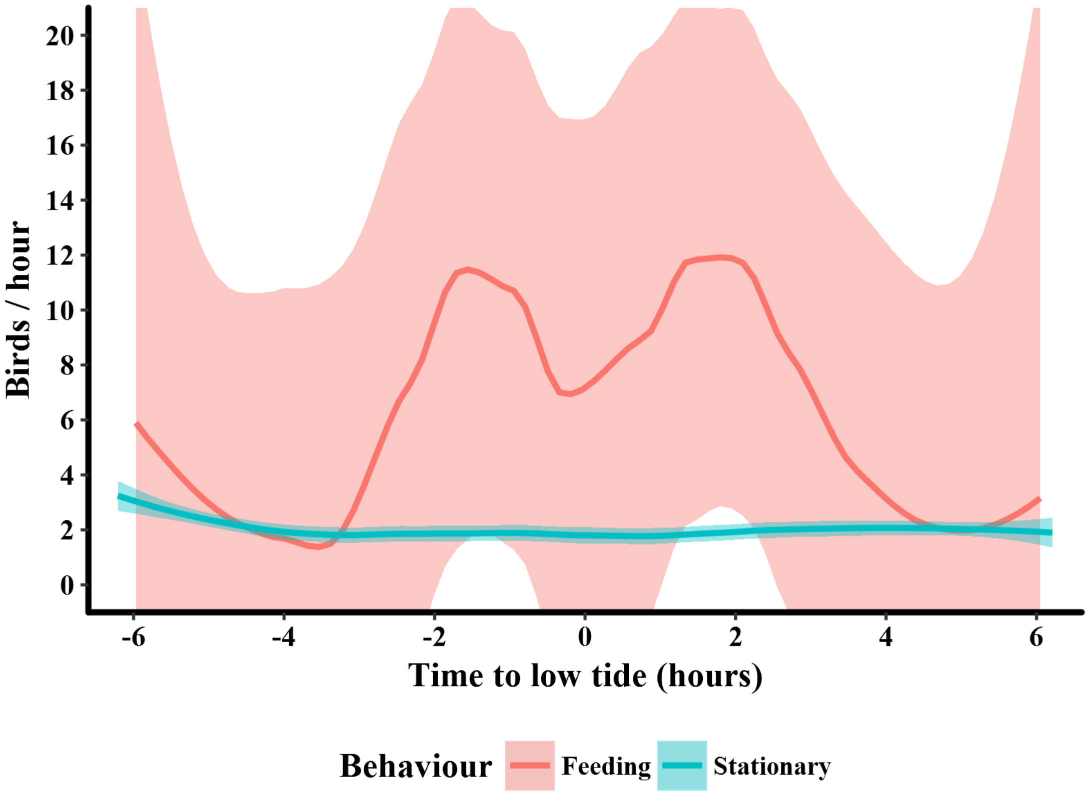


Figure 3.45: Mean number of feeding and resting common guillemots observed per hour, by time from low tide at Billia Croo, with standard error shown by the shaded area.

3.3.12 Razorbill *Alca torda*

A total of 380 razorbills were counted during the observation period. The preferred model (auto-correlation ρ = 0.20) indicates that razorbill numbers are a function of latitude and longitude (*p*=<0.01), and Julian day (*p*=0.01) (S.Tables 2.19 & 2.20).

Razorbills show a very similar distribution to that of the common guillemot at the test site (with fewer data points; S.Figs. 2.19 & 2.20), with higher numbers predicted to be encountered nearer the shore (Fig. 3.46).

Razorbill numbers increase steeply through the spring to a peak in late May/early June, and then rapidly decline (Fig. 3.47). Diurnal variation was observed in razorbill behaviours with a mean peak of 5 feeding birds at 18:00 (Fig. 3.48). While the number of resting razorbills is more consistent throughout the day, a peak of 7.2 birds can be seen at 15:00 (Fig. 3.48). Behaviour was also influenced by the tidal cycle (Fig. 3.49). Highest numbers of feeding birds were encountered on an ebbing tide, while the numbers of resting birds encountered are highest on a flooding tide. There are further peaks in both feeding and resting behaviours (3 birds and 4.6 birds, respectively) one hour before low tide and a final mean peak of 5 feeding birds 4 hours prior to high tide.

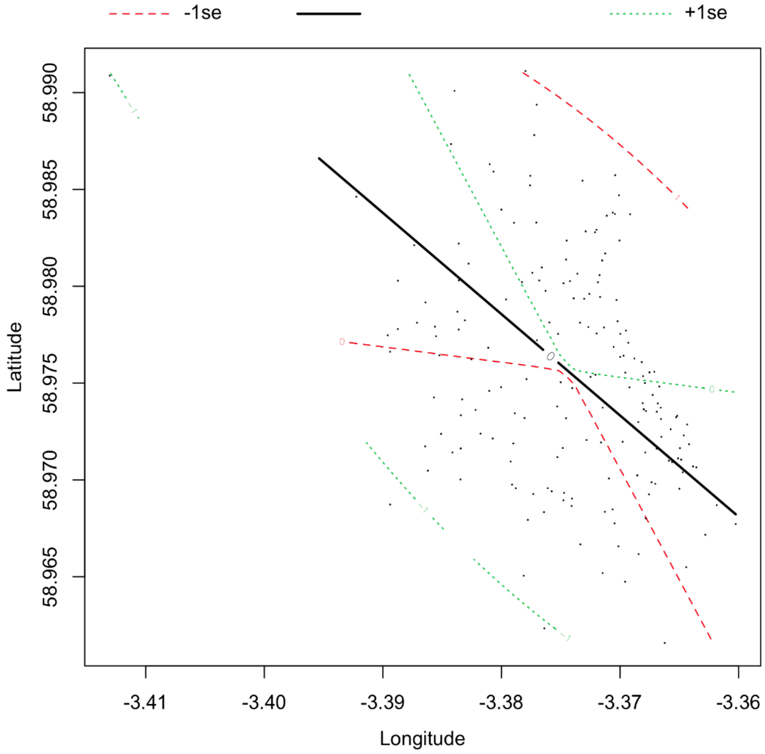


Figure 3.46: The estimated spatial pattern of relative number of razorbills observed. The solid line is the smoothing curve for 0, red dotted lines are -1 standard error from the smoothing curve and the green dotted lines are +1 standard error from the smoothing curve.

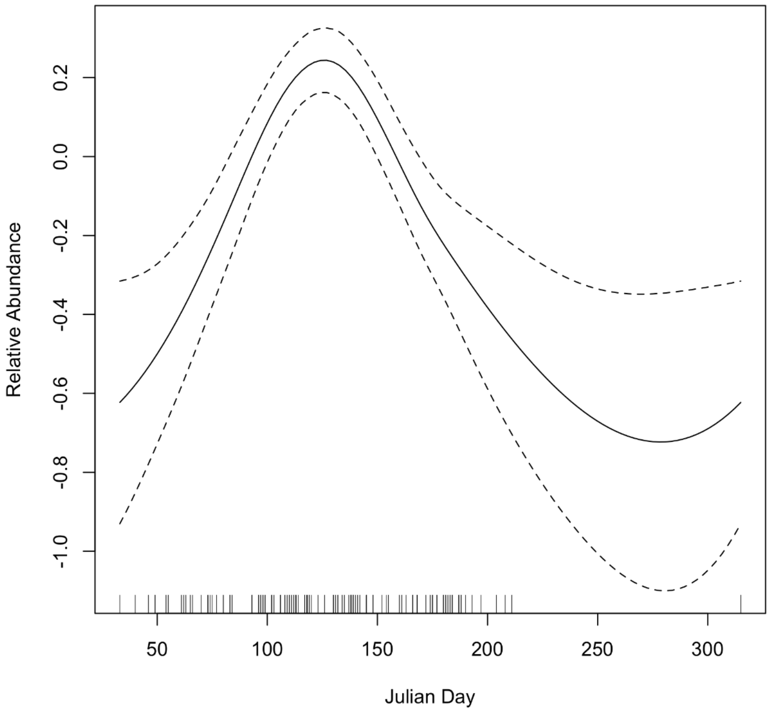


Figure 3.47: The estimated seasonal pattern of relative number of razorbills observed. The solid line is the smoothing curve for Julian day and dotted lines are 95% confidence bands.

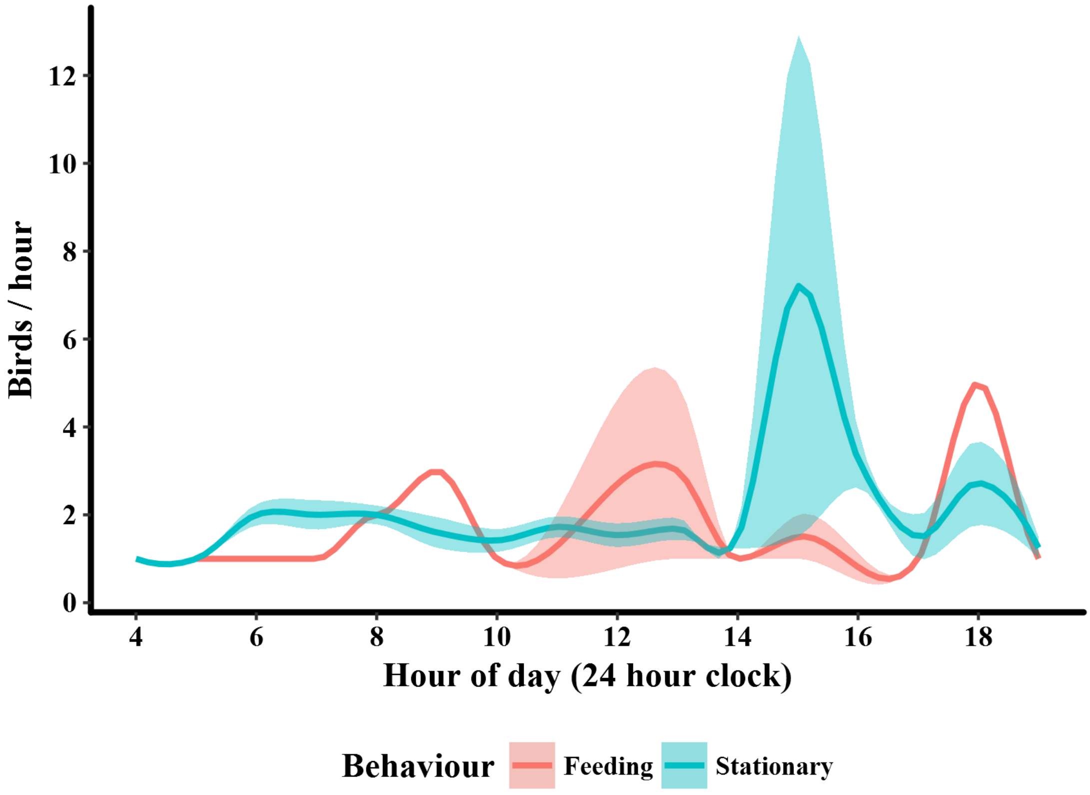


Figure 3.48: Mean number of feeding and resting razorbills observed per hour, throughout the day at Billia Croo, with standard error shown by the shaded areas.

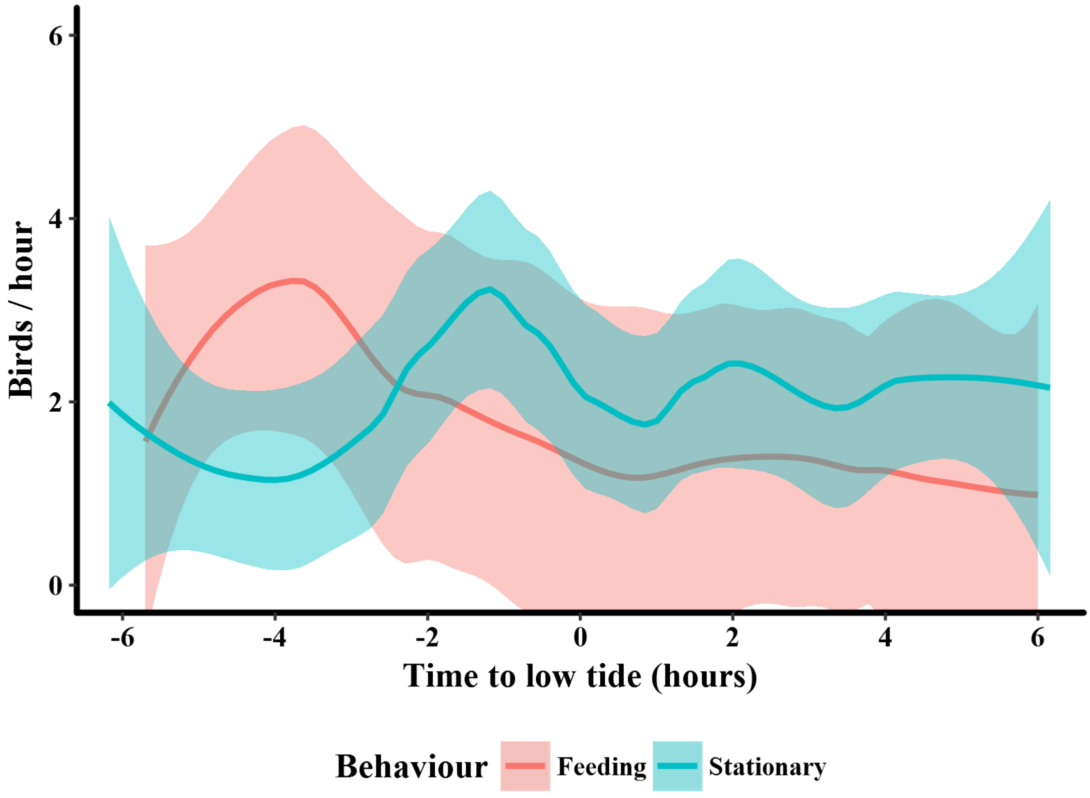


Figure 3.49: Mean number of feeding and resting razorbills observed per hour, by time from low tide at Billia Croo.

3.3.13 Black Guillemot *Cepphus grylle*

A total of 2,786 black guillemots were observed at Billia Croo site. The best fitting model (auto-correlation $\rho = 0.07$) indicates that the observed black guillemot numbers are a function of latitude and longitude (p=<0.01), time of year (Julian date) (p=0.01), time of day (p=<0.01) and wind strength (S.Tables 2.21 & 2.22). The correlation between observations was estimated as $\rho = 0.07$. Most black guillemots were recorded within 2,000m of the coastline between the Black Craig observation tower and Breck Ness (S.Figs. 2.21 & 2.22). The smooth within figure 3.50 also highlights this hotspot. There are some scattered sightings to the south of the site, while <1% of the feeding and resting birds were observed in the northern part of the site. Temporal variation in site usage was also highly significant, despite black guillemots being one of the few resident seabird species. They were more frequently recorded during the spring and summer months, with a peak in May (Fig. 3.51) and their significantly crepuscular trend is seen through the peak in encounters at approximately 04:00-05:00 (Fig. 3.52). Numbers decreased until 15:00, and then once again increased. This crepuscular trend is also reflected in the mean numbers of feeding and resting birds encountered per hour (Fig. 3.54). Resting bird encounters decreased from the 04:00 peak (2.69 birds/hour) until 17:00 after which it increases, peaking at 19:00 (2.56 birds/hour). The encounter rate for feeding birds indicates four peaks of feeding activity: early morning at 05:00 (2.5 birds/hour), mid-morning at 09:00 (1.8 birds/hour), approximately 16:00 and then evening at 19:00 (2.5 birds/hour). Wind strength, also not significant in the model, did provide a better fit, and figure 3.53 shows a decrease in the mean number of black guillemots encountered as wind strength increased (S.Tables 2.21 and 2.22).

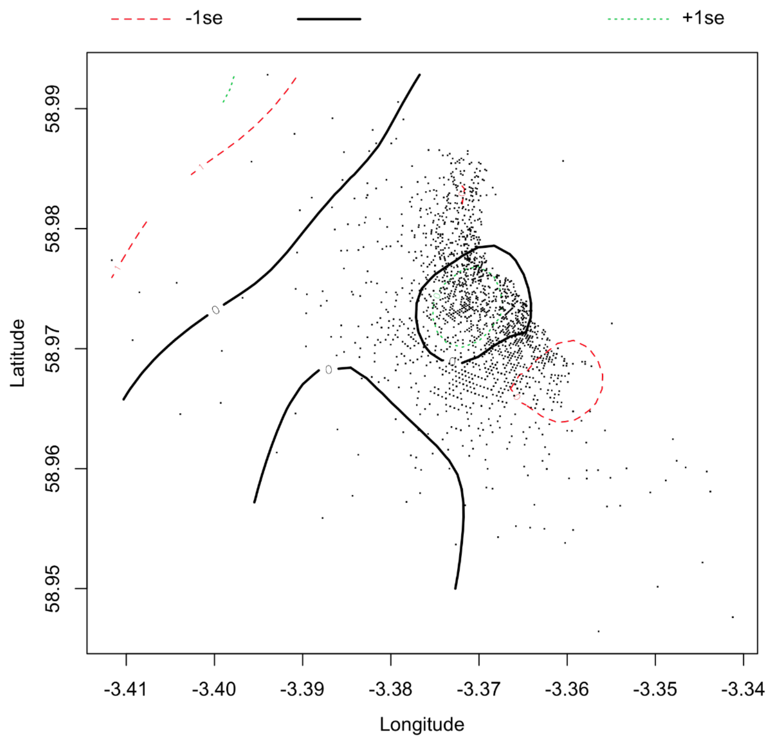


Figure 3.50: The estimated spatial pattern of relative number of black guillemot observed. The solid line is the smoothing curve for 0, red dotted lines are -1 standard error from the smoothing curve and the green dotted lines are +1 standard error from the smoothing curve.

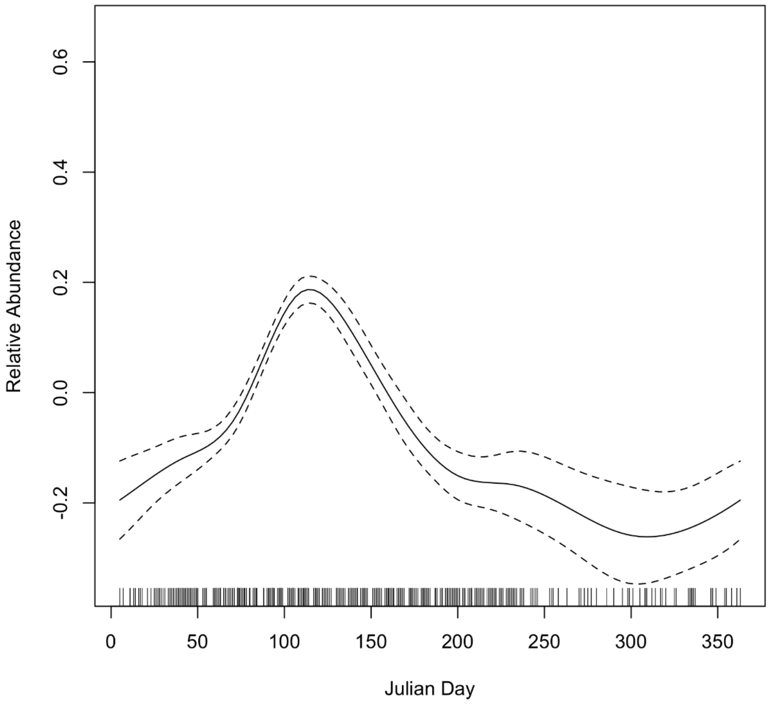


Figure 3.51: The estimated seasonal pattern of relative number of black guillemot observed. The solid line is the smoothing curve for Julian day and dotted lines are 95% confidence bands.

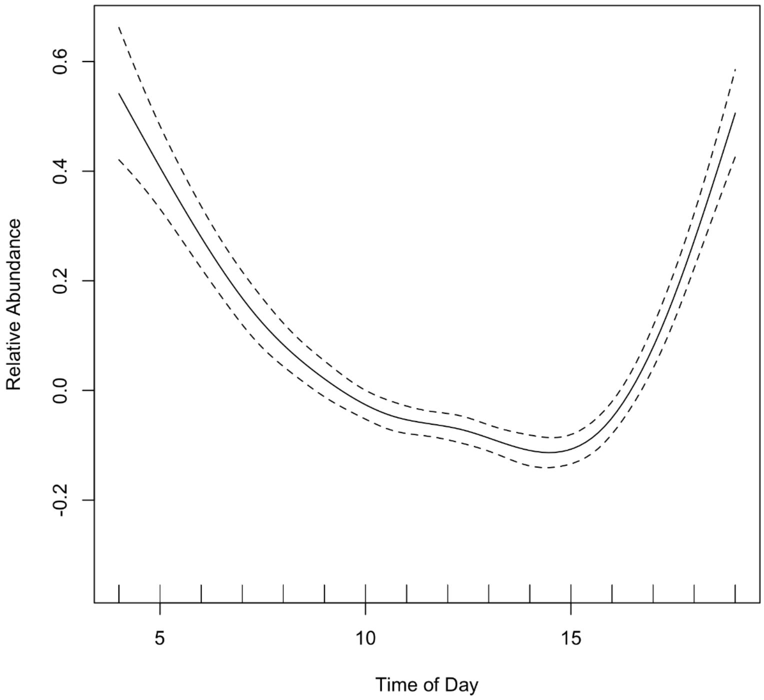


Figure 3.52: The estimated diurnal pattern of relative number of black guillemot observed. The solid line is the smoothing curve for time of day (hours) and dotted lines are 95% confidence bands.

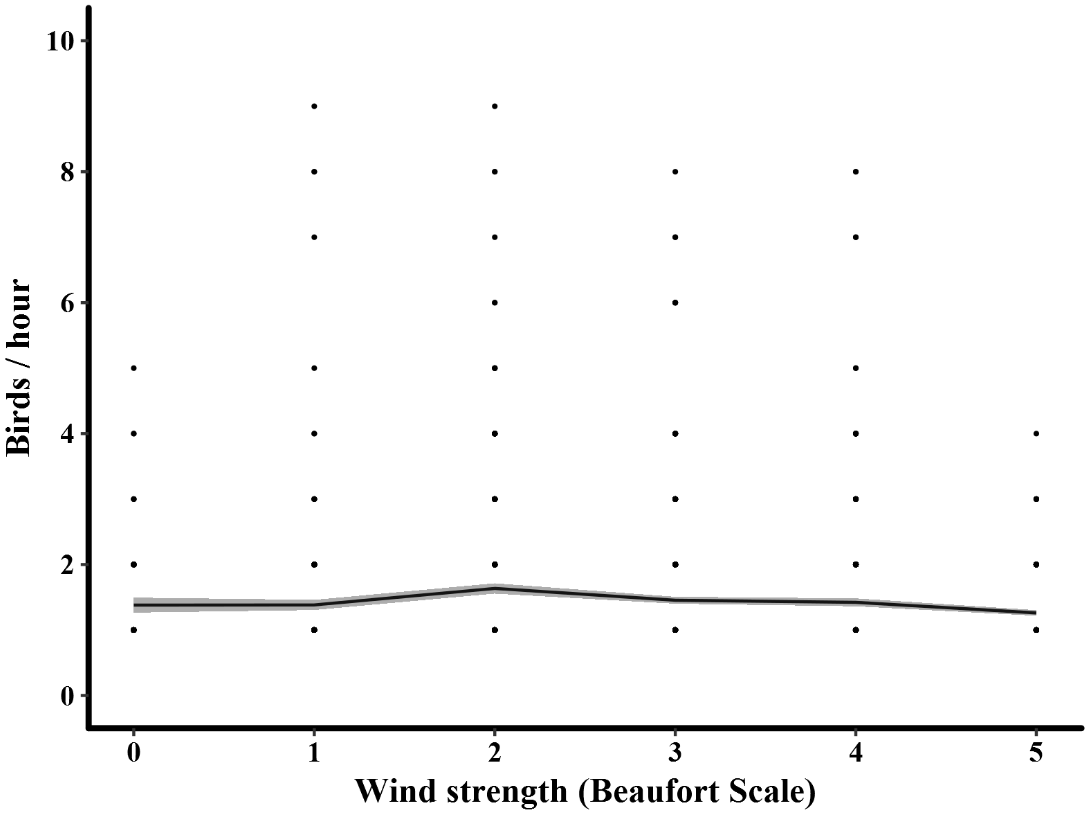


Figure 3.53: Mean number of black guillemots observed per hour during different wind strengths, using the Beaufort Scale, at Billia Croo.

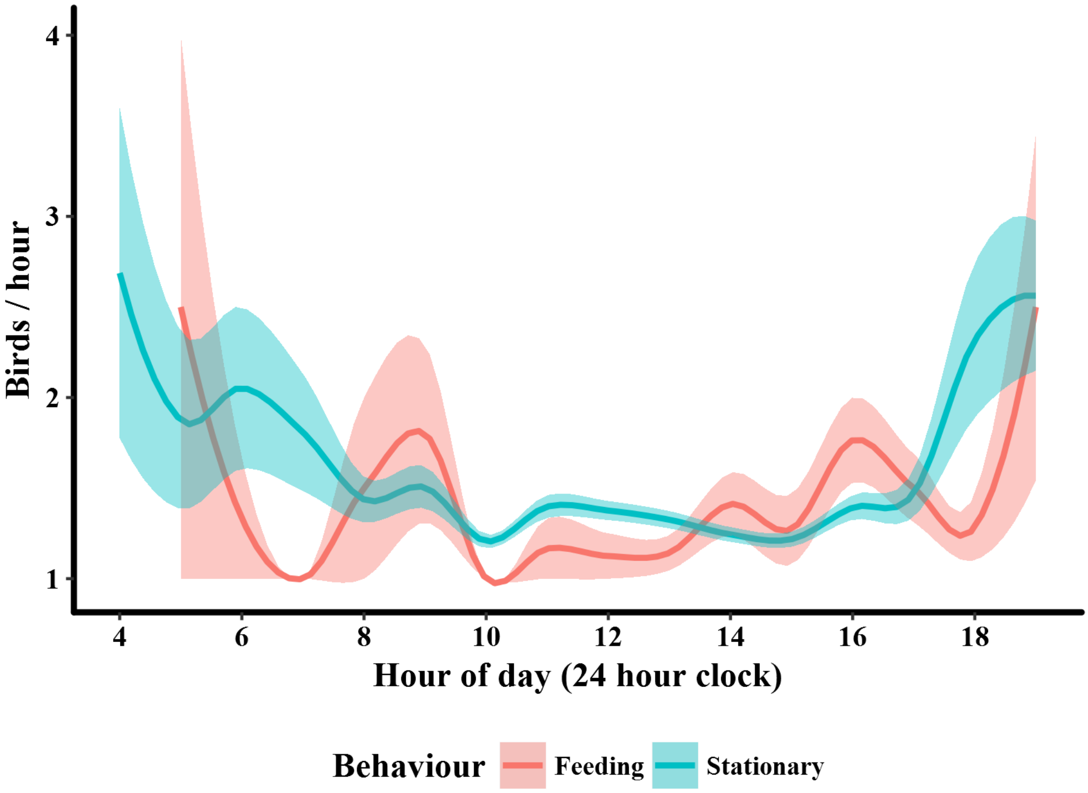


Figure 3.54: Mean number of feeding and resting black guillemot observed per hour, throughout the day at Billia Croo, with standard error shown by shaded areas.

3.3.14 Atlantic Puffin *Fratercula arctica*

A total of 690 puffins were observed at Billia Croo site. The best fitting model (auto-correlation $\rho = -0.06$) indicates that puffin numbers are a function of latitude and longitude, season, time of day, tidal state and glare extent, although not all these variables were significant (S.Tables 2.23 and 2.24). The spatial distribution of puffins across the Billia Croo test site, as with the other auk species, was concentrated in the central part of the site, directly west and within 2km of the observation point (Fig. 3.55 and S.Figs. 2.23 & 2.24), but this was not significant (*p*=0.07).

Puffins were observed more frequently during the spring and summer months (Fig. 3.56). This smooth, albeit with larger confidence intervals, shows a similar seasonal pattern to the other dispersive breeding auk species, with late spring/early summer peak in abundance. However, with the higher *p*-value thresholds, this was not significant (*p*=0.02). Figure 3.60 shows the mean number of feeding and resting puffins encountered per hour. This identifies a clear late morning peak in feeding activity, with a mean of 11 birds at 10:00. The number of resting birds shows little fluctuation throughout the day, and a minor increase in the mean number of birds encountered. Figure 3.61 suggests a consistent encounter rate of resting puffins across the tidal states. However, the encounter rate for feeding puffins fluctuates greatly: it clearly shows peaks in foraging activity at low tide and also an hour after high tide. Puffins also showed a non-significant diurnal pattern, (*p*=0.02), with numbers of puffins encountered increasing throughout the day to peak at approximately 15:00 Fig. 3.57). Another parameter that was not significant but with a visible pattern is puffin numbers relative to the tidal cycle. The smooth highlights a peak in numbers around low tide, which is not significant at the higher threshold (*p*=0.17). Finally, significantly more puffins were observed during moderate glare conditions (*p*=0.01) (S.Table 2.24 and Fig. 3.59).

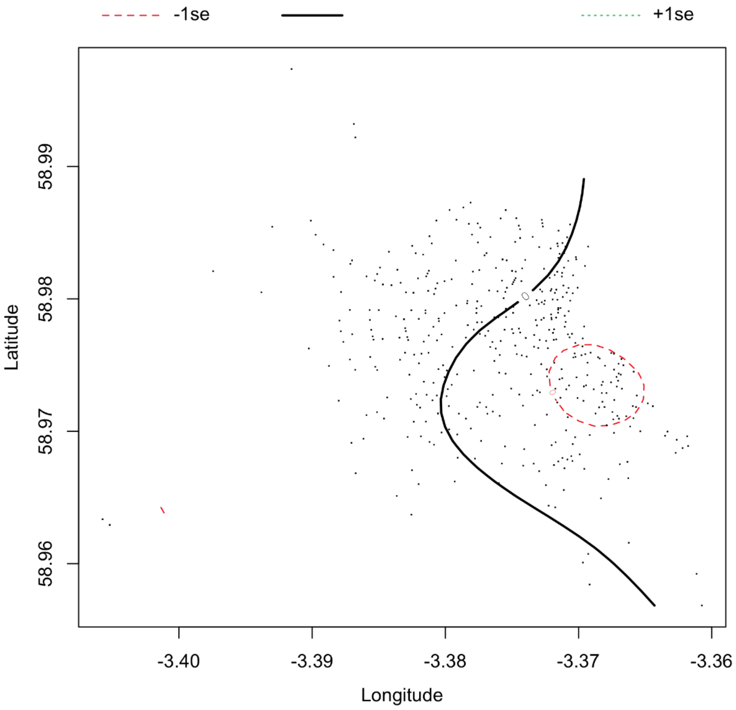


Figure 3.55: The estimated spatial pattern of relative number of Atlantic puffin observed. The solid line is the smoothing curve for 0, red dotted lines are -1 standard error from the smoothing curve and the green dotted lines are +1 standard error from the smoothing curve.

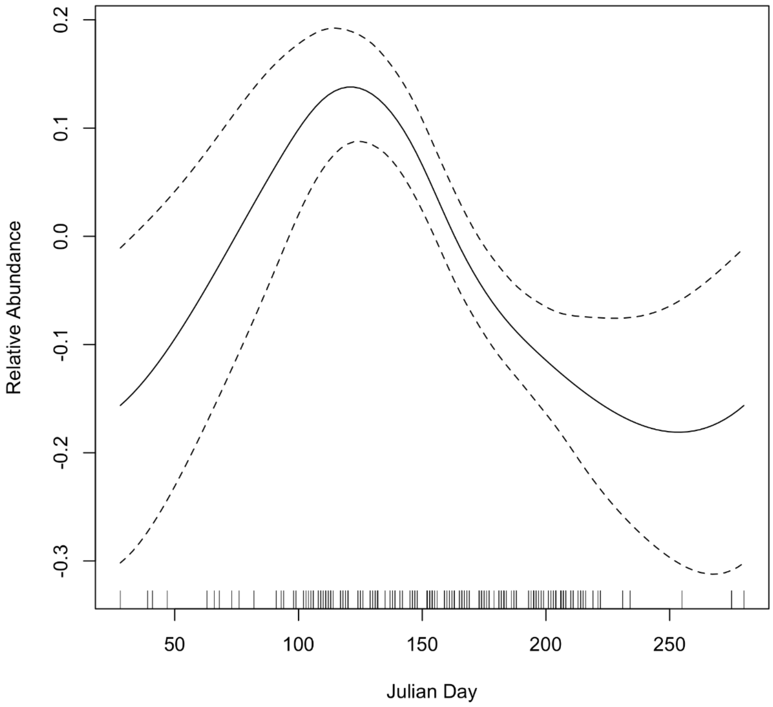


Figure 3.56: The estimated seasonal pattern of relative number of Atlantic puffin observed. The solid line is the smoothing curve for Julian day and dotted lines are 95% confidence bands.

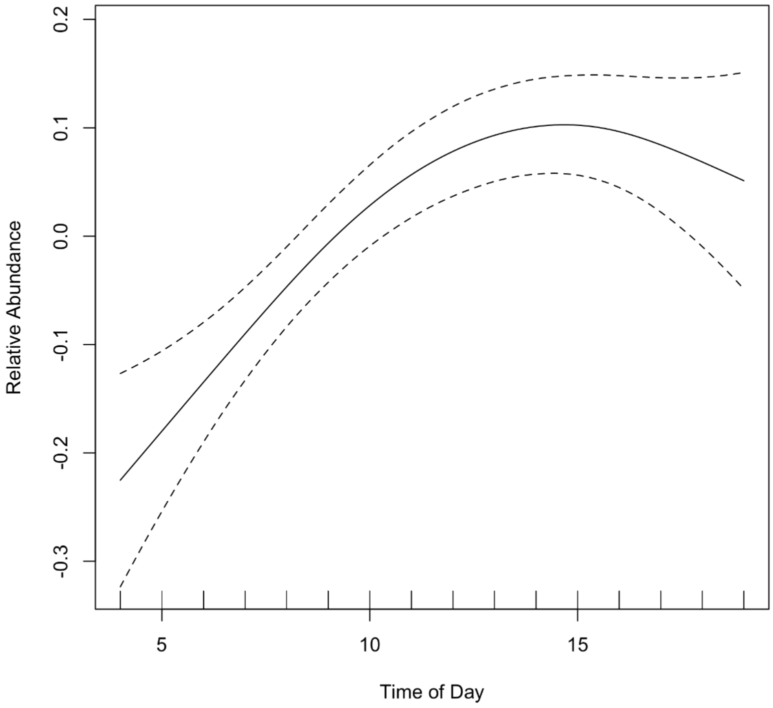


Figure 3.57: The estimated diurnal pattern of relative number of Atlantic puffin observed. The solid line is the smoothing curve for time of day (hours) and dotted lines are 95% confidence bands.

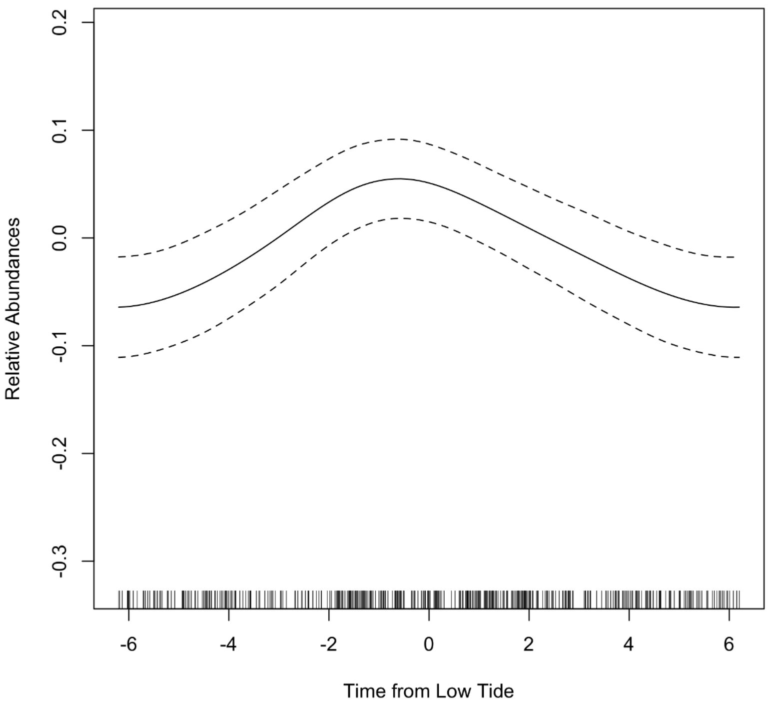


Figure 3.58: The estimated pattern of change in relative number of Atlantic puffin observed across the semi-diurnal tidal cycle. The solid line is the smoothing curve for time from low tide (hours) and dotted lines are 95% confidence bands.

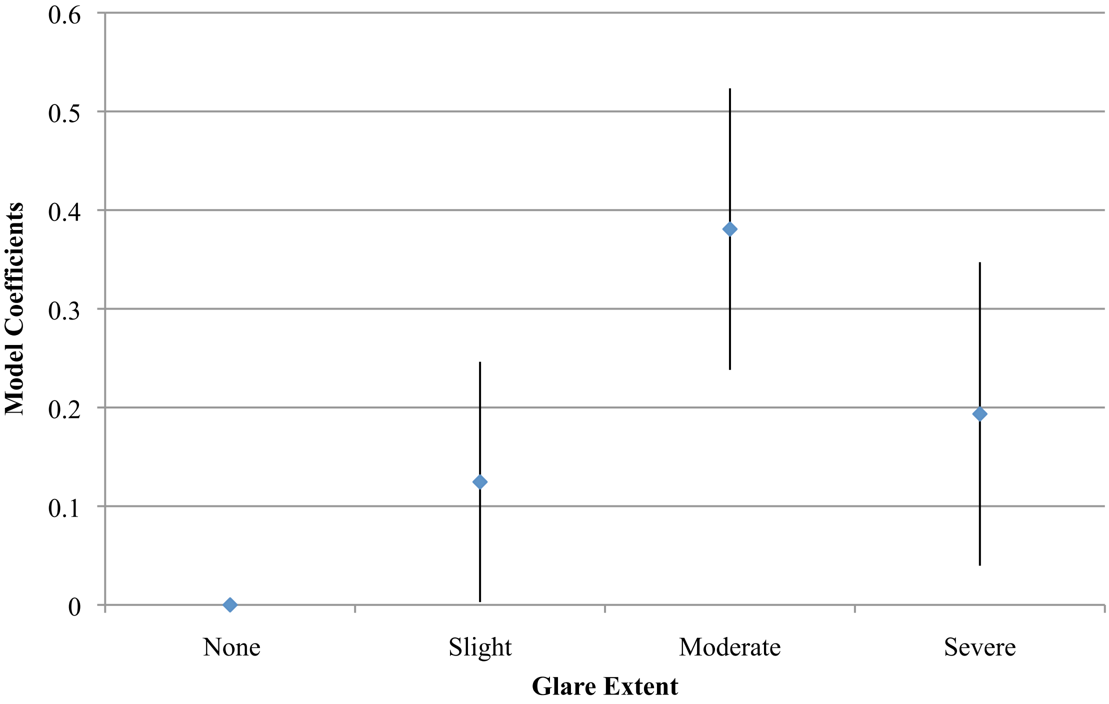


Figure 3.59: GAMM coefficient estimates (and standard errors) for Atlantic puffin observed by glare extent at Billia Croo.

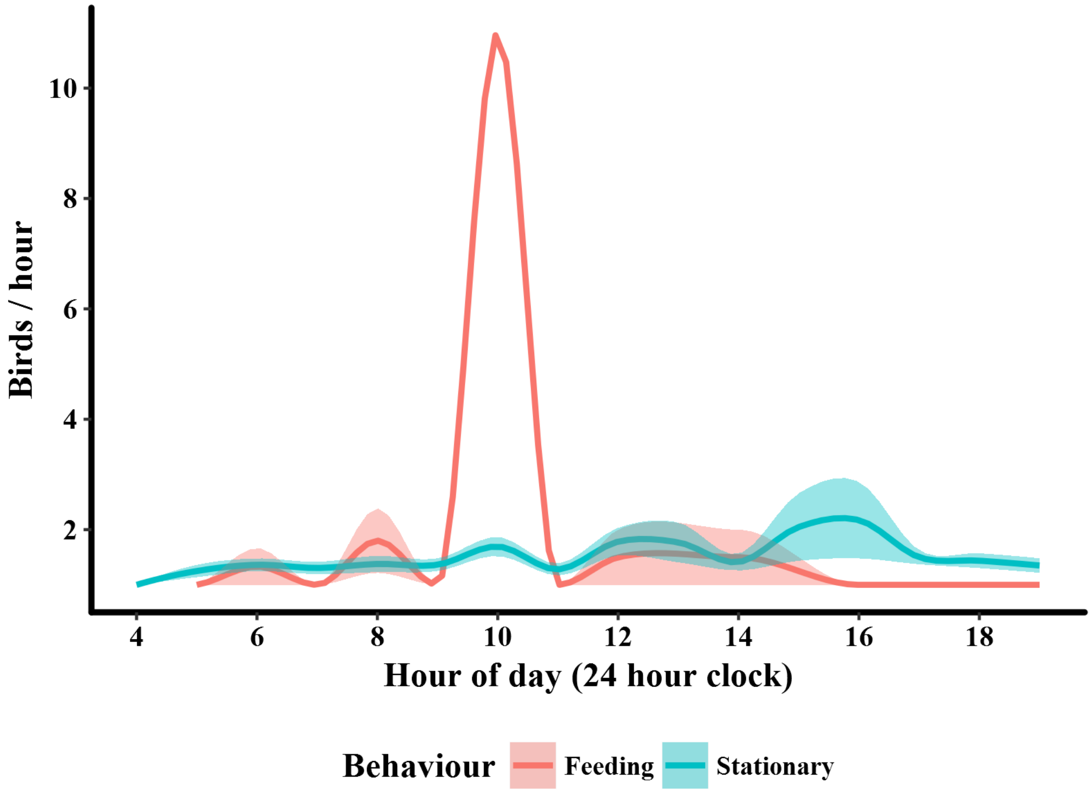


Figure 3.60: Mean number of feeding and resting Atlantic puffin observed per hour, throughout the day at Billia Croo, with standard error shown by shaded areas.

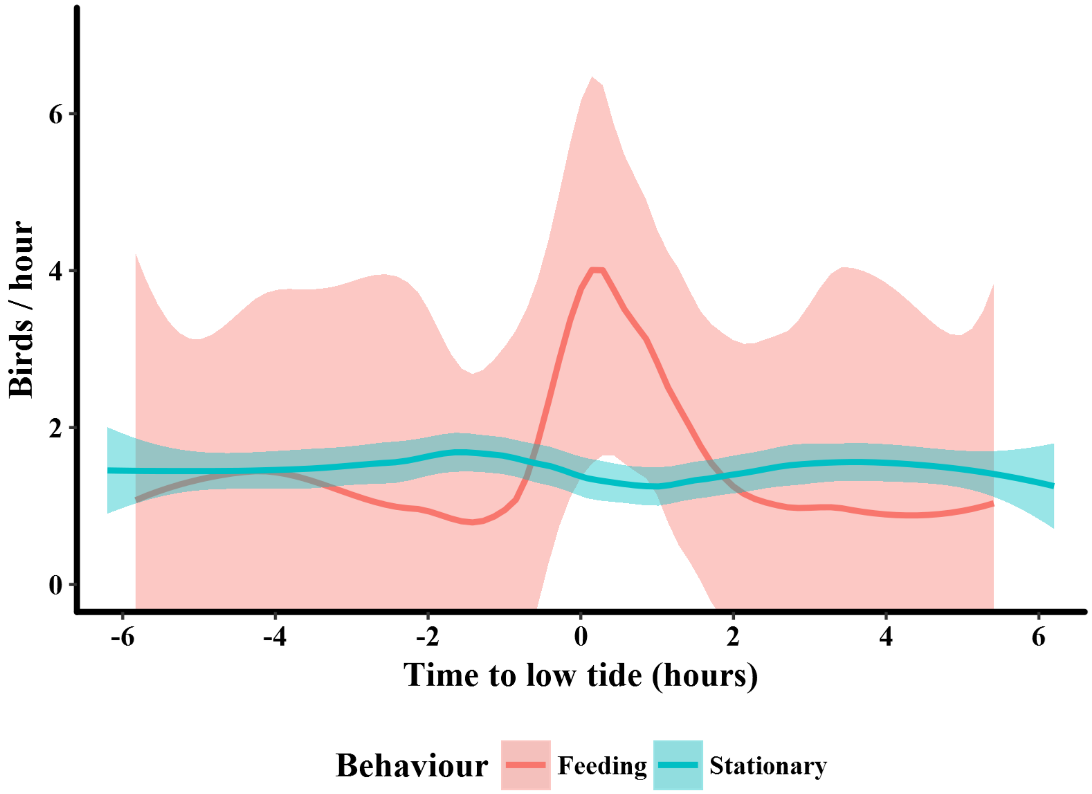


Figure 3.61: Mean number of feeding and resting Atlantic puffin observed per hour, by time from low tide at Billia Croo, with standard error shown by the shaded areas.

3.4 Discussion

Spatial and temporal patterns of the Billia Croo wave-energy test site use varied among species. This study provides the first comprehensive baseline information on the spatial and temporal distribution of the most frequently occurring marine bird species at the Billia Croo wave test site. We can use this information to understand and identify where and when particular species are more likely to encounter test devices or related deployment activity. For example, many species showed slight differences in the locations of 'hotspots' and the extent to which they used the wave test site. Shag, auk and eider sightings were concentrated between the Black Craig observation tower and/or off Breck Ness, which would increase the number of encounters of devices at berths in that area. Observed temporal variations, such as the clear and specific diurnal patterns of site use observed in fulmar, gannet, Arctic tern, black guillemot and puffin, also gives us further detail about potential interaction risks. This could inform decisions regarding time of deployment or operation of wave energy converters so as to minimise impacts to seabirds.

In general, spatial and temporal patterns reflected the breeding and wintering habits typical for each species, with the one notable exception of red-throated divers. For example the spatial results for eider, where consistent with literature, which notes that, as benthic feeders, they forage close to shore and in water up to 4m deep (Owen et al. 1986). While the more offshore patterns found in both kittiwake and fulmar is likely reflects the pelagic foraging nature of fulmars (reviewed in Weimerskirch 1998), which would take them farther than 5km offshore; and is consistent with kittiwakes generally being considered offshore feeders (Camphuysen et al. 2006). Interestingly, feeding great skuas were observed, off Breck Ness and the southern area of the site is feeding pattern mirrors that of locations of feeding gannets, which is a species commonly kleptoparasitised by great skua (Furness 1987). Such observations, according to the Billia Croo protocol, would be recorded with both species, i.e. a mixed flock, which would explain why there was only a significant spatial pattern for great skua when they were associated with another species. While red-throated diver spent a large amount of time foraging offshore, contrary to expectations presented by Gibbons et al. (1997) and Jackson (2002), which states that red-throated divers use the marine environment extensively outside the breeding season, spending a large proportion of their time on the sea, including sleeping (Stone et al. 1995), and forage on inshore waters during the breeding season (Gibbons et al. 1997; Jackson, 2002). This suggests that there may be some localised differences in the way species utilise high-energy environments and underlines the need to perform site and species-specific studies at proposed renewable energy sites to determine patterns that occur in different areas.

Seasonal patterns were found to be consistent with breeding phenology and dispersal or migration in most species. For example, eider were found to decrease in the spring; the onset of incubation starts around April, when females incubate eggs on land and males leave the area to congregate offshore while they moult (Owen et al. 1986). Gannet numbers were found to increase at start of the year to a peak in sightings around May/June which coincides with the gannet incubation period (Nelson 1978); the subsequent decrease after this period until a smaller peak in August coincides with the fledging period (Nelson 1978). Common guillemots and razorbills showed analogous significant seasonal patterns, which reflects their breeding phenology; by late June/early July many of the chicks have left the colonies and gone to sea with their fathers (Harris and Wanless, 2004). While shags, although a year-round resident (Forrester et al. 2007), were found to show seasonal variation with sightings steeply declining at the beginning of the year until early spring (i.e. March) when numbers increase. This coincides with main onset of egg-laying (Forrester et al. 2007). After this, numbers steadily increase until the end of the year, which may reflect the numbers in the area increasing due to fledglings. However, some of the patterns observed may show a level of bias in collection of these data; for example the unidentified *Larus spp.* smooth highlights a trough in sightings, with the lowest number of observations around July (~Julian day 175). This may reflect better weather conditions for observations during the summer, which would enable better identification of individuals to a species level.

The influence of environmental conditions was limited to tidal state, glare extent, wind strength and direction, and responses varied among species. Precipitation was the only parameter not found to have a significant influence on the distribution, abundance or occurrence of any species, however, this was expected as observations were typically paused during preciptation due to poor visability. In future studies I would recommend the inclusion of other habitat type variables, which could potentially influence foraging activity such as seabed substrate, slope or depth. Unfortunately, these variables were unavailable for inclusion in this study within Billia Croo test site.

Tidal state influenced the distribution of eight of the species (eider, red-throated diver, shag, kittiwake, common and black guillemots, razorbill and puffin). Some (red-throated diver and black guillemot) were more active relative to the high or flood tides while others were more strongly associated with the hours around low or slack tides. These differences could be attributed to species specific foraging or resting behavioral preferences. As this site is more suited to wave energy, the issues raised in chapter 2 regarding surveying across a fast-flowing tidal stream current would likely be less of an issue.

A significant relationship was found with glare extent and distribution and site usage of eight of the species/taxa in the study (gannet, shag, great skua, *Larus spp.* kittiwake, Arctic tern, common guillemot and puffin) and as such, was included in the preferred models. Although there is disparity between model predictions and observer counts there is clearly some differentiation among species. This could be explained by a reduced visibility over water that is caused by glare, which may have affected the observers' ability to accurately count birds and/or distinguish species, or under certain conditions could enhance some species (through back lighting) more than others. For example, larger species that are easily distinguishable, such as gannet may be easier to detect in stronger glare conditions than others, such as shags, which are harder to distinguish from great cormorants with a decline in visibility conditions. A further explanation is that the pattern observed may be an artefact of glare occurring at a similar time of day (due to the permanency of the view point direction), which if coincided with a diurnal pattern in bird activity, could result in some species may be seen more in stronger glare conditions while other species would be detected less frequently. However, it has not been possible within this analysis to disentangle this bias.

Finally, wind had limited effect on bird movement patterns with speed or strength only influencing two species and direction being important in the models for two other species. Gulls strong preference for the strongest winds was the opposite of black guillemots, which decreased as wind strength increased. This is probably a direct consequence of physical and flight differences between these two taxa. Interestingly, the two species affected by wind direction were both observed less frequently in offshore or westerly winds. This may be due to these winds precluding the birds' ability to forage within sight of the observation points or, it could be another artefact of observer error.

Wind direction affects sea state (i.e. causing “choppiness”) and this likely reduces the observers’ ability to observe and/or identify the species, thereby affecting counts.

The variation in all these results demonstrates some of the inherent concerns incorporated in land-based observational data. Firstly, there is a decreased probability of detecting wildlife with an increase in distance from the observation point (Bibby et al. 2000; Buckland et al. 2001). Therefore most land-based vantage point studies would result in a bias with the observer to detecting more birds closer to the shoreline and similarly being better able to observe behaviour of birds closer to the shore. Observers may also detect birds at a greater distance but not being able to identify them to species, or not detecting the birds at all, particularly if combines with declining sea or visibility conditions. At Billia Croo the use of the ‘big eyes’ to scan the site provides both stronger magnification, as well as a more accurate estimate of individual location; enabling a larger area to be surveyed with improved species identification to a greater distance compared with the Fall of Warness test site. However, as mentioned in chapter 2, it is not possible to incorporate distance analysis due to the difficulties in disentangling distance declines with the ecological gradient away from the coastline. The results for all land-based studies should be considered as a relative rather than absolute estimate, and spatial patterns should be interepreted with caution. It is also therefore recommended that boat-based surveys undertake line-transects randomly across the test site to calibrate these land-based vantage point observations. These surveys should be carried out according to standardised methodologies (e.g. Buckland et al. 2001 and Camphuysen et al. 2004). However, the survey protocol at this test site does overcome issues relating to the grid system, which prevents the inclusion of detection probabilities (and analysis of clusters of birds), as the spatial accuracy of the observations at the tidal test site is much coarser and it is possible that the usages of the most distant grid squares (from the observer) are being underestimated. At this wave test site there is also greater coverage of species, as it is not restricted to diving birds or birds interacting with the water surface. This also allows investigation of interactions with renewable devices for a wider range of species. It should also be noted that as with chapter 2 the results may not necessarily applicable to periods were no observations have been carried out, for example usage of the test site during the night is still unknown.

The effects of wind and glare conditions on visibility in ocean environments is clear and these factors could have affected how well observers detected or differentiated seabirds. This may account for the variations between model predictions and observational data. Another supporting factor for these conclusions is the influence of ‘Observer ID’ found in models for gannet, fulmar, shag, kittiwake and Arctic tern. Ideally these would be incorporated into a detection function, which would then be applied to a model exploring spatial, temporal and habitat associations.

3.5 Conclusion

This chapter provides a comprehensive model upon which to formulate further studies of such test and proposed renewable energy sites in order to understand potential impacts to marine birds. The improved ability to predict interactions between marine bird species and test devices or related deployment activity will vastly improve marine planning in the development of proposals planning the installation of marine renewable energy devices.

Chapter 4: Quantifying marine bird usage of a high-energy tidal stream, using shore-based vantage points

This chapter comprises my own work including data collection and analyses. A MRes student, Melissa Moore, assisted me for two months during 2011 in collecting the double-observer data. Dr. Grant Humphries performed the ‘random forest’ modeling to predict tidal current velocity and direction for the study period, and also assisted with R coding and ArcGIS mapping. My supervisors have assisted in reviewing and editing the chapter.

Supplementary materials for this chapter, including model outputs are in Appendix 3.

Abstract

Little is still understood about how diving seabirds vary spatially and temporally in their use of high-energy tidal environment. We have combined vantage point observations with modeling to describe these uses for five species. We identify under which tidal conditions species may be more likely to encounter tidal turbines. We found that four out of five species showed significant negative relationship with increasing current velocity and found different habitat associations between species. We outline approaches to overcoming some of the methodological challenges of surveying birds in a fast-flowing tidal environment. This work should lead to less biased assessments of the impacts of marine renewable energy developments, and is already being employed by the government agency, Scottish Natural Heritage.

4.1 Introduction

4.1.1 Marine renewable energy and seabirds

Marine renewable energy schemes, such as tidal-stream turbines, are predicted to have adverse impacts on birds including habitat loss, mortality through collision, disturbance, displacement and barriers to movement (e.g. Langston and Pullan, 2003; Garthe and Hüppop, 2004; Desholm and Kahlert, 2005; Gill, 2005; Langston et al. 2006; Wilson et al. 2007; Masden et al. 2009; Inger et al. 2009; Grecian et al. 2010; Langton et al. 2011). Diving marine bird species, such as black guillemot *Cepphus grylle*, razorbill *Alca torda*, European shag *Phalacrocorax aristotelis*, common guillemot *Uria aalge* and great cormorant *P. carbo* are considered to be particularly vulnerable to impacts from tidal developments (Furness et al. 2012). However, bird species are not equally susceptible to the different impacts of renewable schemes, presenting considerable challenges in devising guidance on siting developments. Some species may have higher risk of collision due to anatomical factors (e.g. size and agility), or factors affecting how and when they encounter the structures (Brown et al. 1992; Garthe and Hüppop, 2004; Lucas et al. 2005; Furness et al. 2012, Furness et al. 2013; Wade et al. 2016). In the context of wet marine renewables, species' flying and diving behaviour, foraging patterns, and seasonal distributions are all likely to affect if and how they might interact with devices, and their vulnerability to any negative impacts (Furness et al. 2012). Until recently, relatively few studies have been undertaken looking at foraging and habitat usage of high-energy tidal streams (e.g. Hunt et al. 1998; Holm and Burger, 2002; Zamon et al. 2003; Masden et al. 2013; Wade 2015;

Waggitt et al. 2016a; 2016b) and much of what drives foraging patterns and use of these high-energy tidal streams (HETS) is still poorly understood (Scott et al. 2013).

Commercial tidal-stream energy developments require mean spring tide current speeds of 2 ms^{-1} or more and depths of 25-50m (Fraenkal 2006), which often occur in coastal areas with complex bathymetry and oceanographic features. Both coastal topographic and bathymetric features can constrict and accelerate tidal current flows creating areas suitable for energy extraction, for example through straits (Adcock et al. 2013) and past shallow headlands (Lewis et al. 2015). High-energy tidal streams are often characterised by spatially and temporally localised, fine-scale oceanographic features such as kolks, boils, eddies, fronts, shearlines and tidal rips, which form principally as a consequence of the interaction between bathymetry and hydrodynamics in these coastally complex areas (Deeleersnijder et al. 1992; Wolanski et al. 1999; Benjamins et al. 2015). These features can be visible on the sea surface, such as slicks from upwellings and boils, shearlines between fast and slow-moving or turbulent waters, creating a complex mosaic of habitat, unique to each HETS.

4.1.2 Habitat associations

The association of seabirds with large-scale oceanographic features, such as shallow sea fronts has been well documented (e.g. Hunt et al. 1999; Piatt et al. 2006; Scales et al. 2014). In high-energy nearshore waters, similar processes also occur in association with the fine-scale oceanographic features (Holm and Burger, 2002; Zamon, 2003). These physical features enhance phytoplankton production and aggregate zooplankton and fish (Wolanski and Hamner, 1988; St John and Pond, 1992; St. John et al. 1992). This provides marine predators with a spatially predictable prey aggregation in a heterogeneous environment, which minimises foraging effort (Johnston et al. 2005). The “tidal-coupling hypothesis” predicts that current coastline interactions create both temporally and spatially predictable tidal rips and jets, thus creating predictable changes in zooplankton distribution, abundance or delivery rate (Zamon, 2003). Planktivorous fishes then exploit these peaks, which attract piscivorous predators, e.g. seabirds, in relation to the daily tidal cycle (Wolanski and Hamner, 1988). The depth of the water at these hydrographic features, and location of prey within the water column also determine which species try to exploit them (Ladd et al., 2005; Daunt et al. 2006). Holm and Burger (2002) explored foraging behaviour in areas with strong tidal currents focusing on whether stronger currents and turbulent waters would inhibit foraging in different diving birds; they found different species exploited different states of the tidal cycle. The exact locations of these zones vary seasonally, in response to different weather conditions, and with changes in currents due to the tidal cycle (Daunt et al., 2006). Studies have found that species vary their foraging behaviour with both ebb and flow states but also between spring and neap tides, which is likely to relate to the concentration and accessibility of prey (e.g. Irons, 1998). Inter- and intra-seasonal variation in seabird foraging distributions can also be driven by prey characteristics and energetic demands of the breeding cycle e.g. incubation vs chick-rearing (Robertson et al. 2014).

4.1.3 Assessing impacts of marine renewable developments

Marine renewable energy schemes in Scottish waters are subject to consenting and licensing processes under both EU and Scottish legislation. Developers are required to assess their developments through an Environmental Impact Assessment (EIA) (required under Section 36 of the Electricity Act 1989) and Habitats Regulation Appraisal (which may be required under the Conservation (Natural Habitats, &c.) Regulations 1994). This assessment is provided within an Environmental Statement, for a number of 'environmental receptors' including marine birds. However, the very nature of these high-energy tidal stream (HETS) sites that makes them so suitable for energy extraction also gives rise to a number of methodological issues, which must be overcome to produce accurate assessments of potential impacts.

Lessons have been learnt from inconsistencies in environmental assessments and monitoring of both terrestrial and offshore wind farm schemes, particularly while the industries were in their infancies. As a consequence, there have been calls from researchers to improve the standard of EIAs for renewable energy schemes (e.g. Langston et al. 2006; Stewart et al. 2007, Maclean et al. 2013; 2014; Masden et al. 2015; *R (RSPB) v The Scottish Ministers [2016]*[1]), by incorporating replicated balanced experimental designs. Much of the work on assessing impacts of wind farm developments has failed to include either a 'control' site or to undertake vital pre- and post-construction monitoring, which reduces the power of any analyses and hinders evaluation of both immediate and long-term changes on the environment. In order to assess potential impacts of tidal developments, developers are required to undertake 'site characterisation', which enables the assessment to focus on key receptor species. For marine birds, this has typically involved regular surveys of the development area and a buffer around it, either from shore-based, boat or

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[1] *R (RSPB) v The Scottish Ministers [2016] CSOH 103*

aerial platforms. Site characterisation surveys are undertaken across seasons, typically over a period of two years (Jackson and Whitfield 2011). They enable a quantitative assessment of the usage of the site by different species and an exploration of the magnitude of potential impacts *e.g.* density estimates used in collision risk models. While these baseline data can also be used as part of impact monitoring, survey designs should be led by clear objectives and questions, and whether they are to inform site characterisation, baseline or impact monitoring (Nuuttila 2015).

The two approaches most often used for monitoring impacts of developments are the ‘Before-After-Control-Impact’ (BACI) approach and ‘Before-After-Gradient’ (BAG) approach (e.g. Ellis and Schneider 1997; Drewitt and Langston 2006; Fox et al. 2006; SNH 2009; Barton et al. 2013). The first approach uses control and impact sites, to monitor before and after the impact to detect changes post impact. In addition, to issues with pseudoreplication, this approach is considered to have little power to detect smaller and/or gradual impacts, and particularly displacement of seabirds around a development (Underwood 1994; 1999), and it is has proven difficult to identify independent but comparable control sites. Consequently, the emphasis from a BACI approach has recently shifted towards a BAG design. This involves monitoring a larger area around the development site, assuming that potential impacts will decrease with increasing distance from the source (Barton et al. 2013; Oedekoven et al., 2013). In the context of wave and tidal sites this is particularly important, given the often unique complexities to bathymetry and oceanographic features, which can influence marine bird usage of an area, making it difficult to find suitably comparable control site(s) (Waggitt et al. in prep, refer to Appendix 6).

4.1.4 Vantage point surveys

Vantage point surveys are more commonly used for near-shore wave and tidal developments, in comparison with boat and aerial surveys. The former can be considered akin to point transect surveys, where the point consists of a viewshed over the water only, rather than a full circle around the observer or recording point (Oedekoven et al. 2013). They benefit from being fairly cost-effective and are more logistically feasible for monitoring some of the smaller, inshore development sites than boat or aerial surveys. However, unlike the ESAS and Cowrie guidelines for boat and aerial surveys (e.g. Camphuysen et al. 2004; Maclean 2009), there is no standarised methodology for undertaking vantage point surveys or analysing data. While Jackson and Whitfield (2011) outline some of the methods used for tidal developments in Scottish waters, these methods are often adapted to suit practical considerations of different sites, for example the location of a suitable vantage point, distance to development site, speed of tidal currents, accessibility and safety of the site to observers, frequency of observations. All vantage point surveys contain as a minimum, scans of the site recording birds and their locations, sometimes referred to as “snapshot” scans. However, as chapter 2 and 3 highlight, methods for scanning and recording birds within the sites (e.g. grid vs. bearing and distance), as well as the species and behaviours recorded can vary, presenting issues for analysis and as found in Waggitt et al. (In Prep) can lead to difficulties in comparing results between development sites.

There are a number of difficulties in surveying bird usage and analysing data collected from high-energy tidal streams. The environment is highly dynamic, resulting in both temporal and spatial heterogeneity, making both survey and modeling challenging. Seabirds themselves are relatively easy to observe and count, particularly due to being centrally-placed foragers, enabling numerous tracking studies to be undertaken and are an often used focus for ecological modeling (e.g. Wakefield et al. 2009). While tracking studies are excellent for defining connectivity between breeding colonies, foraging locations, and potential development areas, the sample sizes, potential for birds from neighbouring colonies (and non-breeding birds) to also use a development area, mean that surveys are still required on development sites to define abundance and inform impact assessments. Chapters 2 and 3 highlight the slightly different survey designs at the EMEC test centres: the Fall of Warness uses a 500x500m grid system, focuses on species that are considered to be more vulnerable and also that are interacting with the surface; Billia Croo uses triangulation to achieve more precise locations and records all species observed. Other issues include adequate temporal coverage; over time of day and tidal cycle (ebb-flood and spring-neap), behavioural bias of species such as gannet which tend to search for food from the air and only interact with the surface when they plunge dive. These previous analyses highlighted the pitfalls of inconsistent recording of weather variables and the importance of setting out clear survey objectives and accounting for observer effort.

4.1.5 Current approaches to analysis

Distance sampling and capture-recapture approaches are the two main methods for estimating animal abundance (Buckland et al. 1993; 2001; 2004; Thomas et al. 2010). However, at present there are some fundamental issues in applying these approaches to shore-based vantage point surveys of high-energy sites as these data violate a number of the key assumptions of distance sampling, leading to unreliable results. Principally the conventional distance sampling methodology requires animals to be independently uniformly distributed around the search or vantage point, and the point(s) to be distributed randomly (Buckland et al. 2001; 2004). Shore-based vantage points of areas of sea, much like surveys from roads or paths violate this assumption due to the ecological gradient, and therefore bias density estimates (Marques et al. 2010, Cox et al. 2013). The difficulty lies in trying to disentangle the decreasing detection of birds over distance across an often unknown ecological gradient. Currently this presents a key analytical limitation to the marine renewable industry. Some suggest a 2km distance from the observer does not affect accuracy of bird detection (Jackson and Whitfield 2011; Waggitt et al. 2014; Oedekoven et al. 2013) and that factors such as turbulence and sea state have a bigger influence (Waggitt et al. 2014). Some approaches have been developed to deal with vantage point analysis issues, such as nupoint (Cox et al. 2013), which estimates both detection function and ecological gradient, however this package is not being actively maintained or developed (Oedekoven et al 2013; Nuuttila 2015). Another promising approach combines distance sampling and mark recapture methods (Marques et al. 2012), spatially explicit capture recapture (SECR). This uses a double –observer platform to capture and recapture sightings of birds, alongside habitat modeling to try and disentangle the non-uniform distribution from imperfect detection (Borchers et al. 2015). The MRSea package has been developed for modeling spatially explicit change, particularly for baseline characterisation and EIA assessments, which goes a long way to standardising analysis methods for assessing marine renewable impacts. However, to date it has not been standard practice to incorporate distance sampling into analysis of shore-based vantage point data (Oedekoven et al. 2013). Further assumptions that are also potentially violated include complete detection at g0 (due to the position of the VP and coastline gradient), the assumption that animals aren't moving, which can be violated if they are actively travelling or passively sat on the water surface and drifting with the current. In addition there are other issues common to boat and aerial platforms that apply to vantage points, such as identification to a species level, accounting for availability bias for species that dive underwater and therefore may not be observed, which can be taken into account.

4.1.6 Aim of chapter

The aim of this chapter is to investigate temporal, spatial and habitat use of a high-energy tidal stream, suitable for renewable development, for key diving seabirds. In addition, I explore the use shore-based vantage point methodologies to try to provide a better understanding of bias and limitations within data collection, such as surveying across a fast moving body of water. The previous two chapters used shore-based vantage point methods to determine habitat usage of wet renewable test sites, however since those data were collected and the analyses carried out there have been many developments in our understanding of the methodological issues, as well as the statistical approaches that are used. Consequently, methods used in this chapter have been developed based on the lessons learned from analysing data from the previous two chapters. I therefore, also aim to both identify issues with methodologies and attempt to develop some solutions.

4.2 Methods

4.2.1 Study Site

This study was undertaken from six shore-based vantage points along the length of Bluemull Sound, Shetland, UK (60º 42’8.2”N, 0º 58’53.9” W) (Fig 4.1). Bluemull Sound is situated between the islands of Yell and Unst and is characterised by fast flowing tidal currents, which can exceed 2.5ms^{-1} and run linearly in a north-south direction (Fig 4.2 and 4.3). They create temporally localised tidal micro-habitats including a shearline off of the Culli Ness headland (Halliday, 2011; Robbins, pers. obs.). The coastline is mostly cliff, with a number of more sheltered inlets and bays, with mixed sediment habitat in these areas (mud, sand and gravel). Bluemull Sound is ~6km long from the northern entrance to the isle of Linga in the southern entrance. The width of the strait varies from 1.7km by Papil Bay to 0.8km by the Point of Grimester. I have observed fulmar, shag and black guillemots breeding on Blue Mull cliffs at the northern end of the sound, scattered black guillemot nests along the sound, and fulmars breeding in the Gutcher quarry cliffs at the southern end. The nearest large seabird colonies, including breeding gannets, auks and kittiwakes, are at Hermaness to the north and Fetlar to the south. Red-throated divers breed on lochans on Yell and Unst (Pennington et al. 2004). The area has also been identified as a proposed Special Protection Area (Bluemull and Colgrave Sounds pSPA) due to it’s importance as a feeding ground for over 15% of the British population of breeding red-throated divers (Scottish Natural Heritage, 2016a). The sound provides commuting seabirds one of the few shortcuts between breeding colonies and foraging locations on the eastern and western aspects of the Shetland archipelago.

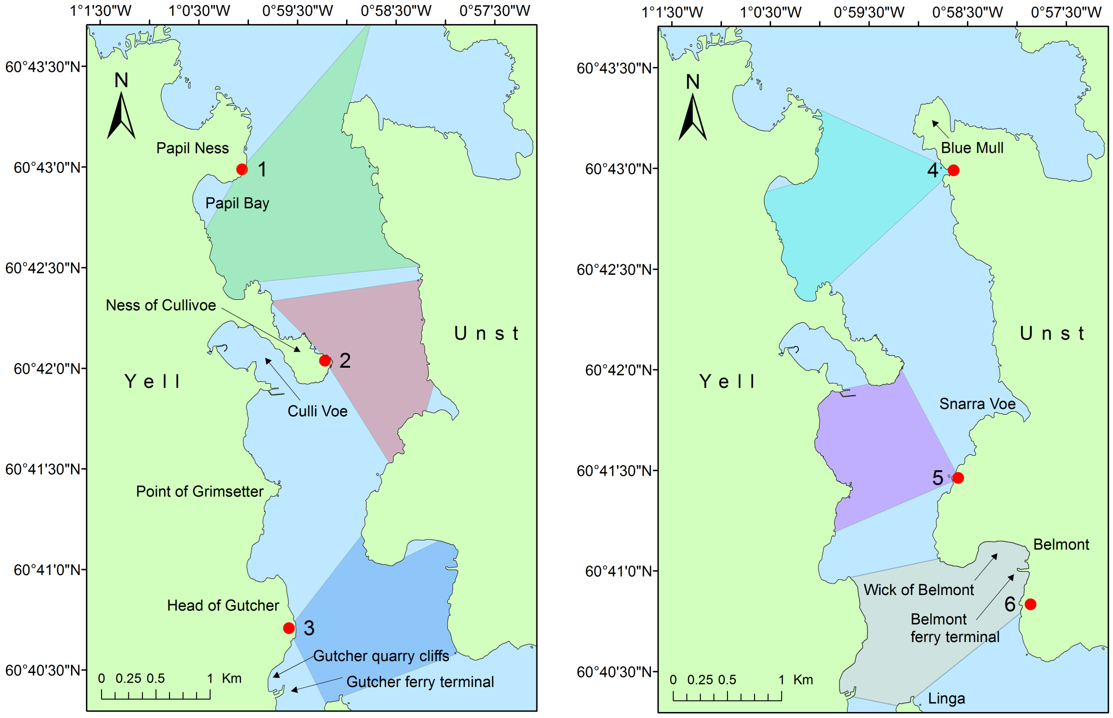

Figure 4.1 Bluemull Sound, Shetland (60º 42’8.2”N, 0º 58’53.9” W). Vantage points 1-3 and their viewsheds are shown in figure a); with vantage points 4-6 and their viewsheds used during summer 2012 shown in figure b).

The observations for this study were undertaken prior to the planned deployment of a tidal stream device. At the time of the surveys the sound had been identified as development area for the world’s first community tidal turbine, the energy from which is being used to power the ice plant at Cullivoe Harbour. A three-bladed 30kW Nova Innovation device, owned by Bluemull Tidal Energy Limited (a subsidiary of North Yell Development Council), was deployed approximately 1km from Culli Ness at a depth of ~30m (http://www.northyell.co.uk/tide.php). In August 2016 two 100kW devices had just been deployed becoming the first tidal turbines in an array (http://www.bbc.co.uk/news/uk-scotland-scotland-business-37212373). It was intended to undertake observations before and during construction/deployment of the tidal stream device, however deployment of the device was delayed and eventual deployment was after all surveys had been undertaken. There is a designated port at Cullivoe Harbour, which is used predominately by salmon and shellfish processing industries. The vantage point layout was designed to cover the length of the sound and enable a BAG around the proposed deployment location. Vantage points 1 and 4 cover the northern entrance to the sound, and vantage points 2 and 5 cover the area around Culli Ness, including the tidal development area. Vantage point 5 also covers the harbour entrance. Vantage points 3 and 6 cover the southern end of the sound where a fish farm is located and also the route for a short regular ro-ro ferry between Gutcher, Yell and Belmont, Unst, which runs throughout the day.

4.2.2 Marine bird observations

In total, 269 shore-based scans were performed at six vantage points in the study area (refer to figure 4.1). Data were collected on 56 days between 3rd June and 26th August 2011, 14 days between 6th January and 6th February 2012, and 55 days between 30th April and 3rd August 2012 (refer to table 4.1 for breakdown of observations by vantage point). Dates between April and August were considered to represent the breeding season, acknowledging that this may including “shouldering” periods on one, or other or both sides of the actual breeding season for some species. The observations in January represented the non-breeding season. This chapter focuses on analysis of site scans, sometimes referred to as ‘snapshot scans’ (Jackson and Whitfield, 2011), analysis of the focal observations is undertaken in Chapter 5. Vantage point locations were determined based on maximizing coverage of the sound and suitability of elevation (VP1 7m; VP2 7m; VP3 18m; VP4 47m; VP5 17m; VP6 10m above sea level). The mean width of the viewsheds (from observer to opposing shoreline) for all vantage points was between 0.8-1.5km, the maximum width of all VP’s was 1.8km in VP1. Due to the issues of declining detection with distance it is preferable to have viewsheds of <2km to ensure identification of birds to a species level and to undertake behavioural observations (Waggitt et al. 2014; 2016a). Vantage points 4-6 were only used during a double-observer data collection period during summer 2012. Of all the birds (*n*=6368) recorded during observations 7.08% were unidentified to a species level; 4.7% were *Sterna spp.* (identified to common/Arctic tern); 2.4% were *Larus spp.* in flocks associated with fishing vessels; Auk spp. and *Gavia spp.* were 0.2%.

Table 4.1 Number of scans undertaken at each vantage point along Bluemull Sound, Shetland during 2011 and 2012.

Vantage Point	2011	2012
1	30	34
2	31	49
3	30	52
4	-	14
5	-	10
6	-	19

Scans of the site were made using an Optolyth TBS 80 telescope with a fixed, angled 30x lens. Scans were undertaken by systematically sweeping the site from one shore to the other moving from left to right (a North-South direction for VP’s 1-3 and South-North direction for VP’s 4-6). It was initially decided to start each scan from the same start point and to sweep the site in the same direction, regardless of current speed and direction, to ensure consistency and repeatability of scans. However after several scans under different tidal current conditions, it was determined that a correction would be needed to account for a different area of sea surface being scanned depending on current velocity and direction (see 4.2.4).

Attempts were made to ensure adequate temporal coverage across both time of day and stage of tide at all vantage points, across all seasons (refer to table 4.2 and 4.3). Each observation consisted of recording environmental variables, a whole site bird scan, and a minimum of five focal observations. Where possible, surveys were under taken over consecutive days but this was not always possible due to weather conditions. Surveys were typically limited to weather conditions of sea state 4 or less. Scan start and end times were recorded along with the duration of any scan pauses, e.g. during periods of rain. The mean scan length was 29 min (range 8-78 min). While every effort was made to scan at the same velocity, the differing scans lengths were due to the different area of the viewshed at each vantage point and number of birds observed (and recording time).

Table 4.2 Number and duration of observations undertaken along Bluemull Sound, Shetland during 2011 and 2012 by hour from high tide.

Hours from High Tide	No. of Scans	hh:mm
-6	17	8:31
-5	24	13:56
-4	20	10:20
-3	18	8:45
-2	20	9:14
-1	29	11:43
0	35	14:56
1	33	13:27
2	17	8:52
3	17	10:18
4	19	11:50
5	20	10:04

Table 4.3 Number and duration of observations undertaken along Bluemull Sound, Shetland during 2011 and 2012 by stage of tide and time period. Time period was delineated by approximately the half way point between the start and end of observations to identify eveness of scans across both tidal and diurnal cycles.

	Ebb		**Flood**	
Time (GMT)	**No. of Scans**	**hh:mm**	**No. of Scans**	**hh:mm**
04:00-13:00	76	12:29	64	7:55
13:00-21:00	63	7:07	66	8:25

Bird positions were located by triangulation using angle and distance from the observer. It was decided to adopt an approach akin to that used for Billia Croo, providing locations for each sighting rather than the grid cell approach at the Fall of Warness. This was to increase precision of bird locations and any spatial modeling but also as a grid could be applied retrospectively. As a bird or group of birds was sighted, the magnetic bearing of the sighting from the observer was measured using a handheld compass. This was retrospectively corrected for analysis to account for annual differences between grid north and magnetic north. The distance of the bird or group of birds from the observer was measured via graduated rangefinder modified from those described by Heinmann (1981). The distance rangefinder sticks are a piece of stiff wood or plastic with set distances marked along the length (like a ruler) at the corresponding ‘calliper interval’. The calliper interval is calculated using a formula (Heinmann, 1981) taking into account the height of the survey vantage point, distance to the horizon and length of the observer’s arm. These graduated ‘distance sticks’, which are now commonly used on boat surveys by marine mammal observers, are held at arms length with the top of the stick held level with the horizon, in the direction of the sighting. The distance of the animal is then measured along the scale from wherever it is located along the stick. However, as the horizon was not visible at any of the sites, the distance between the VP and a number of given locations on the opposing coastline was calculated to ensure that the distance measuring stick was positioned correctly for measuring any given angle of the viewshed. Distance rangfinder sticks were created for each vantage point, taking into account the height of the survey vantage point, distance of the furthest point of the opposite coastline and length of the observer’s arm. Calibration and accuracy of the sticks can be ensured by testing distance estimates against objects of known distance, such as navigational buoys. However, there was only one fixed object within the water that I could use for calibration and accuracy (the fish farm near Belmont). Consequently the rangefinder sticks for VP 3 and 6 were calibrated using this and I used known distances along each shoreline (at specific angles) to ensure accuracy for all the other VP rangefinder sticks.

Bird behaviour was recorded based on the ESAS ethogram (Camphuysen et al. 2004) and retrospectively classified into “foraging”, “travel” and “loafing”. Foraging behaviours included any birds that dived, surfaced, had prey items in their bill or were actively searching; travel behaviours included birds landing, taking off or actively swimming in any direction; loafing behaviours included resting, sleeping, preening and bathing, other behaviours recorded included obtaining nest material (e.g. kelp) and mating displays.

During the breeding season 2012 double-observer data (from the same VP) were collected with the intention of utilising a spatially-explicit capture-recapture (SECR) analysis (e.g. Borchers 2012; Borchers et al. 2015). However, the R package for this is not yet able to analyse data from multiple vantage points, so this could not be achieved within the timeframe for this PhD thesis. Consequently, only single-observer data were used in this analysis, with the intention to utilise the second observer data for the double-observer SECR analysis in the near future.

4.2.3 Environmental characteristics

Due to the nature of the study, environmental characteristics’ data (aside from weather observations) were obtained retrospectively and added to the dataset prior to analysis. Weather variables recorded included sea state (using Beaufort scale), wind direction (bearing), speed (mph), precipitation (scans were paused during periods of precipitation), cloud cover (% cover), visibility (scale of 1-5, with 1 being poor and 5 being excellent), glare (including bearings from observer of sea surface affected). Of those recorded, sea state, precipitation and glare were incorporated into the modeling (see 4.2.5).

4.2.3.1 Hydrodynamic conditions

An Acoustic Doppler Current Profiler (ADCP) deployed off of Culli Ness in Bluemull Sound in 2004 was used to analyse tidal current velocity and direction data, as there were no concurrent tidal data available for any of the observations. The device was deployed on behalf of Shetland Islands Council, resulting in a 15 day data series in 1m vertical bins between 23rd March and 7th April 2004. To extrapolate the appropriate current velocity for each observation a random forests model was firstly trained on the ADCP 2004 dataset using current velocity in ms^{-1} (with negative values used for a southerly direction and positive values used for a northerly direction) as the response variable. The explanatory variables used were time to next spring tide (minutes), time to next tide stage (either low or high, minutes), height of the next tidal cycle and current direction. The latter was added as a weighting parameter to ensure predictions were created with correct sign. The current direction pattern observed from the ADCP dataset was a southerly direction either side of high water; and a northerly direction either side of low water. Slack tide was therefore calculated as the midpoint between high and low tide events (estimated to be 175 minutes after each), and delineated where the direction shift occurred. Using the observation start and end times, the time from nearest low or high tide was calculated and the direction extrapolated from the pattern observed. The model predicted tidal current velocities well, with internal cross validation on out of bag estimates explaining 94.93% of variance in the model; a 5-fold cross validation gave mean Pearson's Rho 0.96±0.019SD. Using the same explanatory variables calculated for 2011 and 2012 the tidal current velocity was forecast for the observation data times. The random forests analysis was undertaken in R (version 3.1.3, R Development Core Team, 2015) using the 'randomForest' package (Breiman and Cutler, 2015).

4.2.3.2 Habitat characteristics

There were no digitized habitat data available for the study site. MIKE 21 hydrodynamic output maps from Halliday (2011) were geo-referenced in ArcGIS (version 10.3.1) and used to characterise the sound by root mean tidal velocity and directional flow type (Fig 4.2 and 4.3). The root mean cubed velocity, (hereafter 'current speed habitat') was considered by Halliday (2011) to be a more useful and temporally representative parameter to the renewable industry, as a weighted average, rather than maximum current velocity, which models the monthly peak flow rate and only occurs for a short time. The sound was classified into five current speed habitat classes: very slow (0-0.5ms^{-1}), slow (0.5-1ms^{-1}), moderate (1-1.5ms^{-1}), fast (1.5-2ms^{-1}) and very fast (>2ms^{-1}). The directional flow type, hereafter 'directional flow', is a magnitude between 1-10 relating to ratio of flow along principal axis to flow across the principal axis. A value of 1 indicates a linear flow with a flood and ebb direction, while a value of 10 indicates a circular flow rotating equally through all directions (Halliday, 2011). Bathymetry data were obtained under licence from EDINA Marine Digimap service (SeaZone Solution Ltd, 2014).

The lack of digitized habitat data and the approach that had to be taken to digitize these maps introduced a potential source of error and therefore mismatch, in applying it to specific bird positions obtained from the bearing and distance. The approach for measuring distance also can lead to binning of data, reducing the precision of each sighting. To overcome this, a circular grid was applied to each viewshed and the mean taken for each of these habitat variables (bathymetry, current speed habitat and direction flow) across the grid cell. The circular grid was created in ArcGIS, using the distance and direction toolkit for weapons (http://solutions.arcgis.com/defense/templates/distance-direction/). The mapped mean velocity and directional flow habitats, and bathymetry values to each sighting were then calculated for the grid cell. This approach also enabled a comparison of depths within the grid cell in order to determine if the seabed was a slope or plateau. The remaining hydrodynamic parameter used within the model was time from high water (in hours).

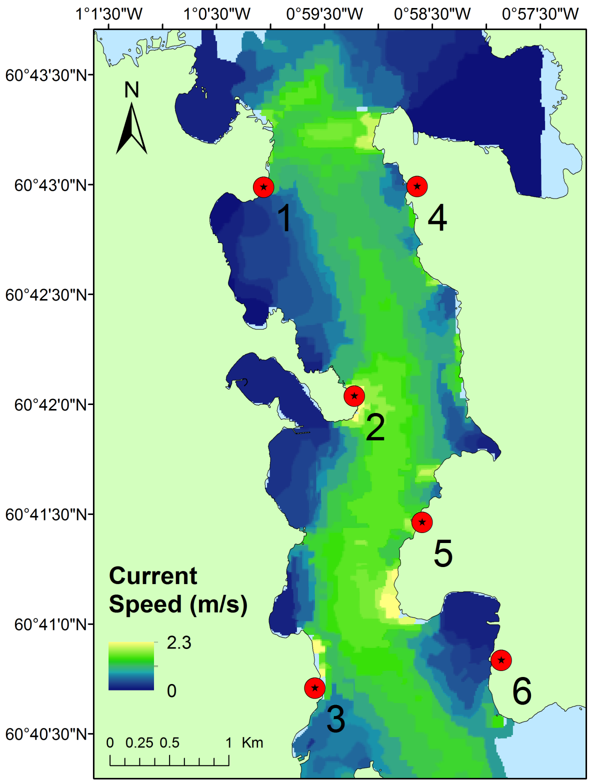

Figure 4.2 The root mean cubed tidal velocity (ms^{-1}) 'current speed habitat' base map for Bluemull Sound. This figure represents the digitsed output after geo-referencing maps from Halliday (2011).

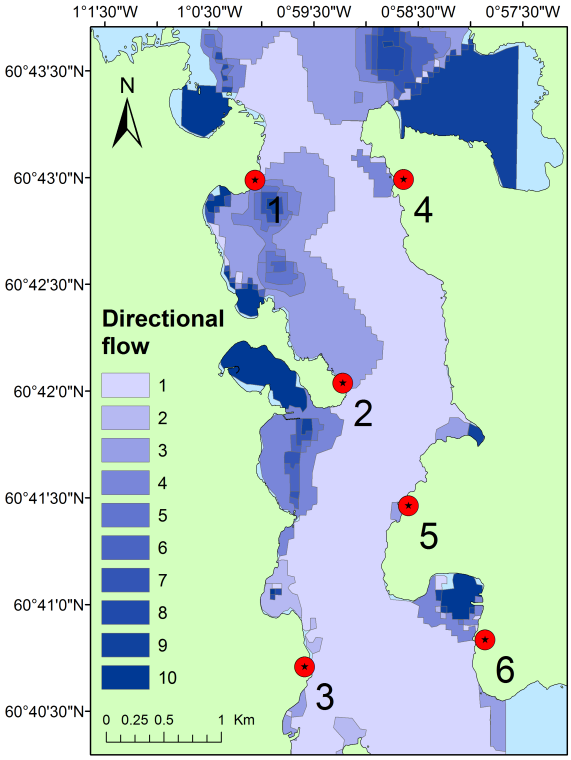

Figure 4.3 The directional flow-type base map for Bluemull Sound. This figure represents the digitised output after geo-referencing maps from Halliday (2011).

4.2.4 Observer effort and tidal currents

Observer effort is generally included within models as scan duration, allowing the model to account for scans where more effort (or time) has been spent looking for birds. However, during data collection I observed most loafing birds were passively carried by the tidal current, akin to them sitting on a conveyor belt, and that the area of sea surface scanned varied dependent on the stage of tide (both direction and speed) and duration. For example, a fast flowing current would bring more birds into the geographical area I was scanning, so the area of sea surface I scanned would be greater compared with a slower stage of the tide; if a larger surface was being scanned this typically took longer as there would be more birds to record. This was further compounded by the changing direction of the current. Thus the ability of the observer to detect birds on the sea surface is influenced by variations in tidal current velocity and direction. As mentioned, initially I decided to start each scan from the same start point and to sweep the site in the same direction, regardless of current speed and direction, to ensure consistency and repeatability of scans. Therefore a correction factor was required to account for variations in scan duration and the effective scan area of the sea surface, rather than the fixed geographic viewshed. I am unaware of such a correction being used prior to undertaking this data collection and initial analyses, subsequently Waggitt et al. (2016b) have used a similar tidal flow correction.

I calculated a tidal flow correction for the effective scan area (*ESA*), which is defined by the following equations:

$$VSW \times Scan \times \left| SSpd + CSpd \right| \tag{1}$$

$$VSW \times Scan \times \left| SSpd - CSpd \right| \tag{2}$$

$$VSW \times Scan \times \left| -SSpd + CSpd \right| \tag{3}$$

The effective scan area (*ESA*) was a function of the mean width of the viewshed (*VWS*), the scan duration (*Scan*), and absolute value of the scan velocity (*SSpd*) plus the tidal current speed (*CSpd*). Three main scenarios exist when scanning from left to right during all surveys; eqn 1) scanning against the tidal current; eqn 2) scanning with the tidal current when the tidal current is slower than the scan speed; and eqn 3) scanning with the tidal current when the tidal current is moving faster than scan speed. Scan velocity was calculated by taking the mean length (length N to S) of the midpoint of the viewshed (km) and dividing by total scan duration (sec). *ESA* and *Area* were all calculated in km^2, *VSW* was measured in km, *CSpd* was converted into kms^{-1} and *Scan* was converted into seconds. *ESA* was calculated for each observation, enabling the viewshed area to be adjusted for each observation where glare prevented the entire area being scanned. The area not visible due to glare was calculated for each scan and then subtracted from the viewshed area. The scan duration was also adjusted to ensure that any pauses were accounted for. In some circumstances the glare area was greater than the area scanned when the tidal current speed and scan speed were at similar velocites and therefore correcting for glare gave a negative area; these were removed. Also when scanning with the tidal current at a similar velocity, the ESA could be so small that the density values became outliers. For example, a total area of 0.052km^2 with 43 birds observed gave an estimated density of 827 birds/km^2. Therefore an aributary decision was made that where the ESA is less than 1/8 of the original area scanned these values were removed. This only occurred under this particular scenario and this corresponded to approximately the 95th percentile of the calculated densities, resulting in the removal of 13 outliers. The log of *ESA* was included in the models as an offset.

4.2.5 Data Analysis

Initially data for all bird species were explored by season, vantage point and behaviour using descriptive statistics. The ESA tidal current correction was estimated and this was explored using minimum, mean and maximum current speed for each observation. The likely influence of the correction was then examined by comparing the corrected and uncorrected densities, and the ESA by time from high water.

For the five key diving species observed, black guillemot, European shag, common guillemot, puffin and northern gannet generalised additive mixed effect models (GAMMs) with quasipoisson distributions were used to model relationships between density of birds and temporal and spatial conditions. Quasipoisson distributions were used as they were found to explain more of the deviance in the model than negative bionomial distributions. The abundance of birds in each scan was the response variable, with month, time of day (hour) and time from high water (hour) considered as temporal explanatory variables. Minimum current speed (ms^{-1}), bathymetry (m), current speed habitat (ordinal classes, see 4.3.2.3), direction flow (ordinal classes, see 4.3.2.3), sea state (Beaufort scale) and seabed profile (plateau vs. slope) were all considered as environment (and spatial) explanatory variables. Month, time of day, time from high water, min. current speed, and bathymetry were all modeled as non-linear continuous variables fitted as smooth terms. The number of knots was set to five, and time from high water was fitted as a cyclic cubic regression spline, to reflect the cyclic nature of the tide. Vantage point and Scan ID were converted to factors and included as random effects. GAMM's were performed in R (version 3.1.3, R Development Core Team, 2015) using 'mgcv' package (Wood, 2006).

Model selection was first undertaken using data dredging, while this was without random effects and used a poisson distribution it gave an indication of the most parsimonious model, using AIC. Backwards model selection was then performed and only statistically significant variables (p <0.05) were retained in the final model (Zuur et al. 2009). Plots of residuals for random effects showed normal distributions. Model validation was undertaken using cross validation techniques, with R values reported, however the random effects could not be included within the cross validation. Therefore, it is difficult to interpret the predictive power of the models.

4.3 Results

There were 269 scans undertaken along the length of Bluemull Sound in 2011-2012 (91 scans were undertaken during the summer 2011 and 178 scans undertaken in 2012; 20 scans undertaken in winter (January, February), 56 scans during spring (March, April, May), and 102 during the summer (June, July, August)). A total of 6368 birds recorded in these scans; with 21 species identified. Overall black guillemot was the most frequently observed species (34.2% of sightings) followed by Northern fulmar (18.0% of sightings) and European shag (17.8% of sightings). Atlantic puffin, northern gannet and common guillemot all consisted of more than 1% of the birds detected (7.9%, 7.1%, 2.8%, respectively). 7.8% of the birds detected were only identified to a family or genus level (*Gavia spp*. 0.2%, *Phalacrocorax spp*. 0.02%, Auk spp. 0.2%, *Sterna spp*. 4.7% and *Larus spp*. 2.4%) with only 2 individuals being listed as 'unidentified'.

4.3.1 Species observed by season

Seasonal variation was observed in the species assemblage and numbers detected in Bluemull Sound. Figure 4.4 highlights that proportions of species observed varied both between vantage points and seasons and tables 4.4-4.9 summarise the corrected and uncorrected densities for all species by vantage point and seasons. Black guillemot was consistently the most frequently detected species in all vantage points during the spring and summer, however shag was the most frequently detected species in both vantage points 1 and 3 during the winter (50% and 53% of sightings, with black guillemot comprising 21%, 49% and 19% of winter sightings in VP1-3). Black guillemot mean corrected density was highest during spring at VP6, with 13.04 birds/km^2. Fulmar comprised a larger proportion of detections during spring, compared with summer and winter (up to 55% of sightings at the 6 VP's during spring, and between 8 and 21% during summer), and they were detected in the highest mean corrected density of regularly occuring species, 16.97 birds/km^2 during the spring at VP6. Species such as eider, little auk and Iceland gull were observed during the winter but not in the spring or summer, while red-throated diver and razorbill were both observed more during the summer months than spring; and guillemot more during the spring than summer.

4.3.2 Species and behaviours observed at study sites

Observed behaviour varied between species and, for some species, across the different vantage points. Fig. 4.5 highlights that species, such as black guillemot showed similar proportions of foraging and loafing behaviours in VP1-3 (19.0-21.8% foraging, with 65.6-72.0% loafing). Shags also showed similar proportions of foraging in VP1 and VP3 (47.0% and 39.5%), and VP4 and VP6 (31.4% and 36.6%), but a lower proportion of birds were detected foraging in VP2 and VP5 (22.3% and 19.4%). Some bird species were consistent in the behaviours observed; for example 91.2-99.0% of fulmars were observed loafing, while *Sterna spp.* were predominantly seen foraging (90.9-100.0%). Razorbills when observed on the water surface (*n*=14)*,* as opposed to commuting flocks (which were excluded), they were foraging in 100% and 76.9% of detections in VP1 and VP2. Red-throated divers were infrequently observed in the central and northerly sites: e.g. VP1 and VP2 (*n*=8, *n*=7), but were seen proportionally more frequently during scans of VP3 and VP6 (*n*=23, *n*=10), and were detected foraging 40.0% of the time in VP6. Gull spp. showed a mix of behaviours across the VPs, but it is worth noting that a number of the gull sightings coincided with fishing vessels returning.

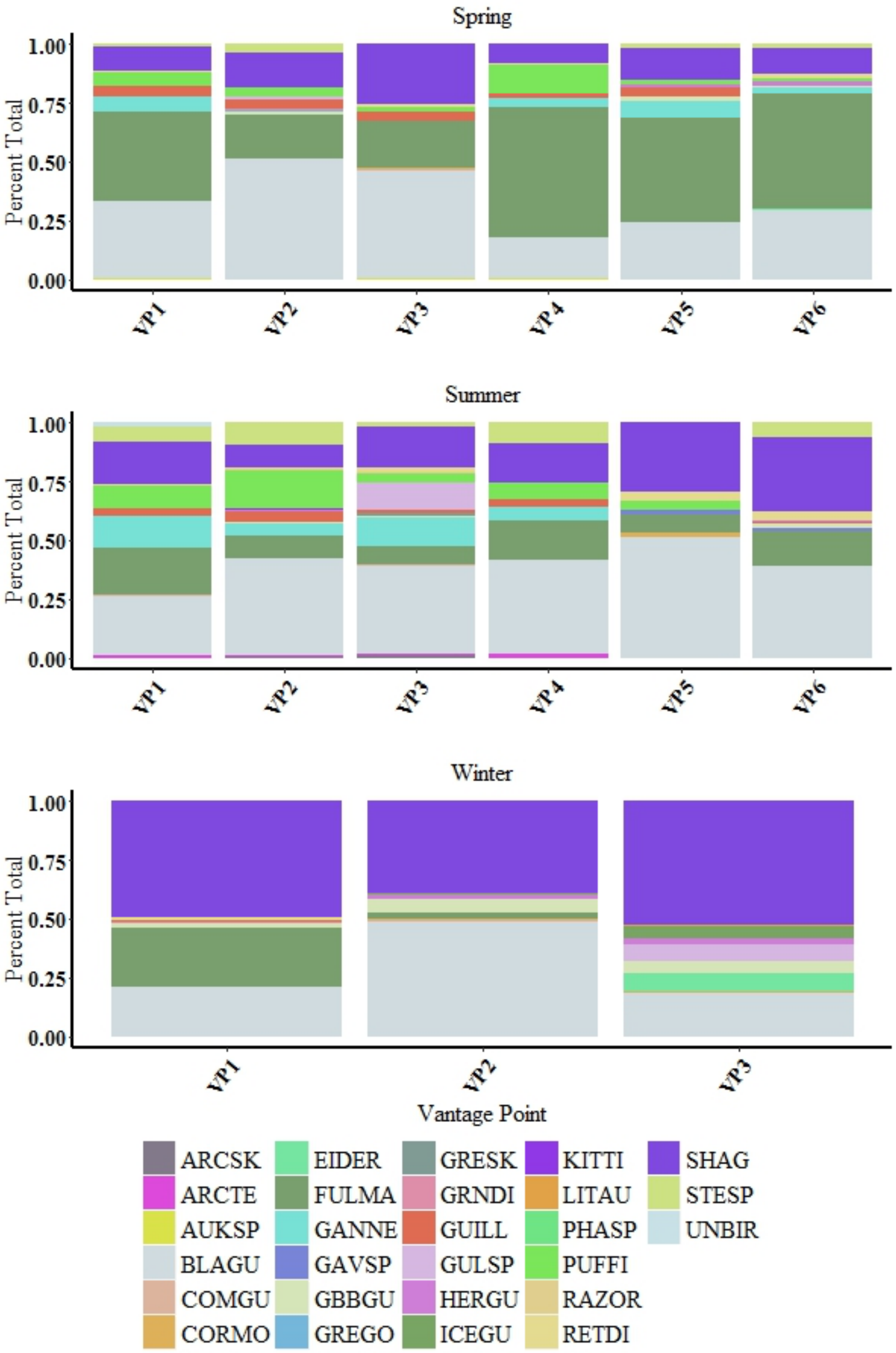

Figure 4.4 Percentage of species observed in each season by Vantage Point. Species codes used are: ARCSK, Arctic Skua; ARCTE, Arctic Tern; AUKSP, Auk *spp;* BLAGU, Black Guillemot; COMGU, Common Gull; CORMO, Great Cormorant; EIDER, Common Eider; FULMA, Northern Fulmar; GANNE, Northern Gannet; GAVSP, *Gavia spp.*; GBBGU, Great Black-Backed Gull; GREGO, Greylag Goose; GRESK, Great Skua; GRNDI, Great Northern Diver; GUILL, Common Guillemot; GULSP, *Larus spp.* HERGU, Herring Gull; ICEGU, Iceland Gull; KITTI, Black-Legged Kittiwake; LITAU, Little Auk; PHASP, *Phalacrocorax spp.*; PUFFI, Atlantic Puffin; RAZOR, Razorbill; RETDI, Red-Throated Diver; SHAG, European Shag; STESP, *Sterna spp.* UNBIR, Unidentified Bird

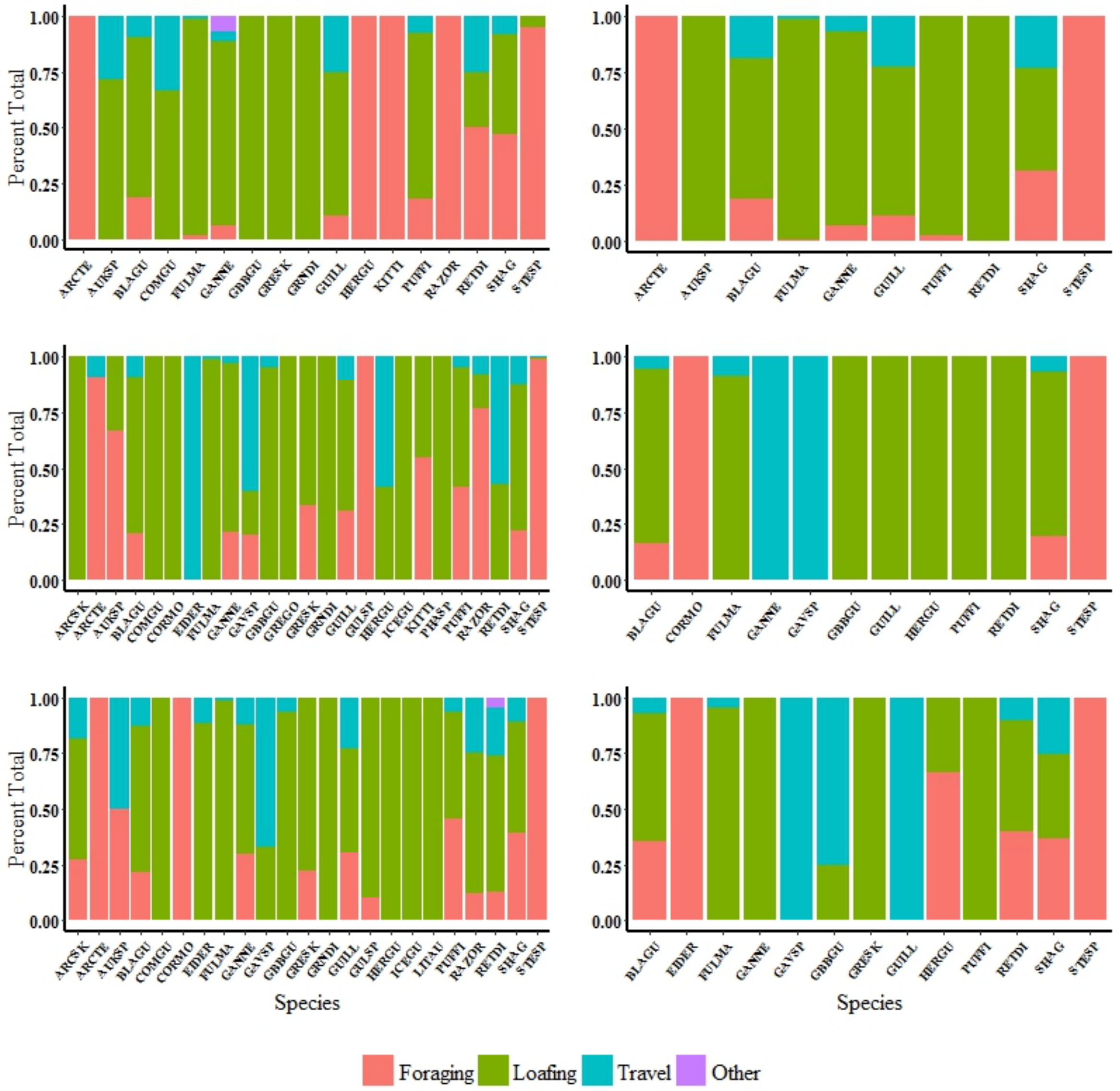

Figure 4.5 Proportion of behaviours observed for each species by Vantage Point, where pink denotes foraging, green loafing, blue travel and purple other. VP 1-3 from top left to bottom left, VP4-6 from top right to bottom right. Species codes used are: ARCSK, Arctic Skua; ARCTE, Arctic Tern; AUKSP, Auk *spp;* BLAGU, Black Guillemot; COMGU, Common Gull; CORMO, Great Cormorant; EIDER, Common Eider; FULMA, Northern Fulmar; GANNE, Northern Gannet; GAVSP, *Gavia spp.*; GBBGU, Great Black-Backed Gull; GREGO, Greylag Goose; GRESK, Great Skua; GRNDI, Great Northern Diver; GUILL, Common Guillemot; GULSP, *Larus spp.* HERGU, Herring Gull; ICEGU, Iceland Gull; KITTI, Black-Legged Kittiwake; LITAU, Little Auk; PHASP, *Phalacrocorax spp.*; PUFFI, Atlantic Puffin; RAZOR, Razorbill; RETDI, Red-Throated Diver; SHAG, European Shag; STESP, *Sterna spp*.

4.3.4 Tidal flow correction factor

The correction factor was calculated to account for changes in tidal flow and subsequently observer effort varied by time from high tide. Figure 4.6 compares uncorrected densities for all birds with the corrected density (using the *ESA*), showing the correction reduces some variation in the densities predicted. This is particularly notable at 2 hours before and after low water. The variation in *ESA* by time from high tide shows a cyclic pattern with increases and decreases reflecting changes in current speed, i.e. where faster current speeds occurred a larger area was scanned, and a bigger correction applied. Tables 4.4-4.9 compare the mean uncorrected and corrected density by season, per species for each vantage point. The magnitude of uncorrected densities that were higher than corrected densities was larger, which reflected more observations involved scanning against the current. When scanning against the current there was an inverse relationship, with faster water resulting in a bigger correction and typically a smaller density. Black guillemot and puffin were two of the diving species that were found to more frequently have a larger correction, along with gull and tern species. The largest corrections were typically proportionate to the largest observed densities, for example; black guillemot corrected density was -4.45 birds/km^2 than the uncorrected density at VP4 during the spring and 7.69 birds/km^2 more than the uncorrected density at VP6 in the spring; and fulmar corrected density was -16.08 birds/km^2 than the uncorrected density at VP4 during the spring.

Table 4.4 Mean uncorrected density of birds/km^2 and corrected density of birds/km^2 (using ESA) per species for Vantage Point 1, Bluemull Sound, Shetland by season

Species	**Spring**		**Summer**		**Winter**	
	Uncorrected density birds/km^2	Corrected density birds/km^2	Uncorrected density birds/km^2	Corrected density birds/km^2	Uncorrected density birds/km^2	Corrected density birds/km^2
Greylag Goose	-	-	-	-	-	-
Common Eider	-	-	-	-	-	-
Red-throated Diver	0.90	2.35	1.13	0.53	0.45	0.31
Great Northern Diver	-	-	-	-	0.45	0.23
Gavia spp.	-	-	-	-	-	-
Northern Fulmar	10.60	9.31	3.83	5.94	-	-
Northern Gannet	4.06	1.99	2.83	2.07	-	-
Great Cormorant	-	-	-	-	-	-
European Shag	3.52	1.69	3.36	4.34	2.71	2.05
Phalacrocorax spp.	-	-	-	-	-	-
Arctic Skua	-	-	-	-	-	-
Great Skua	-	-	0.45	0.08	-	-
Atlantic Puffin	2.08	0.97	1.78	3.13	-	-
Black Guillemot	9.55	7.03	4.31	6.01	1.92	1.69
Razorbill	-	-	0.90	0.32	-	-
Common Guillemot	1.92	0.92	0.94	1.37	-	-
Little Auk	-	-	-	-	-	-
Auk spp.	0.45	0.20	0.61	0.38	-	-
Arctic Tern	-	-	1.10	1.47	-	-
Sterna spp.	0.68	0.21	5.48	17.17	-	-
Black-legged Kittiwake	-	-	0.90	3.40	-	-
Common Gull	-	-	1.13	0.50	-	-
Herring Gull	0.45	1.18	0.45	0.22	-	-
Iceland Gull	-	-	-	-	-	-
Great black-backed Gull	0.45	0.27	0.60	0.64	-	-
Larus spp.	-	-	-	-	-	-

Table 4.5 Mean uncorrected density of birds/km^2 and corrected density of birds/km^2 (using ESA) per species for Vantage Point 2, Bluemull Sound, Shetland by season.

Species	**Spring**		**Summer**		**Winter**	
	Uncorrected density birds/km^2	Corrected density birds/km^2	Uncorrected density birds/km^2	Corrected density birds/km^2	Uncorrected density birds/km^2	Corrected density birds/km^2
Greylag Goose	1.58	2.55	-	-	-	-
Common Eider	-	-	3.16	6.01	-	-
Red-throated Diver	-	-	0.99	2.80	-	-
Great Northern Diver	0.79	0.31	0.79	0.24	-	-
Gavia spp.	-	-	2.19	0.79	-	-
Northern Fulmar	4.90	3.41	3.24	3.46	3.16	0.73
Northern Gannet	0.79	0.22	2.63	4.58	-	-
Great Cormorant	-	-	0.79	2.59	0.79	0.56
European Shag	4.32	2.55	2.60	2.70	9.09	9.36
Phalacrocorax spp.	-	-	0.79	0.44	-	-
Arctic Skua	-	-	0.79	2.73	-	-
Great Skua	-	-	1.19	0.37	-	-
Atlantic Puffin	1.63	1.24	5.04	4.60	-	-
Black Guillemot	13.16	9.63	9.30	10.70	14.38	11.99
Razorbill	-	-	1.75	0.65	-	-
Common Guillemot	1.60	1.39	2.50	3.24	-	-
Little Auk	-	-	-	-	-	-
Auk spp.	-	-	0.83	0.24	-	-
Arctic Tern	-	-	1.47	2.83	-	-
Sterna spp.	2.63	1.11	6.07	4.21	-	-
Black-legged Kittiwake	-	-	2.90	0.74	-	-
Common Gull	-	-	0.79	1.74	-	-
Herring Gull	-	-	3.56	1.04	2.37	2.81
Iceland Gull	-	-	-	-	1.58	1.87
Great black-backed Gull	2.37	1.08	0.79	0.80	4.74	5.26
Larus spp.	2.37	0.84	-	-	-	-

Table 4.6 Mean uncorrected density of birds/km^2 and corrected density of birds/km^2 (using ESA) per species for Vantage Point 3, Bluemull Sound, Shetland by season.

Species	**Spring**		**Summer**		**Winter**	
	Uncorrected density birds/km^2	Corrected density birds/km^2	Uncorrected density birds/km^2	Corrected density birds/km^2	Uncorrected density birds/km^2	Corrected density birds/km^2
Greylag Goose	-	-	-	-	-	-
Common Eider	-	-	-	-	2.78	1.07
Red-throated Diver	0.98	1.24	1.09	0.54	-	-
Great Northern Diver	-	-	0.66	0.18	-	-
Gavia spp.	-	-	0.98	0.28	-	-
Northern Fulmar	3.38	4.59	2.17	3.09	-	-
Northern Gannet	-	-	2.83	3.47	0.66	3.52
Great Cormorant	0.66	1.07	-	-	0.66	0.34
European Shag	4.22	4.08	2.62	3.84	11.33	13.38
Phalacrocorax spp.	-	-	-	-	-	-
Arctic Skua	-	-	0.80	1.12	-	-
Great Skua	-	-	0.98	1.50	-	-
Atlantic Puffin	0.99	0.87	1.28	3.43	-	-
Black Guillemot	7.61	8.90	5.31	7.17	4.02	3.64
Razorbill	-	-	1.31	1.64	-	-
Common Guillemot	2.32	1.86	1.24	2.38	-	-
Little Auk	-	-	-	-	0.66	0.97
Auk spp.	0.66	0.37	0.66	0.27	-	-
Arctic Tern	-	-	0.82	0.36	-	-
Sterna spp.	-	-	1.97	7.82	-	-
Black-legged Kittiwake	-	-	-	-	-	-
Common Gull	0.66	0.33	0.66	0.30	-	-
Herring Gull	-	-	0.66	0.61	2.29	10.64
Iceland Gull	-	-	-	-	3.93	19.50
Great black-backed Gull	-	-	0.66	1.07	1.44	4.26
Larus spp.	-	-	10.81	32.67	10.48	5.50

Table 4.7 Mean uncorrected density of birds/km^2 and corrected density of birds/km^2 (using ESA) per species for Vantage Point 4, Bluemull Sound, Shetland by season.

Species	**Spring**		**Summer**	
	Uncorrected density birds/km^2	Corrected density birds/km^2	Uncorrected density birds/km^2	Corrected density birds/km^2
Greylag Goose	-	-	-	-
Common Eider	-	-	-	-
Red-throated Diver	1.39	0.49	-	-
Great Northern Diver	-	-	-	-
Gavia spp.	-	-	-	-
Northern Fulmar	29.42	13.34	1.78	4.63
Northern Gannet	6.96	7.08	0.70	0.15
Great Cormorant	-	-	-	-
European Shag	4.18	2.81	1.03	1.49
Phalacrocorax spp.	-	-	-	-
Arctic Skua	-	-	-	-
Great Skua	-	-	-	-
Atlantic Puffin	6.62	2.56	0.78	2.09
Black Guillemot	9.40	4.95	4.13	4.07
Razorbill	-	-	-	-
Common Guillemot	1.62	0.87	0.86	0.15
Little Auk	-	-	-	-
Auk spp.	0.70	0.24	-	-
Arctic Tern	-	-	0.70	0.53
Sterna spp.	-	-	2.25	0.45
Black-legged Kittiwake	-	-	-	-
Common Gull	-	-	-	-
Herring Gull	-	-	-	-
Iceland Gull	-	-	-	-
Great black-backed Gull	-	-	-	-
Larus spp.	-	-	-	-

Table 4.8 Mean uncorrected density of birds/km^2 and corrected density of birds/km^2 (using ESA) per species for Vantage Point 5, Bluemull Sound, Shetland by season.

Species	**Spring**		**Summer**	
	Uncorrected density birds/km^2	Corrected density birds/km^2	Uncorrected density birds/km^2	Corrected density birds/km^2
Greylag Goose	-	-	-	-
Common Eider	-	-	-	-
Red-throated Diver	-	-	1.75	3.12
Great Northern Diver	-	-	-	-
Gavia spp.	-	-	0.87	0.57
Northern Fulmar	15.44	6.98	1.75	2.29
Northern Gannet	7.87	1.85	-	-
Great Cormorant	-	-	0.87	0.57
European Shag	4.66	2.20	5.84	3.76
Phalacrocorax spp.	-	-	-	-
Arctic Skua	-	-	-	-
Great Skua	-	-	-	-
Atlantic Puffin	1.31	0.47	2.55	1.04
Black Guillemot	8.45	4.06	8.64	10.15
Razorbill	-	-	-	-
Common Guillemot	2.18	0.72	-	-
Little Auk	-	-	-	-
Auk spp.	-	-	-	-
Arctic Tern	-	-	-	-
Sterna spp.	1.75	1.70	-	-
Black-legged Kittiwake	-	-	-	-
Common Gull	-	-	-	-
Herring Gull	0.87	0.21	-	-
Iceland Gull	-	-	-	-
Great black-backed Gull	1.75	1.70	-	-
Larus spp.	-	-	-	-

Table 4.9 Mean uncorrected density of birds/km^2 and corrected density of birds/km^2 (using ESA) per species for Vantage Point 6, Bluemull Sound, Shetland by season.

Species	**Spring**		**Summer**	
	Uncorrected density birds/km^2	Corrected density birds/km^2	Uncorrected density birds/km^2	Corrected density birds/km^2
Greylag Goose	-	-	-	-
Common Eider	0.63	3.90	-	-
Red-throated Diver	0.63	1.92	1.26	2.74
Great Northern Diver	-	-	-	-
Gavia spp.	-	-	1.26	0.57
Northern Fulmar	13.86	16.97	3.08	1.70
Northern Gannet	2.52	0.93	-	-
Great Cormorant	-	-	-	-
European Shag	1.73	3.83	6.87	5.61
Phalacrocorax spp.	-	-	-	-
Arctic Skua	-	-	-	-
Great Skua	-	-	0.63	0.28
Atlantic Puffin	1.26	0.46	-	-
Black Guillemot	5.35	13.04	7.49	6.26
Razorbill	-	-	-	-
Common Guillemot	-	-	0.63	2.32
Little Auk	-	-	-	-
Auk spp.	-	-	-	-
Arctic Tern	-	-	-	-
Sterna spp.	0.63	1.72	6.93	2.54
Black-legged Kittiwake	-	-	-	-
Common Gull	-	-	-	-
Herring Gull	1.26	0.46	0.63	2.32
Iceland Gull	-	-	-	-
Great black-backed Gull	0.63	1.82	1.26	4.64
Larus spp.	-	-	-	-

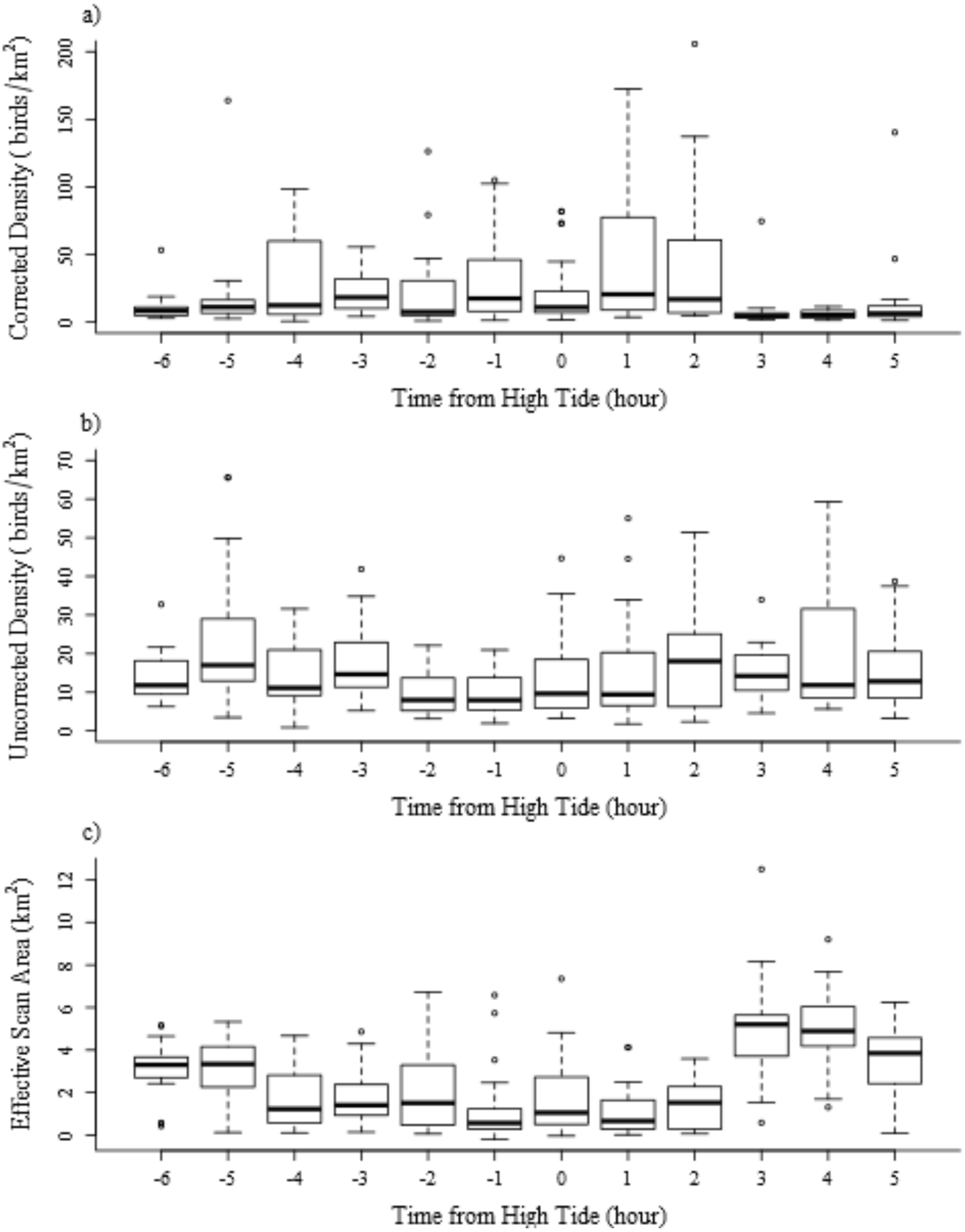

Figure 4.6 Comparison of the influence of the tidal correction factor, *ESA,* across the ebb-flood tidal cycle in Bluemull Sound, Shetland, where -6 is low tide and 0 is high tide: a) density of all birds using the corrected effective scan area (*ESA*); b) uncorrected density of all birds using the viewshed area; c) the effective scan area.

4.3.5 Exploring temporal and spatial associations

Species are considered below in order of decreasing local abundance.

4.3.5.1 Black Guillemot

The density of black guillemots in Bluemull Sound showed a significant negative relationship with current speed; with decreasing numbers at increasing speeds (*p*=<0.01) (Fig. 4.7). Black guillemot did not show any significant relationships with spatial or other environmental variables. Overall the model fit was good (Deviance explained=79%; GCV=0.36403; Scale est.= 0.3871; *n*=1526), however predictive ability of the model (not including random effects) was poor (*R*= 0.117).

4.3.5.2 European Shag

The density of European shags in Bluemull Sound showed significant temporal variation and association to habitat. Significant relationships were found seasonally (*p*=<0.01), with increases in numbers through the breeding season; and a significant negative relationship with current speed (*p*=<0.01) (Fig. 4.8). The preferred model highlighted that significantly more shags were found in the areas of slower current speeds (*p*=<0.01) and plateaued seabed (*p*=0.01). The preferred model also found significantly less loafing and travelling shags compared with foraging shags during scans (*p*=<0.01, *p*=<0.01). Overall the model fit was good (Deviance explained=69.8%; GCV=1.0136; Scale est.= 0.83336; *n*=665), however predictive ability of the model (not including random effects) was poor (*R*= 0.259).

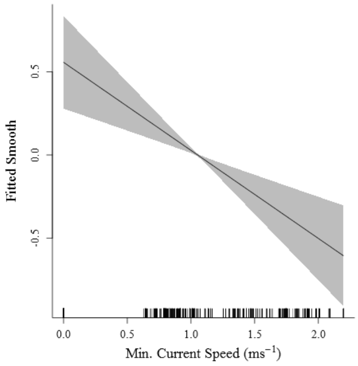

Figure 4.7 Fitted smooth curve (±SE) from GAMMs showing the predicted value of numbers of black guillemot in Bluemull Sound, Shetland as a function of minimum current speed during the observation.

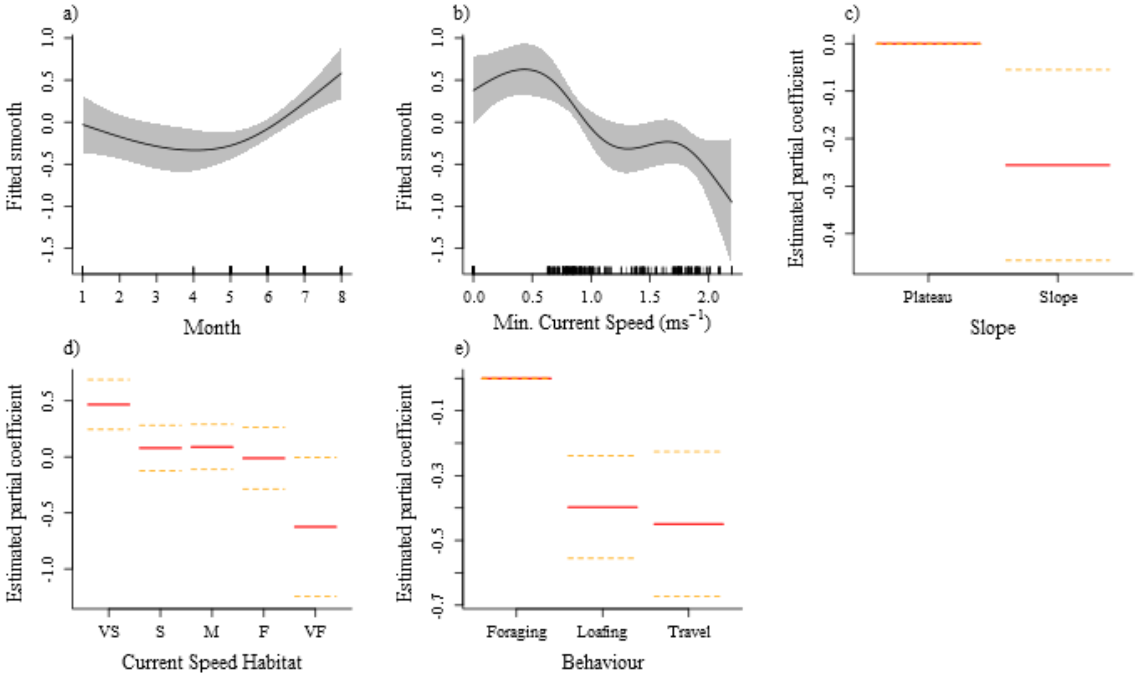

Figure 4.8 Fitted smooth curve (±SE) from GAMMs showing the predicted value of numbers of European shag in Bluemull Sound, Shetland as a function of a) month; b) minimum current speed. Term plots (±SE) from GAMMs showing the predicted value of numbers of European shag in Bluemull Sound, Shetland as a function of parametric terms c) seabed profile; d) current speed habitat; and e) behaviour.

4.3.5.3 Atlantic Puffin

Atlantic puffin showed a significant temporal pattern as well as significant habitat associations in Bluemull Sound (fig 4.9). The preferred model showed a significantly positive relationship with month. It also found puffin to have a decreasing relationship with tidal current speed, however, unlike other species, the spline indicates numbers decline until current speeds of ~1.5 ms^{-1}, after which there is a slight increase in numbers again (p=<0.01). Similarly, puffins were detected significantly less in the fast habitat class (2.0-2.5ms^{-1}) (p=<0.01) but higher numbers were then detected in the very fast class (>2.5ms^{-1}). Overall the model fit was good (Deviance explained=97.9%; GCV=0.77833; Scale est.=0.54194; n=302), however predictive ability of the model (not including random effects) was poor (R= 0.172).

4.3.5.4 Northern Gannet

Northern gannet density in Bluemull Sound showed a significant relationship with tidal current speed and other habitat associations. The preferred model showed significant relationships with bathymetry, directional flow and behaviour (Fig. 4.10). Gannet numbers detected were significantly lower in areas of more linear current flow (p=<0.01), and significantly more gannets were detected loafing (p=<0.01) compared with foraging. Gannet was the only species modeled to include the ‘other’ behaviour category, which was due to birds in spring seen diving for kelp for nesting material. Overall the model fit was moderate (Deviance explained=68.4%; GCV=1.4453; Scale est.= 1.0325; n=226), however predictive ability of the model (not including random effects) was poor (R= 0.214).

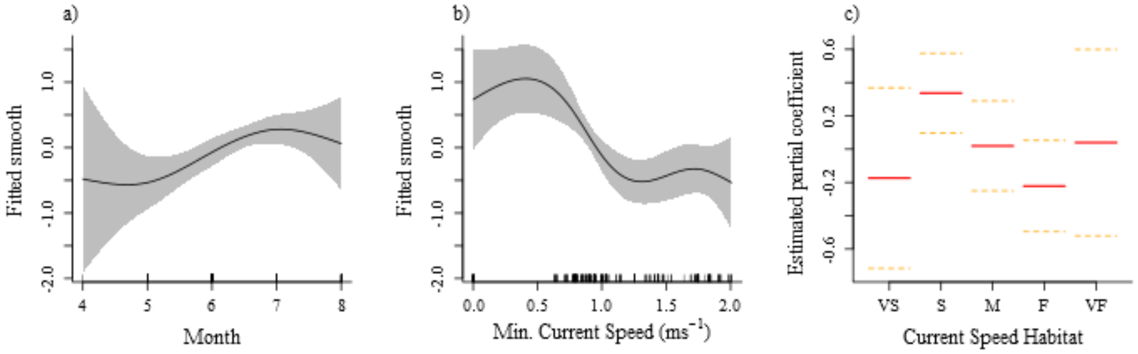

Figure 4.9 Fitted smooth curve (±SE) from GAMMs showing the predicted value of numbers of Atlantic puffin in Bluemull Sound, Shetland as a function of a) month; b) minimum current speed (ms^{-1}). Term plots (±SE) from GAMMs showing the predicted value of numbers of Atlantic puffin in Bluemull Sound, Shetland as a function of parametric terms c) current speed habitat.

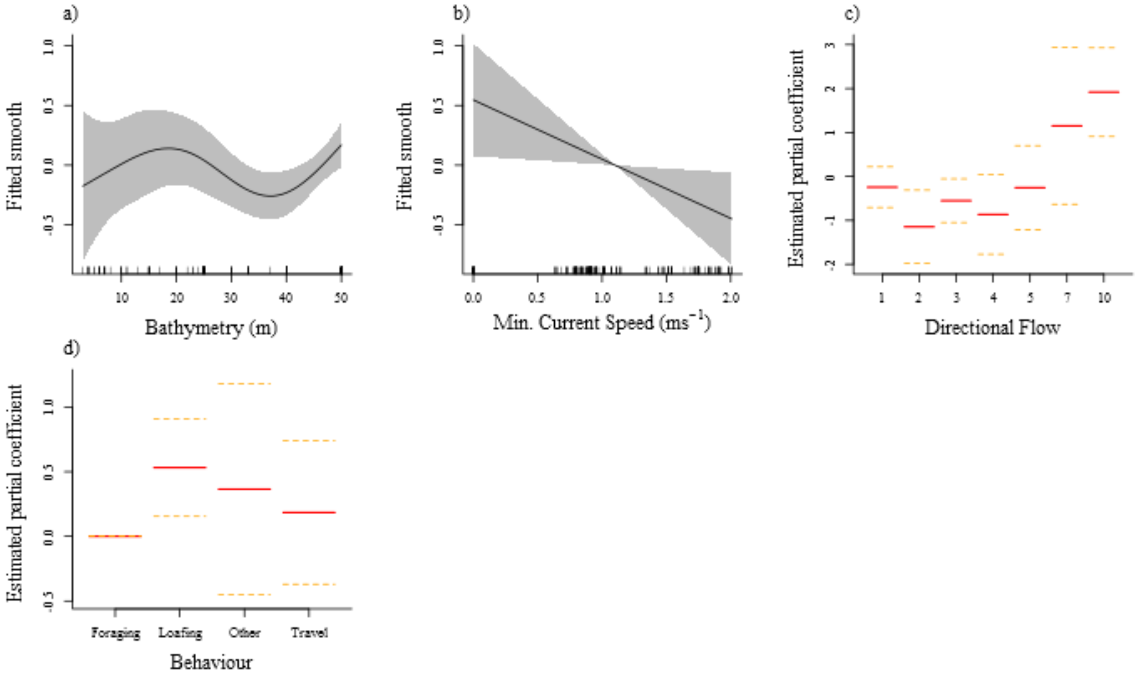

Figure 4.10 Fitted smooth curve (±SE) from GAMMs showing the predicted value of numbers of northern gannet in Bluemull Sound, Shetland as a function of a) bathymetry; b) minimum current speed (ms^{-1}). Term plots (±SE) from GAMMs showing the predicted value of numbers of northern gannet in Bluemull Sound, Shetland as a function of parametric terms c) directional flow; and d) behaviour.

4.3.5.5 Common Guillemot

The preferred model for common guillemot in Bluemull Sound showed significant relationships with temporal and environmental variables. Figure 4.11 shows a significant positive relationship with month (*p*=<0.01), and significant negative relationship was found with time of day (*p*=0.01) and minimum current speed (*p*=<0.01). Guillemot were also observed foraging significantly more than loafing or travelling (*p*=0.03 and *p*=0.03). Overall the model fit was moderate (Deviance explained=82.4%; GCV=0.61426; Scale est.=0.39858; *n*=138), however predictive ability of the model (not including random effects) was poor (*R*=-0.175).

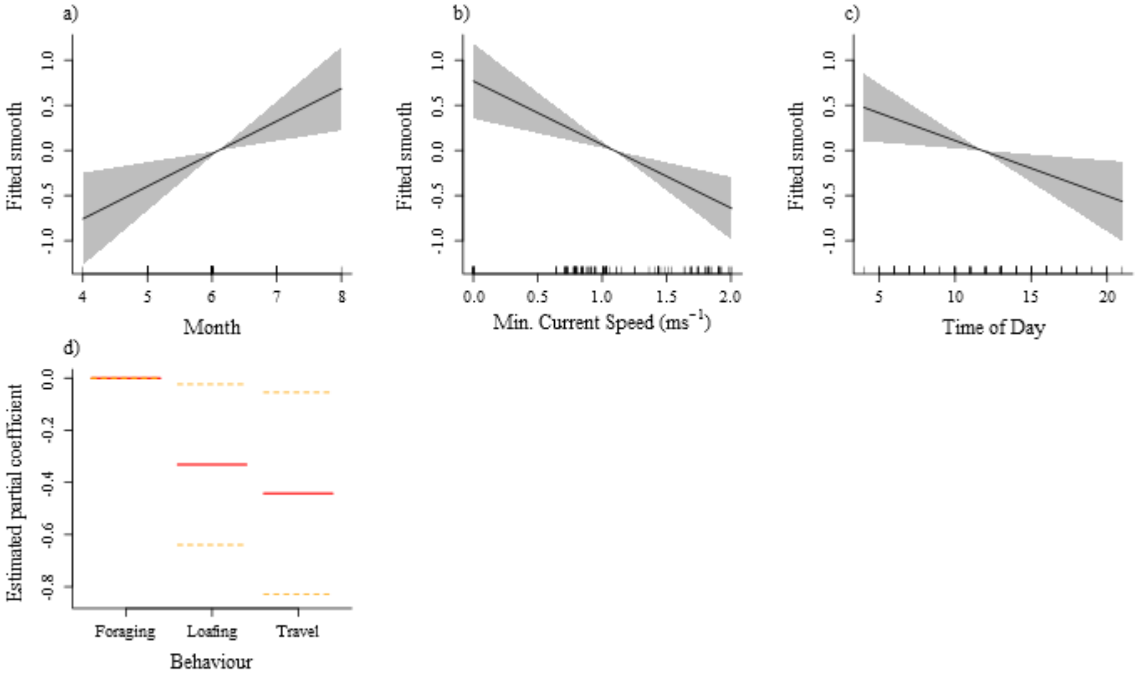

Figure 4.11 Fitted smooth curve (±SE) from GAMMs showing the predicted value of numbers of common guillemot in Bluemull Sound, Shetland as a function of a) month; b) minimum current speed (ms^{-1}); c) time of day. Term plots (±SE) from GAMMs showing the predicted value of numbers of common guillemot in Bluemull Sound, Shetland as a function of parametric terms d) behaviour.

4.4 Discussion

This chapter has explored how marine birds used a high-energy tidal stream in Shetland, suitable for marine renewable energy deployment. The work has combined field observations and modelling to identify methodologies issues, and some solutions for guiding developments. Few studies in UK high-energy tidal streams have been published to date that have both the quantity of combined frequency and duration of observations undertaken here. Results indicate that five key diving species show variation in spatial and temporal usage of Bluemull Sound, and different habitat associations were found between different species, including more similar species, such as auks. I also propose a correction for effective scan area (*ESA*) that accounts for both fluctuations in tidal current speed and observer effort.

4.4.1 Temporal patterns and habitat associations

The marine bird assemblage using Bluemull Sound varied between breeding and non-breeding seasons, reflecting typical migratory and post-breeding dispersal patterns for marine birds in Shetland (Mitchell et al. 2004, Pennington et al. 2004, Forrester et al. 2007). Black guillemot and European shag were the most frequently detected species throughout the year, reflecting their more sedentary strategies (Forrester et al. 2007), indicating a higher vulnerability to marine renewable developments due to continued exposure throughout the year (Furness et al. 2012).

Some species were found to have temporal patterns in their usage of the sound, although a significant diurnal pattern was only found in guillemot numbers, which decreased in throughout the day. Shag, puffin and guillemot also showed a significant increase in numbers over the months observed, which is likely due to the presence of young after the breeding season. These younger, less experienced birds need to spend time trying to hone their foraging skills, which may mean they are more vulnerable to collision (Galbraith et al. 1981; Furness et al. 2012).

All species significantly decreased in numbers as a function of increasing in current speed. Diving birds face a trade-off between energy gain due to prey consumption against energy lost due to the energetics of diving (Wilson et al. 2001; Heath and Gilchrist 2010; Rey at al. 2010). The findings from this study suggest species were avoiding foraging during periods of high current velocities. However, the contrasting findings from other studies suggest that other factors make this a little more complicated. For example, Holm and Burger (2002) found different species exploited different current velocities; the planktivorous ancient murrelet *Synthliboramphus antiquus* and piscivorous pigeon guillemot *Cepphus columba* had a higher abundance during the tidal peak flow; while most of the other piscivorous species observed, preferred slack water and moderate currents, at a wide range of depths. The findings for shag are similar to those in chapter 3 and Waggitt et al. (2016a), who suggested that low current speeds and water elevation may enable them to exploit more sessile benthic prey, with this association being driven by diving costs and prey capture. Waggitt et al. (2016a) also found greater numbers of common guillemot and Atlantic puffin with increasing current velocities; conversely Wade (2015) found fewer shag, auk, divers and common eider in areas with current speeds >2ms^{-1}. A slightly more complex relationship was seen with puffin and current velocity in Bluemull Sound: numbers declined until current speeds of ~1.5 ms^{-1}, after which there is a slight increase in numbers again. This concurs slightly more with the findings of Waggitt et al. (2016a) and could possibly due to the different foraging strategies employed by puffin compared to other auks. During observations at peak flows puffin in Bluemull Sound were seen undertaking very short, and presumably shallow dives in around a turbulent, temporary shearline feature. Martin and Wanless (2015) observed that puffin visual fields were very different to those of common guillemot, and as they carry a high prey load, it may be more energetically beneficial for them to have access to prey that are concentrated by turbulent waters.

Marine birds within Bluemull Sound showed a variety of habitat associations, including current speed, directional flow and seabed profile; however more species were found to have significant habitat associations with current speed habitat. Shag, puffin and common guillemot were all found to have different preferences for areas of slower or faster flowing water. While these areas may not be continually flowing at those speeds (due to tidal cycle) the association was found with the mean speed. Shag were found in greater numbers of slower speeds (0.5-1ms^{-1}) and less in areas of high speeds (>2.5ms^{-1}). This concurs with the findings of Holm and Burger (2002) for the related pelagic cormorants *Phalacrocorax pelagicus*, who they observed avoiding areas of high current and turbulence. This would also indicate that shags are less likely to occur in areas that are optimal for tidal turbines (Fraenkal 2006). Puffin were found in higher numbers in the slow areas (0.5-1ms^{-1}) compared with fast, but higher number were then detected in the very fast areas (>2.5ms^{-1}), corresponding with my observations of puffin foraging in turbulent waters near the shearline off of Culli Ness. The association with faster flows are similar to that observed by Waggitt et al. (2016a) who also observed more guillemot and puffin in fast flowing areas. This indicates that they may be at higher risk of spatial overlap with tidal turbine placement. While this may initially be at odds with the correlation found with current speed it may suggest these fast flow areas provide a temporally and spatially localised resource, which does not occur at maximum tidal current flow. The differences in preferences between species within the sound may reflect different foraging strategies, such as benthic vs pelagic feeders, and foot vs wing-propelled divers, as well as their diving abilities (e.g. Gaston, 2004; Riback et al. 2004; Watanuki et al. 2005), but could also show fine-scale segregation of resources.

4.4.2 Methodological limitations and challenges

High-energy tidal streams present a challenging environment for scientific research and in particular collection of empirical data. There are considerable constraints on making observations of birds, and the deployment of monitoring devices is costly. My fieldwork was one of the first, non-developer lead, shore-based studies of marine birds to be undertaken in Scotland. This meant there were many methodological challenges. During data collection, I found a number of issues arose that could not reasonably be foreseen. This resulted in the need to develop either solutions in the field or to try and collect data consistently to enable retrospective corrections to be applied. One such example is scanning a mobile water surface, which loafing birds are passively moving with. I have developed an approach to dealing with this through the use of the tidal correction factor, ESA. This enables both the incorporation of effort and differences in tidal current conditions between observations, which would otherwise result in methodological bias. Due to scanning in one direction I found that I had to incorporate a directional element to the calculation, and this added a further level of complication. Through trial and error with this calculation I found the directional element clearly has a large impact on the results, particularly the density estimates, and I feel this further justifies the need for such a correction.

The tidal correction factor was applied retrospectively, as the issue of scanning the water surface was noted during observations. Consequently the calculation had to take a basic form to account for data available (i.e. collected concurrent with observations vs. modelled after the fact) and is therefore based on a number of assumptions, which often were a simplification of reality:

1. Birds sit passively on the water surface
2. The areas scanned were rectangular
3. The tidal current is moving at an even velocity across the width of the sound
4. The scan is undertaken at an even velocity and that every bird is detected
5. The modelled tide flow regime is accurately applied

From both the scans and focal observations, I found the first assumption is largely met as it was the passive loafing of birds on the moving sea surface that first alerted me to the scanning problem. During the focal observations I did observe birds swimming against the current but given the field of view of the scope, they were not travelling sufficiently fast enough to violate this assumption. The second assumption is a simplification to enable easier calulation of the area. In reality the viewsheds were typically more trapezoid in shape but this wasn't considered to have a significant impact on the calculation. The third assumption relates to the tidal current moving evenly across the sound. Due to the minimal hydrodynamic data it is difficult to understand to what extent this has been violated. However, with most tidal stream environments, and indeed the modeling that has been undertaken at Bluemull Sound suggests heterogeneity across the sound due to bathymetry and small scale hydrodynamic features. A better understanding of how this violation affects the calculation could be achieved at a study where concurrent detailed hydrodynamic modelling was being undertaken. The fourth assumption has two parts; the first is based on the observer undertaking the scan at a continual pace. This is unlikely to be fully achieved in reality, particularly if there are greater numbers of birds present which need to be recorded. The second part requires 100% detection, which is unrealistic given the previously raised issue of declines in detectability with distance but also its possible that at the fastest current speeds some birds are missed. A way to calibrate this would be to undertake an experiment where a number of floats were released at one end of a study site and an observer (blind to the number of floats released) then scans the study site to count the floats. The final assumption relies on the modeled tidal data (from several years previous) to be accurately applied to the observation dataset, as no concurrent data were available. If this is violated it would introduce noise and potentially bias the results. The modeled data set performed well, however, I would strongly advise that in future this was carried out with concurrent tidal data and also note that for other sites which do not have conincident tide height and flow regimes this correction factor may not be applicable.

Without incorporating this correction any other studies using vantage point methodologies in HETS are breaking the fundamental density assumption of equal effort with regard to total area scanned. Therefore in future methods should look to take into account flowing water to ensure an equal area is sampled during scans or accounted for within the analysis. While this calculation does carry a number of simplistic assumptions it is strongly recommended that any future survey is designed to collect data to enable this, or similar, correction. With further consideration, it is highly likely that a more mathematical approach could be applied. I strongly recommend that scanning is always undertaken against the current's direction and that data should be collected on concurrent tidal current speeds and enable calculation of scanning velocity and area for each scan. Until then I urge caution in the interpretation of any tidal cycle trends where methods used may include this bias.

The methods used within this study already include a number of improvements, including those set out in Chapters 2 and 3, and Waggitt et al. (2014), such as the move away from grids and the use of multiple vantage points. There is more flexibility within this dataset for further analysis than was undertaken here, such as the inclusion of distance sampling (or spatially-explicit capture recapture), although the development of packages is still required to tackle the key violated assumptions. Multiple VP's may facilitate a gradient design. However, my results (notably difference in deviance explained and cross-validation (without random effects)), suggest that the random effects included within the models (Scan ID and VP) are explaining a significant amount of the deviation. Without further investigation it is difficult to establish whether this is due to a measurable variable that has not been included or variation within the system that may be difficult to capture and measure. Due to the nature of the sound it is possible there are multiple, small-scale drivers behind bird usage of the sound, not necessarily covered by the parameters included in the model. The usage between VP's varied significantly enough to indicate the importance of the random effects. Conversely, it could also be considered that the sound is a small geographic area in the context of foraging ranges for many of the species present (Thaxter et al. 2012) and therefore birds may be detected across the wider area.

This study shows that it is possible to collect informative data for an area suitable for tidal development with little budget or technology and equipment. I was able to retrospectively apply habitat and hydrodynamic data, although acknowledge it would be preferable to apply these at a finer scale resolution, and using a concurrent dataset.

Similarly, studies combining hydrodynamic and prey monitoring would support a better understanding of factors driving marine bird usage of these areas.

This study focuses on the use of all birds as opposed to observing just foraging individuals. While foraging is considered to be more important in terms of encountering marine renewables, it is important to understand other behavioural uses of a site, which may be required for assessment, particularly where a development occurs within a designated site (e.g. SPA extension) even if they have fewer consequences for collision risk. It is important to understand why and how birds are using the area, which cannot necessarily be obtained from a snapshot of an animal's behaviour in a scan. This chapter showed differences were found in behaviours observed, however, it is possible that some species behave in particular ways that could lead to a bias, such as gannets plunge diving and then resting on the water between diving bouts. Gannet are more likely to be recorded as loafing, and less likely to be observed foraging; whereas shags, due to their 'wettable' plumage, are more likely to be seen foraging, after which they fly to nearby roosts to dry.

4.4.3 Implications for marine renewable surveys

This chapter has contributed to our understanding of five key diving species usage of a high-energy tidal stream. However, the results, when compared with findings of other studies, show birds utilise HETS differently. This is a likely consequence of each HETS creating a unique mosaic of habitats and complex hydrodyamics, therefore birds may adapt their foraging approaches to maximise foraging. Consequently, my results are not necessarily applicable to other HETS. This strongly indicates the need for site-specific studies to be undertaken for assessment of developments. The decrease in detections of most species during faster current speeds suggests decreased risk of overlap with marine renewables. However, a preference for faster flowing turbulent areas by puffin may counter this.

This study shows the importance of understanding the methodological limitations of surveys in these high-energy environments. The application of a correction factor suggests significant bias in methods that don't account for changing current speeds, and that densities used without a correction may be either inflated or deflated. This could have large consequences for any underwater collision risk modeling estimates. I strongly recommend that further work is undertaken to explore the implications of this, and to improve the methods used to calculate this corrections. I also recommend switching the direction of scans to reduce the variation and simplify the correction. The next steps for this study are to use the double-observer data to explore distance issues.

4.5 Conclusion

Studying marine birds' usage of these dynamic environments is important in determining vulnerability to impacts. Such studies, and monitoring for any impacts, are key to the consenting process for marine renewable developments. Despite being a more cost effective approach, shore-based vantage point surveys poses a number of logistical challenges, which have the potential to bias the accuracy of results. My work has revealed these challenges, and provided some methods for overcoming some of them.

Chapter 5: A synthesis of marine bird diving behaviour to inform underwater collision risk with tidal-stream turbines

This chapter is in preparation as a paper to be submitted to a journal as ‘Robbins, A.M.C., Thaxter, C.B., Cook, A.S.C.P., Furness, R.W., Daunt, F. and Masden, E.A. Seabird diving behaviour to inform underwater collision risk with tidal stream turbines; a synthesis and data gaps’.

My contribution to this paper included conducting the literature searches for six of the twenty-two species, extracting dive and foraging parameters for analysis. I updated references for the remaining sixteen species and cleaned these data. I undertook the analysis and writing of the manuscript. The other co-authors undertook literature searches on the remaining species and assisted in writing and editing of the manuscript.

Supplementary materials for this chapter, including tables, figures and full list of references, are in Appendix 4.

Abstract

Tidal turbines have the potential to impact diving birds, primarily through collision with turbine blades. There is a legal requirement to assess these impacts. Collision risk modelling has been used widely to quantify collision risk to birds flying through wind farms. Intuitively, the same approach can be taken when assessing risk of underwater turbines to diving birds. Such models require data on a bird's foraging and diving behaviour to calculate their likely exposure to a tidal turbine array while foraging underwater. Accordingly, we have reviewed studies from peer-reviewed literature that present estimates for diving parameters for diving marine birds that occur in UK waters. These values can be used within underwater collision risk models. This work will provide a key resource to the consenting process as it can be used in the assessment of environmental impacts of marine renewable developments.

5.1 Introduction

Climate change is driving governments worldwide to set targets to reduce greenhouse-gas emissions through sustainable electricity generation. Consequently the marine renewable energy industry, including offshore wind, wave and tidal energy, is growing rapidly. With a quarter of Europe's tidal power in the seas around Scotland there is a strong desire to harness tidal energy through tidal stream turbines in this region (Marine Scotland, 2012). However, the industry is still in its infancy, with the first commercial array only recently consented in the Pentland Firth, and test sites, such as the European Marine Energy Centre in Orkney, providing one of the few locations to monitor environmental impacts in a test phase of development. Tidal stream devices are made up of a number of different parts: some devices require fixed structures within the water column, such as support piles, while floating devices incorporate substantial mooring equipment, including cables and chains that extend through the water column. The most common design for tidal stream energy generators is akin to that of a wind turbine using the current flow to rotate blades, however, there is great variation in design and technologies used to harness the tidal flow energy (see http://en.wikipedia.org/wiki/Tidal_stream_generator).

Underwater tidal turbines have the potential to affect seabird populations through collision with turbine blades, as well as disturbance and habitat loss (e.g. Wilson et al. 2007; Inger et al. 2009; ICES 2010; Langston et al. 2011; Furness et al. 2012). The issue of mortality through collision with man-made structures is not new; numerous studies have been undertaken on avian collision, particularly with wind turbines (e.g. Bevanger 1994; Garthe and Hüppop, 2004; Chamberlain et al. 2006; Drewitt and Langston, 2008; Lucas et al. 2008). The moving part(s) of the turbine structure are most commonly associated with collisions (Hüppop et al. 2006). Collisions with underwater turbines are likely to result in either death or injury; appendages utilised in propulsion are more vulnerable to collision, and if injured, may reduce foraging efficiency and mobility with consequences on body condition, ultimately increasing mortality risk. A number of factors can influence or cause a bird to collide with a structure, such as the location, structural attributes, weather or hydrographic conditions, as well as the birds' morphology and behaviour. Evidence from wind farms shows that species vary in both their vulnerability to collision and the ability to avoid structures (Garthe and Hüppop, 2004; Lucas et al. 2008; Furness et al. 2013; Bradbury et al. 2014). Similarly, the design and location of tidal-stream turbines within the water column are both likely to influence which species are more vulnerable to collisions (Furness et al. 2012). For example, annular designs may provoke different detection rates and avoidance behaviours from birds, compared with horizontal axis designs. Likewise, the use of shields around the blades to increase water flow into the turbine could lead to entrapment (Grecian, et al. 2010). However, as few tidal stream turbines have been deployed, and methods for detecting underwater collisions are still developing, the magnitudes of any of these effects are not known.

Seabirds are typically long-lived with low birth rates and delayed maturity, and so their populations are most susceptible to changes in adult survival rates (Gaston, 2004). Around the UK, particularly the North Sea, seabird populations are already facing a number of anthropogenic pressures, which have been demonstrated, by long-term studies, to have affected their survival rates and breeding success (Mitchell et al. 2004; Wanless et al. 2010; Burthe et al. 2014). There is concern that marine renewable schemes have the potential to place a new pressure on seabird populations, many of which are already currently declining (Wanless et al. 2010; Foster and Marrs 2012; Burthe et al. 2014).

Seabirds occur in the UK in internationally important numbers. Consequently many of the breeding seabird colonies around the UK have been identified as Special Protection Areas (SPA), providing these populations with protection. In Scottish waters, a suite of additional marine SPAs have also been identified as part of work developing a network of SPAs in the marine environment (SNH et al. 2014). Several recently designated Marine Protected Areas (MPAs) include black guillemot *Cepphus grylle* as a protected feature as this is the only breeding seabird species in Scotland not included in the SPA network.

Many of the seabirds from protected populations have foraging ranges which overlap with areas identified for tidal-stream energy development. There is a legal requirement to assess the potential impacts of these developments through an Environmental Impact Assessment (EIA), and Habitat Regulations Appraisal (HRA) where SPA populations are involved.

In the absence of empirical evidence, collision risk modelling can be used to estimate the risk of bird collisions. Collision risk modelling approaches have been used widely in the context of terrestrial and marine wind farms (Band 2000; Band et al. 2007; Band et al. 2012) to quantify collision risk to flying birds and, intuitively, a similar method can be used when assessing the risk of underwater turbines to diving birds. However, approaches to assessing the risk of underwater turbines to diving birds are still in their infancy. A recent guidance note (Scottish Natural Heritage 2016b) summarises the three models that have been developed for collision risk assessment with tidal turbines: 1) the encounter rate model (ERM) (Wilson et al. 2007); 2) the collision risk model (CRM) (Band 2000; Band et al. 2007; Band et al. 2012); 3) the exposure time model (ETPM) (Grant, Trinder and Harding 2014). All these models calculate the likely exposure of a bird to a tidal turbine array while foraging underwater. Crucially, they all require data on the foraging and diving behaviour of seabirds, including dive depth, duration and frequency, descent and ascent speeds and foraging trip duration and frequency.

5.1.1 Diving strategies

The foraging and diving habitat preferences and behaviour of birds, and specifically their use of high-energy tidal streams in which the devices may be placed have important implications. These preferences are likely to affect how they might interact with devices and their vulnerability to negative impacts, yet our knowledge base on this is currently very limited. Research has been undertaken on aspects of diving behaviour, and with developments in animal-borne technology (including cameras and accelerometers) data on dive profiles and foraging behaviour are increasingly available (e.g. Daunt et al. 2003; Watanuki et al. 2008). However, species coverage has been extremely varied. Initially, much of the research focused on penguins (e.g. Lishman and Croxall, 1983) and then shifted to work on auks (e.g. Piatt and Nettleship, 1985), gannets (Garthe et al. 2000) and cormorants, including the European shag (e.g. Wanless et al. 1993), species which have been intensively studied. In contrast, for groups such as storm petrels, shearwaters and divers very little is known. Similarly, there are important knowledge gaps on diving behaviour outside of the breeding season, in immature or non-breeding individuals, or specifically within high-energy tidal environments (Furness et al. 2012).

Marine birds have evolved specific foraging methods to exploit their prey, however the species most vulnerable to collision are presumed to be those that forage within the water column where the turbines are deployed (Furness et al. 2012). One of the more common foraging methods is pursuit diving, utilised by penguins, divers, auks, cormorants and also at times by gannets (Gaston, 2004). Gannets also perform plunge dives, where they dive from a great height in the air into the water, using the force on entry to propel them underwater (Ropert-Courdert et al. 2004). Pursuit diving species vary in their methods of propulsion, i.e. wings or feet, which can lead to differences in swim speeds. Wing propulsion can generate thrust during retraction, while foot propulsion has a drag effect, resulting in an increase in overall drag at higher speeds during foot propulsion (Lovvorn, 2001). However, Richman and Lovvorn (2008) suggested that advantage to foot propulsion comes when the bird is required to undertake small forward movements or hover underwater, manoeuvres that benthic feeders utilise, whilst feeding along the bottom or navigating through complex shallow habitats. Interestingly, Duffy et al. (1987) observed that pigeon guillemots (*Cepphus columba*) used both feet and wings as they foraged along the bottom substrate. Other benthic feeders or birds foraging in shallow conditions also tend to be foot propelled i.e. divers, grebes, seaducks and cormorants (Lovvorn 1991).

Most foraging dives have three different stages; descent, feeding and ascent (Gaston, 2004). Dive shape variation is widely considered to represent benthic versus pelagic foraging, with pelagic prey items being caught primarily on V-shaped dives and benthic prey items being caught primarily on U-shaped dives (Elliott et al. 2009). However, species with strongly stratified epipelagic prey also show U-shaped dive patterns (Chappell et al. 1993), and some species show both dive patterns when foraging. For example, gannets display U-shaped dives when feeding on capelin schools and V-shaped dives when feeding on surface fish (Garthe et al. 2000). Many studies, such as Butler and Jones (1997) have noted a link between dive duration and depth, and that larger animals tend to remain submerged for longer than smaller animals, even at the same depths. Diving birds show some flexibility in relation to foraging strategies and changes in environmental conditions (Finney et al. 1999; Watanuku et al. 2008).

Diving birds face a trade-off between energy gain due to prey consumption against energy lost due to the energetic costs of diving (Heath and Gilchrist, 2010). Due to the physiological constraints of oxygen supply (Green et al. 2005) diving is often performed in bouts i.e. a number of successive dives, with relatively short pauses between dives (Butler and Jones, 1997). However, these bouts and time spent on the water vary with different species and also with the bird's own foraging needs, for example benthic foragers can make a full recovery between dives due to the sessile nature of their food source, whereas the pursuit of a mobile school of fish can result in birds postponing the full recovery, in order to maximise prey consumption (Ydenberg and Guillemette 1991). Ropert-Coudert et al. (2004) and Montevecchi et al. (2009) found gannets spent more time on the water following more intense diving bouts, which is likely to have enabled digestion but also proposed that very long pauses were likely to have followed self-feeding bouts and shorter pauses were followed by return to the colony for chick provisioning.

Fraenkal (2006) suggested that diving birds may be swept around turbine blades due to the flow of water. However, at least in some species, foraging underwater involves swimming against the current (Heath et al. 2006; Shiomi et al. 2008), so that considering diving birds as inert particles carried by the water flow would be inappropriate.

5.1.2 Scope of review and aim

In this chapter, I review the peer-reviewed literature for estimates of diving and foraging variables for each diving seabird and other marine bird species that occurs in UK waters (divers, grebes, shearwaters, gannet, cormorants, seaducks and auks). The aim of this chapter is to provide representative values, from peer-reviewed literature, for marine bird diving and foraging behaviour for use within underwater collision risk assessments. Furthermore, I seek to identify current knowledge gaps on diving behaviour to help focus future research and highlight areas of uncertainty within impact assessments.

5.2 Methods

An extensive literature review was undertaken, compiling all available information on diving and foraging behaviour in peer-reviewed literature for 18 parameters and 22 diving species of marine birds that occur in UK waters. To narrow the scope of the species reviewed, only UK seabird species that have been recorded diving to within or beyond the depth range occupied by tidal turbines (3-80m) (Langton et al. 2011) were included. The variables were identified by including all foraging and diving variables used within the three underwater collision risk models that are currently in development and/or use. These variables included dive depth, duration and frequency, pause duration, bout duration and ratio, swimming speeds (including ascent, horizontal and descent) and foraging trip duration and frequency. Primary references have been used where possible; where this was not possible, secondary references have been retained, as detailed in Appendix 4.

5.2.1 Estimation of parameters

To calculate the diving and foraging metrics the approach adopted by Thaxter et al. (2012) was used to weight studies based on quality of the methods used. To do this, studies were grouped into the following categories:

1) **Direct studies.** These studies used the most technologically advanced methods including bird-borne devices such as the use of time-depth recorders, satellite and GPS tags, PTT (platform transmitter terminal) loggers, radio-tracking (VHF) devices. Similarly, direct underwater observations including video and multi-beam sonar were included where species ID was also confirmed through observation.
2) **Observational studies.** These studies included behavioural observations made either from land, boat or aerial platforms.
3) **Indirect studies.** This category included indirect observations such as incidental bycatch in fishing nets.
4) **Speculative studies.** This category included anecdotal or speculative observations, or approximated information.

This enabled prioritisation of studies where direct methods have been used to calculate variables and also provided a measure of confidence for the data reviewed (see Table 5.1). For each variable and grouping the following were calculated:

1) **Maximum**. The maximum value from all studies reviewed.
2) **Mean maximum**. The mean of the maximum range reported for all studies. For studies that provide a range of maximum values the mean value was taken, but where multiple studies were undertaken at a single site mean values were averaged across all studies.
3) **Global mean**. The global or grand is the mean of means, for a given variable, reported for each study, averaged across all studies. For some studies this includes a ‘most’ or median value. Where possible, a standard deviation around the global mean is also presented. Variance data were collated as part of the literature review, however, due to the mismatch of values presented it was not possible to combine these to estimate a pooled variance. Where only one study reports a mean value that is presented with the variance reported.

Many studies reported more than one value per measure as previously noted by Thaxter et al. (2012), such as differences between stages of the breeding season or sexes. In addition, multiple studies were from the same study sites and multiple years were reported in some individual studies. To reduce any bias towards better-studied locations, data were averaged across these measures for each study location, before averaging across all study locations to calculate the global mean. A number of these decisions had to be made on an ad-hoc basis to enable comptability between studies.

Table 5.1. Definitions of confidence measures

Confidence Measure	Definition
High	>5 direct studies
Moderate	2-5 direct studies
Low	Observation studies or only 1 direct study
Uncertain	Indirect estimates
Poor	Few indirect estimates, speculative or unknown data sources

While many studies on diving behaviour have been undertaken on captive marine birds these have been excluded from this literature review. Captive birds are not exposed to the same energetic costs as free-ranging birds and similarly restrictions on a dive tank environment may limit behaviour observed, therefore metric estimates may not be applicable to wild situations.

5.2.2 Tidal turbine variables

To explore the overlap of the different species' diving depth range with current tidal-stream turbine designs, a number of tidal developers were approached for device parameter estimates for moving, static and mooring parts. These included support structure height from seabed, distance from seabed to hub centre, rotor diameter and clearance from highest blade tip to surface at lowest astronomical tide (LAT). Noting that only devices from developers that responded could be represented within figure 5.1, other devices of differing parameters may also be in development.

5.3 Results

A total of 225 studies (192 primary and 35 secondary sources) were reviewed, which when considered by species (i.e. where multiple species were reported in a study) totalled 292 species-studies. Most of the studies within this literature review used either direct or observational methods (38.2% and 32.9% respectively). The remaining studies consisted of 8.9% indirect methods, 2.2% speculative methods and 18.7% used unknown methods. The last is likely to be a consequence of 15.6% of the studies coming from secondary sources (refer to table 5.2 for method types by species). Overall, the most parameters are available for razorbill, common guillemot and European shag (refer to table 5.3), however the most studies and more studies using direct methods were for northern gannet. This reflects that many studies have been carried out on northern gannet foraging behaviour but with fewer studies reporting the range of diving parameters. The species with fewest studies and parameters reported was the Manx shearwater. Other species with few parameters reported and/or few direct studies included diver and grebe species and seaducks such as greater scaup and common scoter (refer to table 5.3).

Table 5.2 Number of studies (both primary and secondary) contributing foraging and diving information for each category of direct (DI), observational (OB), indirect (IN) and speculative (SP) with overall assessed confidence in data for that species.

	Category						
Species	**DI**	**OB**	**IN**	**SP**	**UK**	**Total**	**Confidence**
Red-throated diver	0	2	1	0	8	11	*Low*
Black-throated diver	0	2	0	0	2	4	*Low*
Great northern diver	0	13	1	0	2	16	*Low*
Great-crested grebe	0	6	2	0	6	14	*Low*
Slavonian grebe	0	4	0	0	6	10	*Low*
Sooty shearwater	5	2	0	1	0	8	*High*
Manx shearwater	1	0	0	0	0	1	*Low*
Northern gannet	23	4	1	0	3	31	*High*
Great cormorant	10	6	0	0	0	16	*High*
European shag	8	5	0	0	1	14	*High*
Greater scaup	0	3	1	0	4	8	*Low*
Common eider	2	5	2	0	2	11	*Moderate*
Long-tailed duck	0	5	3	0	2	10	*Low*
Common scoter	1	9	4	0	6	20	*Low*
Velvet scoter	3	7	4	0	7	21	*Moderate*
Common goldeneye	0	8	4	0	6	18	*Low*
Red-breasted merganser	0	2	1	0	1	4	*Low*
Common guillemot	19	6	3	4	1	33	*High*
Razorbill	13	1	1	1	0	16	*High*
Black guillemot	4	2	1	0	0	7	*Moderate*
Little auk	6	2	0	0	3	11	*High*
Atlantic puffin	7	0	1	0	0	8	*High*

Table 5.3 Ranking of species by number of studies, number of direct studies and score for the number of reported parameters used in literature review. For each reported variable a score was given; 0 for no studies, 1 for one study, or 2 for more than one study. These were then totalled for each species and the ranking based on the total.

Species	**Studies**	**Direct Studies**	**Reported Variables**
Razorbill	*6*	*3*	*1*
Common guillemot	*2*	*2*	*2*
European shag	*9*	*5*	*3*
Little auk	*11*	*7*	*4*
Atlantic puffin	*16*	*6*	*5*
Common eider	*11*	*11*	*6*
Great cormorant	*6*	*4*	*6*
Great northern diver	*6*	*14*	*6*
Long-tailed duck	*15*	*14*	*6*
Northern gannet	*1*	*1*	*10*
Black guillemot	*19*	*9*	*11*
Common goldeneye	*5*	*14*	*11*
Great-crested grebe	*10*	*14*	*11*
Velvet scoter	*4*	*10*	*11*
Black-throated diver	*20*	*14*	*15*
Common scoter	*3*	*12*	*15*
Greater scaup	*16*	*14*	*15*
Red-breasted merganser	*20*	*14*	*15*
Red-throated diver	*11*	*14*	*15*
Slavonian grebe	*14*	*14*	*20*
Sooty shearwater	*16*	*8*	*20*
Manx shearwater	*22*	*12*	*22*

Table 5.4 gives an overview of number of study sites, number of birds and number of study years (where values were reported), note that if a number studies were carried out over the same years in the same location, those years were only counted once. The countries where the most studies have been carried out are the USA, UK and Canada (refer to Fig. 5.1).

Table 5.4 Overview of study site locations, total number of birds and study years reported, however not all studies reported these values. References for each species are also given (refer to Appendix 4).

Species	Study sites	Total Birds	Site years	References
Red-throated diver	7	-	6	16; 37; 47; 51; 58; 100; 105; 119; 151; 201
Black-throated diver	4		2	24; 109; 157; 162
Great northern diver	14	5	22	2; 6; 32; 45; 46; 66; 70; 117; 138; 146; 166; 170; 178; 193; 200
Great-crested grebe	8	199	13	37; 52; 89; 91; 95; 101; 105; 119; 126; 161; 182; 189; 203; 225
Slavonian grebe	4	-	4	13; 38; 49; 53; 60; 103; 108; 122; 161; 212
Sooty shearwater	7	151	9	22; 48; 50; 148; 183; 184; 198; 218
Manx shearwater	1	-	1	72
Northern gannet	13	563	13	20; 29; 30; 37; 61; 62; 63; 64; 65; 78; 86; 87; 88; 90; 121; 122; 127; 128; 129; 140; 142; 153; 158; 159; 163; 167; 171; 179; 190; 192; 207
Great cormorant	16	235	10	35; 73; 74; 75; 76; 77; 79; 80; 114; 123; 125; 172; 173; 174; 195; 221
European shag	8	274	24	9; 29; 31; 35; 45; 75; 123; 134; 210; 211; 212; 213; 214
Greater scaup	4	338	9	21; 37; 38; 42; 45; 135; 136; 145
Common eider	6	199	19	11; 26; 81; 82; 83; 84; 96; 97; 116; 145; 223
Long-tailed duck	7	25	9	4; 5; 14; 27; 47; 105; 145; 168; 193
Common scoter	14	1561077	26	12; 37; 41; 42; 43; 45; 68; 69; 104; 111; 112; 135; 136; 141; 147; 155; 156; 182; 188; 204
Velvet scoter	10	397	26	3; 10; 21; 37; 42; 45; 68; 85; 102; 103; 118; 130; 131; 132; 169; 177; 188; 205; 206; 224; 225
Common goldeneye	14	24023	16	17; 21; 37; 44; 45; 47; 49; 54; 99; 105; 108; 144; 145; 149; 154; 155; 197
Red-breasted merganser	4	-	10	47; 105; 118; 145
Common guillemot	19	327	47	1; 8; 9; 19; 33; 39; 94; 110; 115; 133; 152; 160; 185; 187; 199; 215; 217
Razorbill	11	134	22	9; 15; 19; 34; 40; 97; 113; 118; 137; 162; 170; 195; 197; 210; 226; 228
Black guillemot	6	224	13	18; 28; 137; 160; 186; 208; 209
Little auk	8	253	17	23; 57; 59; 92; 93; 106; 107; 113; 219; 220; 222
Atlantic puffin	6	46	7	7; 9; 25; 94; 160; 185; 191; 215

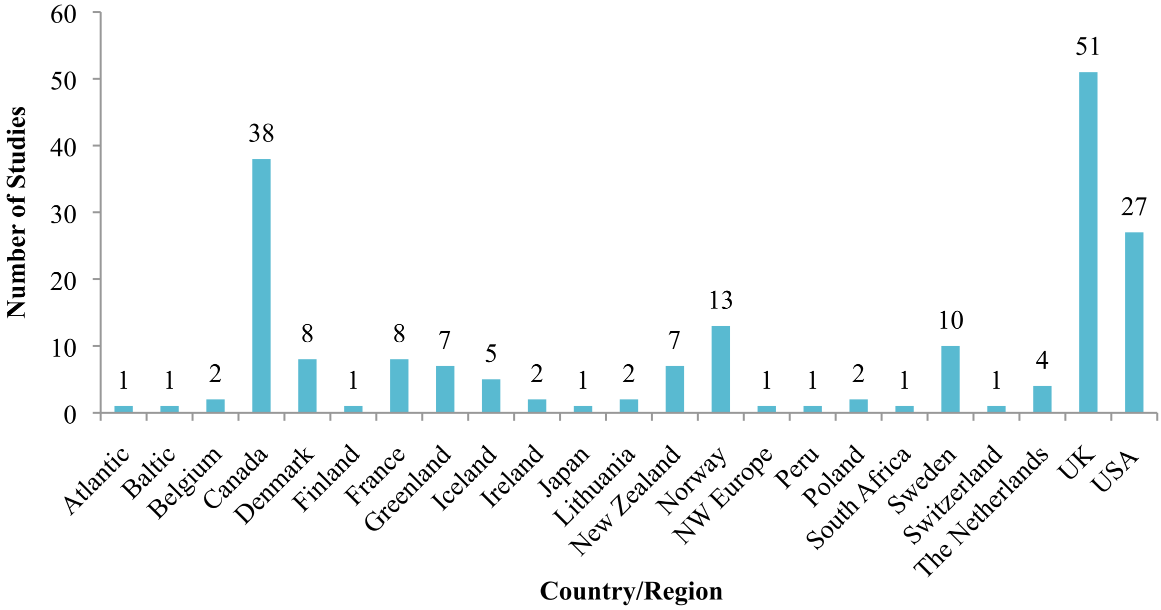

Figure 5.1 Number of studies by country/region. Six studies were undertaken in multiple countries; Canada and UK, 1; France and UK, 1; Greenland and Norway, 3; UK and Ireland, 1. We were unable to identify the location for thirty-six studies. These have all been excluded from the table below.

5.3.1 Diving Behaviour

Dive depth

A maximum dive depth estimate was obtained for all species, excepting Manx shearwater and Slavonian grebe. Where more than one maximum dive depth was reported it was possible to estimate a mean maximum, and a global mean dive depth value was estimated for 15 of the species reviewed (table 5.5). Figure 5.2 highlights the overlap between the depth frequently dived by the marine bird species reviewed (based on the global mean) and the depth capable of diving (based on the maximum value reported) with the typical depth of a range of tidal turbine devices, noting that these are likely to vary from site to site. Common guillemot and European shag have the greatest global mean diving depth reported (32.2m ± 4.7SD, *n*=5; 22.1m ±10.5SD, *n*=3). The species with highest confidence values are common guillemot and razorbill.

Figure 5.2 The foraging depths of diving marine birds reviewed in this study, including the depth capable of diving and frequently dived (maximum and global mean values from table 5.4) and estimates of the depth of moorings, moving parts and static parts for a range of tidal turbines devices when placed at their optimal operating depths.

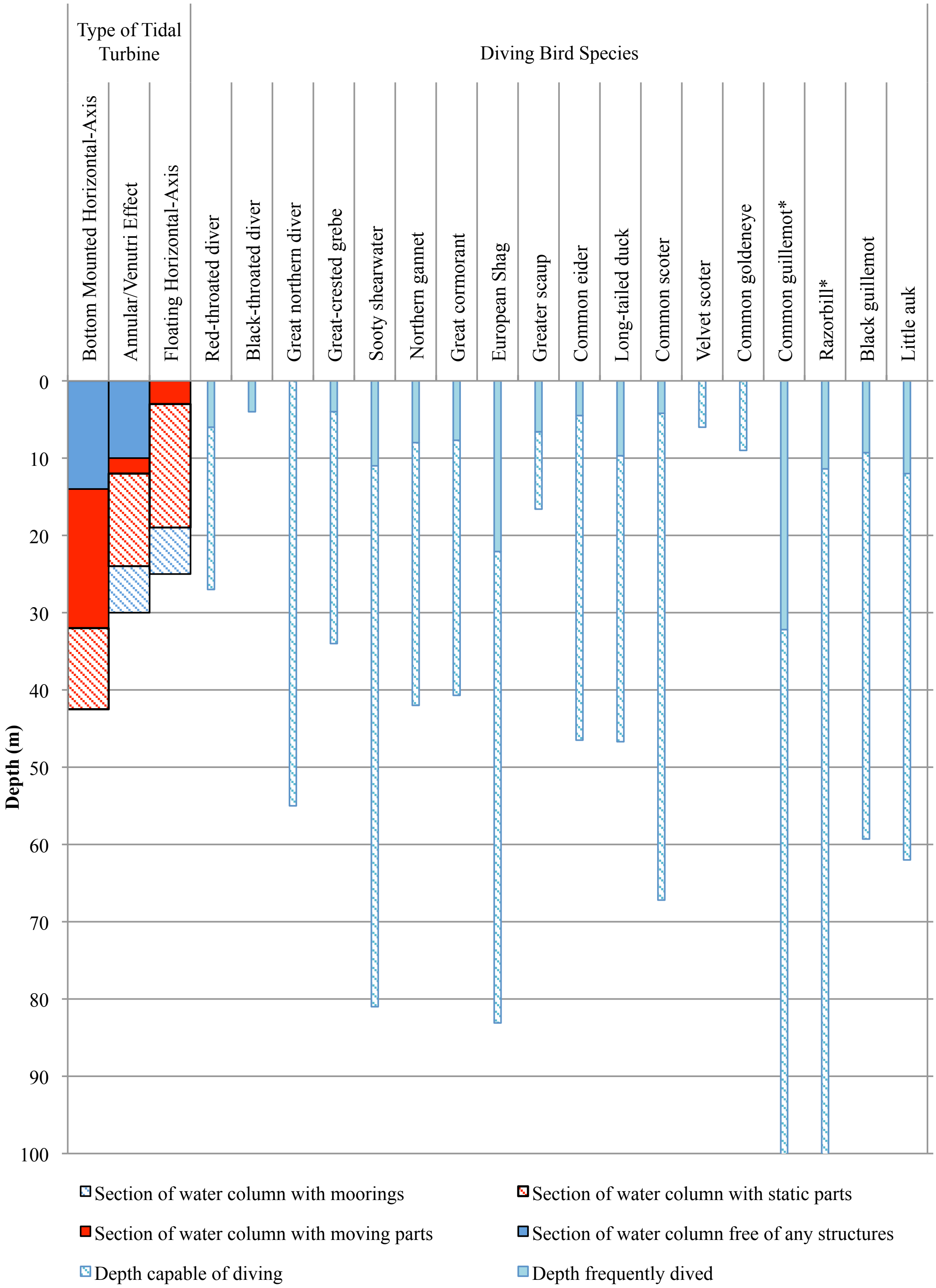

Table 5.5 Summary of dive depth values (m), including maximum, mean maximum and global mean; error presented is ±1SD and the sample sizes are given in parentheses. Sample sizes refer to the number of pieces of information used after averaging across multiple estimates for individual studies and/or colonies – see full description in methods. The category of studies used is the best available method for providing the estimate, prioritising direct, observation, indirect, speculative/all data – see text for full description; a measure of confidence is given for each species – refer to table 5.1.

	Dive Depth			**Category used**	**Confidence in data**
Species	**Max**	**Mean max.**	**Mean**		
Red-throated diver	21	12.3±4.8 (6)	6.0±2.8 (2)	All	*Uncertain*
Black-throated diver	-	-	2 - 4 (1)		*Low*
Great northern diver	55	-	-	Indirect (All)	*Uncertain*
Great-crested grebe	30	11.0±13.2 (4)	2 - 4 (1)	All	*Low*
Slavonian grebe	5	-	-	All	*Low*
Sooty shearwater	70	49.0±28.6 (4)	11.2±4.5 (3)	Direct (All)	*Moderate*
Manx shearwater	-	-	-	-	-
Northern gannet	34	19.2±8.6 (6)	8.0±7.8 (4)	All	*Moderate-High*
Great cormorant	33	27.5±5.9 (4)	7.7±2.3 (4)	All	*Moderate*
European shag	61	46.9±12.2 (3)	22.1±10.5 (3)	Direct (All)	*Moderate*
Greater scaup	10	6.6±2.7 (4)	-	All	*Uncertain*
Common eider	42	19.3±15.8 (4)	4.5±2.5 (3)	All	*Low*
Long-tailed duck	37	23.8±18.1 (2)	-	All	*Low*
Common scoter	63	21.8±27.5 (4)	9.7±9.0 (3)	All (except unknown)	*Low*
Velvet scoter	35	24.6±10.3 (5)	4.2±3.1 (3)	All	*Low*
Common goldeneye	6	4.8±1.1 (2)	-	All	*Uncertain*
Red-breasted merganser	9	-	-	All	*Low*
Common guillemot	177	98.0±59.5 (5)	32.2±4.7 (5)	Direct Only	*High*
Razorbill	140	51.8±42.2 (10)	11.4±4.8 (5)	All	*High*
Black guillemot	50	39.9±14.5 (4)	9.3±2.8 (1)	All	*Low-Moderate*
Little auk	50	43.9±8.6 (2)	12.0±11.0 (1)	All	*Low*
Atlantic puffin	68	49.0±12.9 (6)	10.8±1.5 (2)	All	*Moderate-High*

Dive duration

Mean dive duration values were reported for every species reviewed, except Manx shearwater. Maximum dive durations were reported for all species excluding Manx and sooty shearwater and little auk (refer to table 5.6). Common and black guillemot had the longest mean dive durations (73.1±21.6 *n*=6, 68.8±14.9 *n*=6), which were almost double the mean duration for two of the other auk species, razorbill and Atlantic puffin (36.0±11.7 *n*=5, 35.5±10.8 *n*=4). Northern gannet had the shortest mean duration (7.3±2.3 *n*=4). Great northern diver, Slavonian grebe, great cormorant, European shag, common guillemot, black guillemot and Atlantic puffin all had maximum dive durations of over 100 seconds. The confidence for many species is low due to the number of studies that used observations rather than tracking data to record dive durations.

Table 5.6 Summary of dive duration values (sec.), including maximum, mean maximum and global mean; error presented is ±1SD and the sample sizes are given in parentheses. Sample sizes refer to the number of pieces of information used after averaging across multiple estimates for individual studies and/or colonies – see full description in methods. The category of studies used is the best available method for providing the estimate, prioritising direct, observation, indirect, speculative/all data – see text for full description; a measure of confidence is given for each species – refer to table 5.1.

	Dive Duration			**Category used**	**Confidence in data**
Species	**Max**	**Mean max.**	**Mean**		
Red-throated diver	43	-	27.2±1.6 (4)	All	*Low*
Black-throated diver	63	-	32.3±18.6 (1)	Obs. (All)	*Low*
Great northern diver	124	78.7±26.7 (7)	40.0±15.6 (13)	All	*Low*
Great-crested grebe	62	43.8±10.9 (8)	26.2±11.1 (8)	All	*Low*
Slavonian grebe	160	62.9±60.0 (6)	23.2±7.1 (4)	All	*Low*
Sooty shearwater	-	-	20.8±16.8 (3)	All	*Low*
Manx shearwater	-	-	-	-	-
Northern gannet	40	22.3±9.2 (5)	7.3±2.3 (4)	All	*Moderate-High*
Great cormorant	152	92.2±33.4 (3)	29.8±13.2 (10)	All	*Moderate*
European shag	163	75.1±25.0 (3)	41.1±11.8 (7)	All	*Moderate-High*
Greater scaup	35	30.6±4.4 (4)	23.0±5.0 (4)	All	*Low*
Common eider	59	-	34.4±13.8 (5)	All	*Low*
Long-tailed duck	70	54.8±13.7 (5)	41.3±11.8 (6)	All	*Low*
Common scoter	44	36.0±8.5 (2)	29.7±6.3 (3)	All	*Low*
Velvet scoter	65	57±7.5 (3)	33.8±12.8 (4)	All	*Low*
Common goldeneye	41	31.9±8.2 (4)	22.9±5.7 (8)	All	*Low*
Red-breasted merganser	47	35.2±10.5 (3)	30.0±12.0 (3)	All	*Low*
Common guillemot	249	200.8±50.3 (5)	73.1±21.6 (6)	Direct Only	*High*
Razorbill	93	74.0±21.6 (4)	36.0±11.7 (5)	All	*Moderate-High*
Black guillemot	131	-	68.8±14.9 (6)	All	*Moderate*
Little auk	-	-	41.0±5.7 (2)	Direct (All)	*Moderate*
Atlantic puffin	123	119.1±5.7 (2)	35.5±10.8 (4)	Direct (All)	*Moderate*

Pause duration

Mean pause durations were reported for 18 of the 22 species reviewed, however maximum values were only found for eight of the species and mean maximums could only be calculated for four species (Table 5.7). Common eider and common guillemot had the longest mean pause durations (57.4±70.4 *n*=5, 34.8±13.3 *n*=4), while Atlantic puffin had the shortest mean pause duration (6.0±10.0 *n*=1). As fewer studies were found reporting pause durations, no species qualified for high confidence in the data.

Table 5.7 Summary of pause duration values (sec), including maximum, mean maximum and global mean; error presented is ±1SD and the sample sizes are given in parentheses. Sample sizes refer to the number of pieces of information used after averaging across multiple estimates for individual studies and/or colonies – see full description in methods. The category of studies used is the best available method for providing the estimate, prioritising direct, observation, indirect, speculative/all data – see text for full description; a measure of confidence is given for each species – refer to table 5.1.

Species	**Pause Duration**			**Category used**	**Confidence in data**
	Max	**Mean max.**	**Mean**		
Red-throated diver	-	-	12.2±4.4 (1)	All	*Uncertain*
Black-throated diver	-	-	10 - 18 (1)	Obs. (All)	*Low*
Great northern diver	120	81.0±55.2 (2)	21.5±10.7 (4)	All	*Low*
Great-crested grebe	-	-	-	-	-
Slavonian grebe	18	16.3±1.8 (2)	12.4±0.8 (1)	All	*Low*
Sooty shearwater	-	-	-	-	-
Manx shearwater	-	-	-	-	-
Northern gannet	291	-	-	Direct (All)	*Low*
Great cormorant	32	-	11.1±3.0 (7)	All	*Low-Moderate*
European shag	-	-	22.2±19.5 (6)	All	*Moderate*
Greater scaup	-	-	13.5±3.5 (1)	Indirect (All)	*Uncertain*
Common eider	-	-	57.4±70.4 (5)	All	*Low*
Long-tailed duck	28	21.1±10.0 (2)	11.0±2.4 (5)	All	*Low*
Common scoter	-	-	17.7±9.5 (1)	All	*Low*
Velvet scoter	-	-	17.6±8.6 (1)	All	*Low*
Common goldeneye	21	-	11.7±3.9 (3)	All	*Low*
Red-breasted merganser	29	-	17.4±9.3 (2)	All	*Low*
Common guillemot	-	-	34.8±13.3 (4)	Direct Only	*Moderate*
Razorbill	60	47.5±17.7 (2)	24.3±4.9 (4)	Direct (All)	*Moderate*
Black guillemot	-	-	31.2±19.7 (1)	Direct (All)	*Low*
Little auk	-	-	21.5±2.4 (3)	Direct (All)	*Low-Moderate*
Atlantic puffin	-	-	6.0±10.0 (1)	Direct (All)	*Low*

Diving Bouts

A number of dive bout metrics were found in the literature review; however these were inconsistently reported between and within species, so that it has not been possible to report any clearly-defined diving bout values for a number of species. We found the most studies reporting bout values were undertaken on auk species, and these also were all direct studies (refer to table 5.8). Other species groups such as divers, cormorants and seaducks had some values reported but many of these were not direct and so confidence in these data is classified as low. Both common guillemot and black guillemot had a similar number of dives per bout (6.4±1.2 *n*=4, 6.5±1.2 *n*=1), as did razorbill and little auk (10.4±8.8 *n*=3, 10.2±2.8 *n*=3). Atlantic puffin, however, had more than double the number of dives per bout (28 *n*=1), and the mean bout duration was ~10 seconds longer than for common guillemot (22.9±3.8 *n*=2, 12.7±3.5 *n*=3). European shag had the longest mean bout duration (85.6±86.1 *n*=2), although it is worth noting both the low confidence rating and high SD for this species.

Table 5.8 Summary of dive bout values, including dives/bout, mean bout duration (min) and dives/minute; error presented is ±1SD and the sample sizes are given in parentheses. Sample sizes refer to the number of pieces of information used after averaging across multiple estimates for individual studies and/or colonies – see full description in methods. The category of studies used is the best available method for providing the estimate, prioritising direct, observation, indirect, speculative/all data – see text for full description; a measure of confidence is given for each species – refer to table 5.1.

Species	Dives/Bout	Mean Bout Duration (min)	Dives/minute	Category used	Confidence in data
Red-throated diver	-	-	-	-	-
Black-throated diver	-	-	*18.4 (1)*	*Obs. (All)*	*Low*
Great northern diver	*2.1:1 ratio*	-	*0.26*	*All*	*Low*
Great-crested grebe	-	-	-	-	-
Slavonian grebe	-	-	-	-	-
Sooty shearwater	-	-	-	-	-
Manx shearwater	-	-	-	-	-
Northern gannet	-	-	-	-	-
Great cormorant	23.3±39.7 (5)	16.5±5.5 (1)		All	Moderate
European Shag	-	85.6±86.1 (2)	1.3	All	Low
Greater scaup	-	-	-	-	-
Common eider	-	54.9±7.1 (1)	-	Obs. (All)	Low
Long-tailed duck	*4.2±0.1 (2)*	-	-	*Obs. (All)*	*Low*
Common scoter	-	-	-	-	-
Velvet scoter	-	-	-	-	-
Common goldeneye	-	-	-	-	-
Red-breasted merganser	-	-	-	-	-
Common guillemot	6.4±1.2 (4)	12.7±3.5 (3)	-	Direct (All)	Moderate
Razorbill	10.4±8.8 (3)	10.8±4.3 (3)	-	Direct (All)	Moderate
Black guillemot	6.5±1.2 (1)	-	-	Direct (All)	Low
Little auk	10.2±2.8 (3)	-	-	Direct (All)	Moderate
Atlantic Puffin	28	22.9±3.8 (2)	-	Direct (All)	Moderate

Swim speeds

We report swim speeds for eight species (refer to table 5.9); however we only found multiple studies on common guillemot, razorbill and little auk. Northern gannet had fastest descent speed reported (5.7m/s^{-1}), while velvet scoter had the slowest descent speed (0.4m/s^{-1}). We were only able to report horizontal speeds for three species; European shag, common eider and Atlantic puffin, their horizontal swim speeds ranged from 1.3 to 1.8m/s^{-1}. The three species with multiple studies were assigned a confidence score of moderate, while others were given a low confidence score.

Table 5.9 Summary of swim speed values, including ascent, descent and horizontal speeds (m/s^{-1}); error presented is ±1SD and the sample sizes are given in parentheses. Sample sizes refer to the number of pieces of information used after averaging across multiple estimates for individual studies and/or colonies – see full description in methods. The category of studies used is the best available method for providing the estimate, prioritising direct, observation, indirect, speculative/all data – see text for full description; a measure of confidence is given for each species – refer to table 5.1.

	Swim Speed (m/s^{-1})			**Category used**	**Confidence in data**
Species	**Ascent**	**Descent**	**Horizontal**		
Red-throated diver	-	-	-	-	-
Black-throated diver	-	-	-	-	-
Great northern diver	-	-	-	-	-
Great-crested grebe	-	-	-	-	-
Slavonian grebe	-	-	-	-	-
Sooty shearwater	-	-	-	-	-
Manx shearwater	-	-	-	-	-
Northern gannet	-	5.7±1.7 (1)	-	Direct (All)	*Low*
Great cormorant	0.9±0.7 (2)	1.1±0.8 (2)	0.9±0.1 (2)	Direct (All)	*Moderate*
European shag	1.7±0.1 (1)	1.3±0.1 (1)	1.7-1.8 (1)	All	*Low*
Greater scaup	-	-	-	-	-
Common eider	-	1.1±0.2 (1)	1.3±0.1 (1)	Direct (All)	*Low*
Long-tailed duck	-	-	-	-	-
Common scoter	-	-	-	-	-
Velvet scoter	-	0.4±0.0 (1)*	-	All	*Low*
Common goldeneye	-	-	-	-	-
Red-breasted merganser	-	-	-	-	-
Common guillemot	1.1±0.3 (3)	1.1±0.2 (3)	-	Direct (All)	*Moderate*
Razorbill	0.9±0.3 (3)	0.8±0.3 (3)	-	Direct (All)	*Moderate*
Black guillemot	-	-	-	-	-
Little auk	0.9±0.2 (3)	-	-	Direct (All)	*Low-Moderate*
Atlantic Puffin	-	-	1.5±0.1 (1)	Direct (All)	*Low*

5.3.2 Foraging behaviour

We reviewed a range of metrics for foraging behaviour, however these were not consistently reported between species and/or studies (refer to table 5.10). We were only able to report one or more of these metrics for 10 of the species reviewed. We found more studies reported foraging dives/trip and/or foraging trips/day than reported dive bouts/day. Some studies reported metrics such as dives/hour.

Table 5.10 Summary of foraging values, including dives/bout, mean bout duration (min) and dives/minute; error presented is ±1SD and the sample sizes are given in parentheses. Sample sizes refer to the number of pieces of information used after averaging across multiple estimates for individual studies and/or colonies – see full description in methods. The category of studies used is the best available method for providing the estimate, prioritising direct, observation, indirect, speculative/all data – see text for full description; a measure of confidence is given for each species – refer to table 5.1.

	Additional Foraging Trip Metrics					
Species	**Dives/ trip**	**Dives/day**	**Trips/day**	**Bouts/ day**	**Category used**	**Confidence in data**
Red-throated diver	-	-	20±7.3 (1)^1	-	All	Poor
Black-throated diver	-	-	-	-	-	-
Great northern diver	34 (1)	-	-	-	All	Uncertain
Great-crested grebe	-	-	2 (1)^2	-	Obs. (All)	Uncertain
Slavonian grebe	-	-	-	-	-	-
Sooty shearwater	-	-	-	-	-	-
Manx shearwater	-	-	-	-	-	-
Northern gannet	38.2±3.9 (2)	-	-	-	Direct (All)	Low-Moderate
Great cormorant	-	152.4±81.8 (1)	-	6.0±1.6 (1)	Direct (All)	Low
European shag	27.3±2.4 (1)		2.7±0.5 (1)^2	-	All	Low
Greater scaup	-	-	-	-	-	-
Common eider	-	-	-	-	-	-
Long-tailed duck	89 (1)	-	-	-	Obs. (All)	Low
Common scoter	-	-	-	-	-	-
Velvet scoter		-	-	-	-	-
Common goldeneye	-	-	-	-	-	-
Red-breasted merganser	-	-	-	-	-	-
Common guillemot	55.3± 33.6 (3)	52 (1)	3.2±1.4(1)^2	-	Direct (All)	Low-Moderate
Razorbill	268.3± 227.7(4)	397±105 (1)	2.4±1.2 (1)^2		Direct (All)	Low-Moderate
Black guillemot	-	186±79 (1)	-	-	Direct (All)	Low
Little auk Short trip	67.0± 10.8 (3)	366± 96.7 (3)	4.4±0.4 (3)^2	-	Direct (All)	Low-Moderate
Long trip	289.7± 65.4 (3)	-	-	-	-	-
Atlantic puffin	-	332.9± 107.6(2)		32.1± 11.9(1)	Direct (All)	Low-Moderate

^{1} per pair
^{2} per individual

5.4 Discussion

This study provides the most extensive and up-to-date review of diving and foraging behaviour for UK marine bird species. It has been possible to report a number of diving and foraging variables for most of the species reviewed and through this to identify knowledge gaps including the species and parameters for which there is a paucity of studies. Two online resources present similar information; Ropert-Courdert and Kato (2012) (http://penguinessbook.scarmarbin.be/), have collated peer-reviewed literature on dive depth and duration for diving birds, mammals, sea-turtles and other species; Birdlife International (http://seabird.wikispaces.com/) was online at the start of this review, but subsequently has gone offline due to hosting issues. This provided a useful resource covering many additional species and a wide range of ecological parameters.

5.4.1 Foraging and diving strategies

The results show a number of differences in both foraging and diving strategies, both between species groups and within related species. For example, Atlantic puffin had more than double the number of dives per bout reported for other auk species, with shorter dive and pause durations, suggesting their foraging strategy is to take more frequent, shorter dives. These differences in foraging between related species has been noted in other studies such as Martin and Wanless (2015). While there are ecological benefits, such a niche partitioning, which reduces interspecies competition, this also has consequences for a species' vulnerability to encountering tidal turbines and their potential collision risk. However, due to the variation in tidal turbine technology our results highlight that the potential collision risk could vary considerably dependent on the location of the moving turbine structure within the water column. Finally, the differences in foraging and diving strategies highlights potential issues in the application of these parameters from proxy species, where a species has been poorly studied and the parameter is otherwise unavailable. We would therefore advise caution in doing so.

5.4.2 Applications: underwater collision risk modelling

This review should aid impact assessments required for marine renewable developments; we provide many of the values that are required to undertake underwater collision risk modelling. The parameters were selected from those required for the three models included within the Scottish Natural Heritage (2016b) underwater collision risk modeling guidance, and the provision of maximum, mean maximum and mean values (with variation) enables uncertainty around diving behaviour to be taken into account. Additionally, within the guidance the parameters such as dive and pause duration can also be used to calculate areal density, adjusting the density of birds used to correct for birds that were not observed while underwater. Therefore this review could similarly be used to calculate availability bias.

We have been able to identify that for many species, including razorbill, Atlantic puffin, black guillemot, great cormorant and northern gannet the depth they have been reported as capable of diving to overlaps with the optimal depth for a number of the tidal devices, however the depth at which some of these species frequently dive to overlaps with fewer devices. Both turbine depth and foraging depth will be dependent on site-specific factors, such as bathymetry and hydrodynamics. We therefore suggest this can only be used as a guide to the sensitivity of species and does not obviate the need for site-specific surveys. Nevertheless, the data in this review may help to identify whether or not tidal devices deployed at different depths are likely to represent a risk to particular marine bird species.

Well-studied species, such as European shag, northern gannet and common guillemot are likely to be the most valuable for studying effects of wet renewables. They have proven tractable in both obtaining logger data including good sample sizes, and other associated data e.g. breeding performance. However, we currently know little about how any species interact with these underwater devices, so all species that are identified as being at risk from collision with tidal turbines within impact assessments will have to be considered for future research. Quantifying avoidance in the models will be challenging and highlights the importance for robust post-construction monitoring to inform future assessments.

5.4.3 Limitations

Most of the studies within this literature review used either direct or observational methods, however, it does not always follow that species with an overall high degree of confidence (i.e. more direct studies) have a full range of parameters reported. Studies report results based on the ecological interest of a particular species, and for a number of species that we included in this review their diving behaviour has not traditionally been the focus of research, therefore they may be well studied but not have many (or in the case of Manx shearwaters, any) studies reporting diving parameters. For example, northern gannet has a high degree of confidence in foraging behaviour but as fewer studies have focused on parameters relating diving behaviour after they have plunged into the water. We also observed inconsistencies in the way parameters, such as those relating to diving bouts are reported; dive ratio, diving efficiency, bottom time efficiency, dives per minute, dives per bout etc. This resulted in incompatibility when trying to compile results and therefore reduced the number of studies we could include for certain parameters.

Direct methods provide the highest level of confidence; however, there are limitations with these, for example operating limits of devices. We found inconsistencies in the way most parameters were reported or in some cases where results were only reported in graphs. The species with highest confidence values for dive depth are common guillemot and razorbill. However, it is worth noting that some methods e.g. capillary tubes (a direct method) and bycatch (an indirect method) only measure maximum dive depths. Therefore the confidence rating may be greater for the mean maximum than for the global mean, e.g. Atlantic puffin.

There are a number of limitations that apply to the use of 'generic' values, which are particularly relevant to this study. While the results present the best available data for the species reviewed, the results may not necessarily be representative of all regions. Parameters such as dive depth and dive duration are both influenced by the bathymetry of the foraging location; for example, birds that could dive deeper cannot do so where sea depth is shallow. Furthermore for nearly all species presented there is not a logical link between depth and dive duration, because of the balance of parameters across years or study location. For example, in European shag more studies have measured the dive duration than dive depth, including in shallower water, meaning that global mean dive duration is lower than would be expected given the global mean depth reported.

We noted when undertaking the mean maximum calculation that there were two ways of calculating this. The approach we used was for studies that provide a range of maximum values we took the mean value, where multiple studies were undertaken at a single site we averaged this mean value across all studies at that site. The alternative approach would have been for taking the upper value from studies that reported a range of maximum values and also for multiple studies at a single location that provide a range of maximum values. The first approach was applied because it resulted in a more representative mean maximum, rather than one that could have been skewed by an outlier.

As Thaxter et al. (2012) found, we also acknowledge the potential difficulty in basing representative foraging and diving parameters on a small number of high quality estimates instead of a larger number of lower quality estimates. However, we explored this for dive depth, duration and pause duration for common guillemot, (see Appendix 4) and found that while the standard deviation estimates were narrower for direct studies only, there was overlap for each parameter. Therefore for all other species we selected all studies as the most appropriate measure for deriving representative values due to more studies being available, rather than deriving representative estimates based on single direct studies (Table 6.2). However, for a number of species and parameters there was only one study found and therefore reported. When new data become available, refinement will be needed for many species presented here.

5.4.4 Future research

This review has highlighted that current knowledge of foraging and diving behaviour is highly variable across species and also across parameters. Some of the more vulnerable species, such as divers and black guillemot (Furness et al. 2012), are difficult to study and confidence in these data is lower. However, improvements in technology, such as remote downloading, may increase the potential for future studies of these species. This study has also shown there are inconsistencies in both the way parameters are reported and for many studies data are collected but then not published, and therefore are unavailable for use within environmental impact assessments. While many studies are published to answer a specific ecological question there are many more applied policy uses of those results. We therefore strongly recommend that researchers seek to make other parameters that have been estimated more widely available through supplementary material or collaborative databases.

Our results show that parameters relating to diving bouts, and foraging trips and diving bouts within the context of a day are inconsistently reported, similarly swim speed (excepting lab studies) is poorly reported. To improve any collision risk modelling there is a need to measure parameters such horizontal speeds at depth, proportion of time spent at different depths, and also to explore swim speeds in relation to current speeds. We acknowledge these are not readily measured with TDRs and standard accelerometers, hence a lack of estimates compared descent and ascent speeds, yet these are likely to be particularly relevant for tidal turbine collision risk.

5.5 Conclusion

This review has provided representative values, from peer-reviewed literature, for marine bird diving and foraging behaviour, which can be utilised in underwater collision risk assessments. We have provided a measure of confidence for each parameter reported so this can be taken into account within any modelling and assessment of collision risk. Furthermore we have been able to identify present knowledge gaps on diving behaviour, as well as issues in the way parameters are currently reported. It is hoped that this study can be used to help focus future research and highlight areas of uncertainty within environmental impact assessments.

Chapter 6: General Discussion

6.1 Summary of main findings

With the wet renewable energy industry still in its infancy and many of the potential impacts on seabirds still largely unknown, this thesis aimed to address key knowledge gaps relating to our understanding of seabird ecology in high-energy marine environments. In this final chapter I bring together the results of this thesis and discuss them in the context of the two key aims. Firstly, to identify spatial and temporal patterns of use and behaviour of seabirds in these high-energy environments and to expand the scientific basis used to assess the impacts of marine renewable energy devices (MREDs) on marine birds. The second aim was to increase our understanding of bird survey requirements for wet renewable energy schemes by optimising methods for environmental impact assessment.

This study has demonstrated that marine birds show differing spatial and temporal patterns of abundance and behaviour within high-energy marine environments. By using shore-based vantage point methods in three high-energy areas in the Northern Isles, suitable for wet renewable energy arrays, I also have been able to explore some variations in vantage point methods.

Chapters 2 and 3 explored two existing long-term wildlife monitoring datasets that had been collected for the European Marine Energy Centre (EMEC) wet renewable test sites in Orkney. The wave energy and tidal-stream test sites, two of the first areas in the world for wet renewable devices to be deployed and tested, utilised slightly different approaches to their shore-based wildlife monitoring. These chapters identified relationships between the more frequently observed bird species' site usage and habitat and environmental variables and were crucial in the development of the survey design for my own fieldwork (**Chapter 4**); by reviewing the survey methods used and analysing these two long-term data sets I was able to plan for other challenges. Chapters 2 and 3 also allowed deficiencies in the protocols established for fieldwork at Fall of Warness and Billia Croo to be identified, and improvements to data collection methodologies to be suggested.

Chapter 4 builds on the findings of **Chapters 2 and 3,** focusing on the marine bird usage of a high-energy tidal stream (HETS) in Bluemull Sound, Shetland. I collected observational data from vantage points along the length of the sound over two breeding seasons and one winter period. This chapter has contributed to our understanding of five key diving species usage of a HETS, identifying habitat preferences, foraging patterns and behaviour. However, the results, when compared with findings of other studies show birds utilise different HETS differently. This is likely a consequence of each HETS creating a unique mosaic of habitats and complex hydrodynamics. If so, birds may adapt their foraging approaches to maximise foraging at a local context, meaning results from one study may not necessarily be applicable to other HETS. This strongly indicates the need for site-specific studies to be undertaken for assessment of developments. The reduced usage of the tidal sound by most species during faster current speeds suggests decreased risk of overlap with marine renewables. However, preferences for relatively faster flowing areas by puffin and common guillemot may counter this.

Chapter 5 provides the most extensive and up-to-date review of diving and foraging behaviour for UK marine bird species. I was able to report a number of diving and foraging parameters for most of the species reviewed and through this to identify knowledge gaps including the species and parameters for which there are a paucity of studies. Having identified the diving habits and depths to which marine bird species dive when foraging, the extent to which they are likely to come into conflict with tidal devices can more readily be assessed. For example, razorbill, Atlantic puffin, black guillemot, great cormorant and northern gannet overlap with the optimal depth for a number of the tidal devices, but the depth to which they most frequently dive has reduced overlap. Consequently, this review should aid impact assessments required for marine renewable developments. Furthermore, I also provide many of the values that are required to undertake underwater collision risk modeling.

6.2 Seabird ecology and implications for marine renewable energy

Prior to the development and expansion of the marine renewable energy industry, studies of marine bird usage of high-energy environments, such as tidal-streams had largely been limited to North America (e.g. Hunt et al. 1998; Zamon 2001; 2003; Holm and Burger 2002; Elliott 2004; Ladd et al. 2005), with paucity of understanding of their use in the UK and Europe. Many of these North American studies have been undertaken on closely related species to those that occur around the UK, including species of cormorant, auklet and pigeon guillemot (e.g. Hunt et al. 1998; Holm and Burger 2002; Elliott 2004; Ladd et al. 2005). Often in data poor situations, such as when novel technologies are proposed, policy makers are required to make qualitative judgments on the best available data, sometimes using proxy species (Garthe and Hüppop 2004; Furness et al. 2012). However, given that behavioural and dietary variations are known to occur at a species level even within geographically similar areas between individuals (e.g. Elliot et al. 2008), it is vitally important to consider the local foraging and feeding specialisations to determine vulnerability to encountering marine renewable developments. Further baseline studies were, and are still, required to better understand the ecological importance of these environments.

Each HETS provides a unique system for exploitation by marine predators, with the combination of hydrodynamic features and habitats potentially making prey more abundant, or concentrating prey to make it more accessible (Wolanski and Hamner 1988). Benjamins et al. (2015) suggest that marine megafauna may be attracted to tidal streams because of the enhanced vulnerability of prey to capture. These environments can provide fine-scale temporally and spatially predictable resources, however, due to the tidal nature driving these, features are often localised both temporally and spatially, with features forming and disappearing. Species are likely to exploit these in different ways dependent on their optimal foraging strategies and behaviour. Benjamins et al. (2015) highlights that a multitude of factors are likely to be affecting the distribution of marine predators in these sites, including experience, local bathymetry, current speeds, oceanographic conditions, density, distribution energy content and behaviour of prey, which will interact and lead to a range of behavioural patterns across the tidal cycle between and within different HETS, and between and within the same species. It is important to understand the different behavioural requirements of marine birds using these systems to help inform what the implications might be for interacting with wet renewables. Exactly the same situation applies for marine mammals, where the association with narrow channels and high tidal flow appears to be a significant influence on their distribution and foraging activity (Wilson et al. 1997; Zamon 2001; Hastie et al. 2004).

The two most consistently observed marine bird species in HETS (both in Orkney and Shetland) were black guillemot and European shag, which concurs with findings of Waggitt et al. (2016a) and Wade (2015). Both species are present not only during the breeding season but found to be present year round, due to more sedentary behaviour or reduced post-breeding dispersal (Forrester et al. 2007). This increases their vulnerability and exposure to wet renewables. Both species also do not breed in such dense colonies as other seabird species (Mitchell et al. 2004), and many scattered or clustered nests can be found close to or adjacent to tidal streams. Habitat selection by a colonial animal likely depends on the choices of conspecifics as well as prey conditions within foraging ranges (e.g. Hunt et al. 1986; Birt et al. 1987; Miliniski & Parker 1991), and the extent to which marine bird densities reflect prey distributions over finer spatial scales is at least partially dependent upon prey availability at larger spatial scales (Vliestra 2005). However, species such as black guillemot and shag that do not breed in such dense numbers, and where clustered nests occur along HETS may have a cost-benefit balance to a more energetically costly resource that is closer to home. It would be interesting to further explore the benefits to foraging in a more costly environment, which requires less of a commute, and this may have consequences for proximity of arrays to breeding locations. A further observation in Bluemull Sound was that for both species the numbers of birds observed increased after the breeding season, suggesting that fledglings were utilising the HETS. Immature birds are known to be more vulnerable as they hone their practice and foraging skills (e.g. Burger 1980; Greig et al. 1983; Daunt et al. 2007), therefore they may be more vulnerable than any adults that have been foraging in or around an array, and who may have acquired knowledge of avoiding collisions.

The distributional patterns of animals generally reflect how they meet their energetic requirements for survival (Horne and Schneider 1994). Species that forage within high-speed areas will be at greater risk of either collision or displacement by devices. Chapter 4 suggests that fewer birds are observed at higher current flow speeds, but there could be other factors not fully teased out by analyses, including relationships with local features that aggregate prey close to but not within the high flow itself. Steep velocity gradients in tidal streams may mean that birds briefly enter faster flowing currents to pursue prey before returning to calmer or counterflowing waters (Johnston et al 2005; Benjamins et al. 2015) – patterns of use seen by puffin in Bluemull Sound suggest this could be possible, with areas of very fast and slow areas preferred.

Complex hydrodynamics makes analysis difficult. However, use of ‘habitat variables’ as a proxy for more detailed modelling enables patterns to be identified but does not always enable drivers to be identified. This highlights the difference between what is required for site characterization for EIA and more detailed research that could help inform assessment of risk. Both of these are important as industry is developing at such a rapid pace.

Much of the limited research on seabirds in high tidal flow areas concentrates on foraging individuals, however marine birds may use an area for a multitude of reasons, and those will influence what impacts a wet renewables device or array may have. Foraging behaviour is the primary consideration, as unlike offshore wind, it is likely only to be foraging birds that encounter, and potentially collide with sub-surface devices, such as tidal turbines. Therefore an understanding of this is vital to assess potential collision risk, and loss of foraging resource for implications for displacement. However, areas used by loafing birds for self-maintenance behaviour are also important, particularly near breeding areas such as those identified in recent SPA extensions; therefore disturbance and displacement to loafing birds, particularly in proximity to breeding sites, should also be considered. Finally birds may use HETS, particularly channels, for travel and commuting. While many wet renewable arrays are unlikely to present a barrier to birds in the way that offshore wind farms could do, both wave and tidal arrays, particularly those that are surface piercing may represent barrier to auks dispersing from breeding colonies. I found that Bluemull Sound was used for different purposes by different seabird species. Interestingly, the three species that breed along the sound all varied in the behaviours observed. Black guillemots were observed foraging in less than a quarter of sightings, but in similar proportions along the length of the sound; while two thirds were observed loafing. Shags were observed foraging more than black guillemots, and more so at either end of the sound. This may reflect that shags have different (or stronger) habitat preferences within the sound, but also the differences in foraging strategies, where shags, due to their wettable plumage, tend to only spend short periods of time in the water foraging and then roost on land to dry off (Forrester et al. 2007). Some bird species were consistent in the behaviours observed; for example almost all fulmars were observed loafing, while almost all razorbill when observed on the water surface, as opposed to commuting flocks (which were excluded), were foraging.

Marine birds can utilise HETS as corridors between foraging and breeding locations (Benjamins et al. 2015). My own observations in Bluemull Sound suggest this is particularly true for a number of species, including auks and gannets. Interestingly, razorbill has been identified as a species that is vulnerable to tidal developments (Furness et al. 2012), and was frequently seen flying through the site in mixed auk commuting flocks but was rarely observed on the surface of the water either in site scans or focal observations in Bluemull. The few observations where they were observed stopping was while a mixed forage flock had aggregated. This suggests that for this species, in this location at least, that they were picking up cues for suitable foraging locations, but only chose to stop when there was a guaranteed food source, reducing their likelihood of encountering devices regularly. Other observations for this species, and for common guillemot in Bluemull Sound were of dispersing fathers taking their chicks out to sea, and in some cases the young getting carried by tidal rips across the sound and away from their foraging fathers. Guillemot fathers have been observed altering their diving strategies, including undertaking shorter dives while caring for their young at sea (Linnebjerg, pers comm.) Wet renewables, both wave and tidal, may present an additional hazard to these as they disperse from the colonies first, with regard to adults that are preoccupied with their young; and also any surface piercing devices, such as wave arrays, may be difficult to navigate around and also provide predators such as skuas and gulls additional opportunities to roost while selecting vulnerable young to attack.

6.3 Methodological considerations and limitations

When applying ecological understanding to policy driven questions, it is important to consider the context to ascertain level and depth of data that are required. For example, the difference between more general site characterisation by all receptors for an Environmental Impact Assessment (EIA) and/or Habitats Regulation Appraisal (HRA) vs. post-consent monitoring focused on a research question relating to a group of species for a specific impact. It is important for developers to maximise scientific output from data collected, however at the initial stages there are political and economic constraints that requires a focus on what is most useful to inform their EIA and HRA. Once a development has been consented the post-consent monitoring is likely to shift to answer hypothesis driven questions that have arisen out of the EIA or HRA. While this has the potential to lead to different drivers for research; academics seeking answers to ecological questions and developers seeking answers to political questions, it also has the potential for scientific questions to drive an industry forward. Science can seek to provide pragmatic approaches and this has been one of the key drivers for this applied research; provision of robust data, which can form the baseline to aid development management and policy decisions.

This study has shown the importance of understanding the methodological limitations of surveys in these high-energy environments. By exploring the application of a tidal correction factor to vantage point scans, I have demonstrated that a significant bias exists when surveys do not account for changing current speeds, and that densities used without a correction may be either inflated or deflated (dependent on scan and flow directions). This could have large consequences for any underwater collision risk modelling estimates. I therefore strongly recommend that further work is undertaken to explore the implications of this and improve the methods used to calculate this correction. I also recommend switching the direction of scans (i.e scanning against the current) to reduce the variation and simplify the correction.

This study has been limited in chapters 2, 3, 4 by availability of concurrent hydrodynamic data sets, which has presented some issues in being able to fully tease out the temporal and localized nature of certain features, such as shearlines. This, in part, has also been seen in previous studies, which have largely used qualitative catergorisation of habitats. However, I have clearly been able to demonstrate that robust data can be collected from these HETS, which can shed insight on how they are being used by marine birds. It is hoped that developers can adopt the approaches identified here, which will improve the accuracy of their assessments and improve monitoring of any potential impacts.

6.4 Future research and recommendations

Since this study started in 2010 the importance of understanding the usage of areas suitable for wet renewables by marine megafauna has increased, particularly in the highlands and islands around Scotland (Benjamins et al. 2015). However, these dynamic environments present challenges both in terms of scientific understanding and logistical undertaking of data collection. Studies have found interspecific variation in temporal, spatial patterns and habitat associations, which leads to species-specific and site-specific vulnerability to impacts from renewables. Some of the studies have been able to include more of an ecosystem approach, or utilise methods such as sonar or tags to ascertain behaviour of focal individuals underwater. These are crucial to make sense of monitoring.

Chapter 5 highlighted that there are inconsistencies in both the way diving and foraging parameters are reported and for many studies data are collected but then not published, and therefore are unavailable for use within environmental impact assessments. While many studies are published to answer a specific ecological question there are many more applied policy uses of those results. I therefore strongly recommend that researchers seek to make other parameters that have been estimated from such studies more widely available through supplementary material or collaborative databases. To improve collision risk modelling there is a need to measure parameters such horizontal speeds at depth, proportion of time spent at different depths, and also to explore swim speeds in relation to current speeds.

6.4.1 Additional Analyses

I collected additional data as part of my fieldwork at Bluemull Sound, including focal observations and double-observer data, which due to time constraints and availability of analysis techniques has fallen outside the purview of this thesis. One of the key limitations to vantage point methods at present, is the inability to disentangle decay in detections of birds over distance from any ecological gradients that occur across these sites, e.g. with distance from shoreline (and observer). While I had tried to ensure data collected would enable inclusion of this, by undertaking double-observer surveys in my second field season, the statistical packages are still not quite able to deal with this. Therefore one of the key areas of work to develop going forward would be a spatially-explicit capture recapture analysis that allows for the use of double-observer data in gradient survey design, i.e. the use of multiple vantage points. This would move survey techniques for near-shore renewable developments substantially further forward, enabling more accurate calculation of density estimates.

I had collected over 1000 focal observations over the same time period as the site scans, with some examples of these included in Appendix 5. Preliminary explorations of these data suggest that focal observations enable identification of behavioural adaptations to HETS that may be missed by traditional site scans. By using the observations like tracking data, I have observed black guillemot and European shag drifting downstream with the current, undertaking short flights upstream against the current, before landing and diving; using the water surface acting like a "feeding conveyor belt". This behaviour has been observed by Holm and Burger (2002) in the closely related pigeon guillemot *Cepphus columba*. I suggest this maximises foraging within a smaller area. However, it also has potential implications for turnover rates and estimated abundances of birds within development sites (Searle et al. 2015). I suggest this highlights the potential importance of focal observations in understanding behavioural usage of tidal environments. The next steps include a more detailed analysis of these data, akin to approaches used in tracking. I hope these observations can provide a low-cost alternative to tracking (particularly where tracking may not be suitable) that gives context to site abundance and density estimates, and can give an indication of turnover within development sites. This can lead to a more accurate assessment of the impacts of marine renewable energy developments.

6.5 Final conclusion

These high-energy environments, which are abundant around the Scottish coastline, are highly complex, providing ecologically and physically interesting systems to study. However, they remain relatively poorly studied, but provide an opportunity to see how spatial and temporal heterogeneity affects animal distribution and success. Due to the complexities and uniqueness of such systems, they are quite mysterious in how they function and how ecological and physical processes interact. Therefore these systems provide a challenge for research, industry and policy to assess the ecological implications of marine renewable energy developments. This thesis has tried to address some of these challenges and enablea more robust approach to impact assessments for this industry to move forward, which should allow more informed and ecologically appropriate developments to be consented.

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Tables

Common Eider

Supplementary Table 1.1: The top 5 GLMM models, as selected by AIC values, for modelling common eider use of the Fall of Warness.

Model	Df	AIC
Season + time of day + tide state + depth + habitat type + cloud cover	12	19028
Season + time of day + tide state + depth + habitat type + wind strength + precipitation + cloud cover	14	19029
Season + time of day + tide state + depth + habitat type + wind strength + cloud cover	13	19030
*Season + time of day*tide state + depth + habitat type + wind strength + precipitation + cloud cover*	17	19030
Season + tide state + depth + habitat type + cloud cover	11	19032

Gavia spp.

Supplementary Table 1.2: The top 5 GLMM models, as selected by AIC values, for modelling *Gavia spp.* use of the Fall of Warness.

Model	Df	AIC
Season + time of day + depth + wind strength + tide state	10	1239
Season + depth + wind strength + tide state	9	1242
Season + time of day + depth + habitat type + wind strength +tide state + cloud cover	12	1242
Season + depth + wind strength	7	1243
Season + time of day + depth + habitat type + wind strength +tide state + cloud cover + precipitation	13	1243

Northern Gannet

Supplementary Table 1.3: The top 5 GLMM models, as selected by AIC values, for modelling northern gannet use of the Fall of Warness.

Model	Df	AIC
Season + habitat type + tide state	8	6026
Season + habitat type + tide state + cloud cover	9	6027
Season + habitat type + depth + tide state + cloud cover	10	6028
Season + time of day + habitat type + depth + tide state + cloud cover	11	6030
Season + time of day + habitat type + depth + wind strength + tide state + cloud cover	12	6031

Phalacrocorax spp.

Supplementary Table 1.4: The top 5 GLMM models, as selected by AIC values, for modelling *Phalacrocorax spp.* use of the Fall of Warness.

Model	Df	AIC
*Season + time of day*tide state + depth + habitat type + wind strength + cloud cover*	14	132881
*Season + time of day*tide state + depth + habitat type + wind strength + precipitation + cloud cover*	15	132882
Season + time of day + depth + habitat type + tide state + wind direction + precipitation + cloud cover	17	132920
Season + time of day + depth + habitat type + tide state + wind strength + cloud cover	12	132983
Season + time of day + depth + habitat type + tide state + wind strength + precipitation + cloud cover	13	132984

Common Guillemot

Supplementary Table 1.5: The top 5 GLMM models, as selected by AIC values, for modelling common guillemot use of the Fall of Warness.

Model	Df	AIC
Season + time of day + habitat type + wind direction + tide state + precipitation + cloud cover	16	34975
Season + time of day + depth + habitat type + wind direction + tide state + precipitation + cloud cover	17	34975
Season + time of day + habitat type + tide state + precipitation + wind direction	15	34992
*Season + time of day*tide state + depth + habitat type + wind strength + precipitation + cloud cover*	15	35985
Season + time of day + habitat type + tide state + wind direction	14	35472

Razorbills

Supplementary Table 1.6: The top 5 GLMM models, as selected by AIC values, for modelling razorbills use of the Fall of Warness.

Model	Df	AIC
Season + time of day + depth + habitat type + wind direction + tide state + precipitation + cloud cover	17	750
Season + time of day + habitat type + wind direction + tide state + precipitation + cloud cover	16	751
Season + time of day + habitat type + wind direction + tide state	14	755
Season + time of day + habitat type + wind direction + precipitation	15	757
Season + time of day + depth + wind direction + tide state + precipitation + cloud cover	16	765

Black Guillemot

Supplementary Table 1.7: The top 5 GLMM models, as selected by AIC values, for modelling black guillemots use of the Fall of Warness.

Model	Df	AIC
*Season + time of day*tide state + depth + habitat type + wind strength + precipitation + cloud cover*	15	39720
Season + time of day + tide state + depth + habitat type + wind strength + precipitation + cloud cover	13	39770
Season + time of day + tide state + depth + habitat type + wind strength + cloud cover	12	39771
Season + time of day + tide state + depth + habitat type + wind direction + precipitation + cloud cover	17	39929
Season + time of day + habitat type + tide state + precipitation + wind strength	11	39935

Atlantic Puffin

Supplementary Table 1.8: The top 5 GLMM models, as selected by AIC values, for modelling Atlantic puffin use of the Fall of Warness.

Model	Df	AIC
Season + time of day + depth + habitat type	8	3194
Time of day + depth + habitat type	5	3198
Season + time of day + tide state + depth + habitat type + cloud cover	11	3199
Season + time of day + depth + habitat type + wind strength + tide state + cloud cover	12	3200
Season + time of day + depth + habitat type + wind strength + tide state + precipitation + cloud cover	13	3200

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Tables

Common Eider

Supplementary Table 2.1: The significance of the parametric and smooth terms in the chosen model for common eider use of Billia Croo.

Model: gamm(NUMBER ~ s(JULIANDAY,bs="cc") + oTIDE + oGLAREEXTENT, correlation=corAR1 (form=~1|DAYLAPSE), family=negative.binomial (theta=1.7), gamma=1.4, data=common eider1)

Significance of parametric terms:

	df	F	p-value	Signif.[2]
Tidal State	3	2.897	0.0346	*
Glare Extent	3	2.184	0.0889	

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value	Signif.
s(Julian Day)	3.651	3.651	5.445	0.000427	***

Supplementary Table 2.2: Parameter estimates, standard errors, probability values for the GAMM investigating common eider counts as a function of julian day, tidal state and glare extent.

	Estimate	Std. error	Wald	Pr (>|W|)	Signif.
(Intercept)	1.33284	0.09766	13.648	< 2e-16	***
Tide: Ebb	0.04019	0.10415	0.386	0.69975	
Tide: Low Slack	-0.07691	0.09805	-0.784	0.43313	
Tide: Flood	-0.25417	0.09083	-2.798	0.00531	**
Glare: Slight	0.12659	0.13043	0.971	0.33217	
Glare: Moderate	0.48238	0.19364	2.491	0.01302	*
Glare: Severe	0.04297	0.23743	0.181	0.85644	

R-sq.(adj) = 0.0486 Scale est. = 1.4575 n = 577

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[2] Significance codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ‘ 1

Northern Fulmar

Supplementary Table 2.3: The significance of the parametric and smooth terms in the chosen model for northern fulmar use of Billia Croo.

Model: gamm(NUMBER~s(Long,Lat)+s(JULIANDAY,bs="cc")+s(TIMEHOUR, bs="cs") +Observer, correlation=corAR1(form=~1|DAYLAPSE), family=negative.binomial (theta=1.00078), gamma=1.4,data=northern fulmar1)

Significance of parametric terms:

	df	F	p-value	Signif.
Observer	1	8.07	0.00451	**

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value	Signif.
s(Long,Lat)	11.295	11.295	7.538	2.54E-13	***
s(Julian Day)	6.655	6.655	9.44	3.09E-11	***
s(Timehour)	2.394	2.394	4.405	0.00803	**

Supplementary Table 2.4: Parameter estimates, standard errors, probability values for the GAMM investigating northern fulmar counts as a function of latitude and longitude, julian day, time of day and observer.

	Estimate	Std. error	Wald	Pr (>|W|)	Signif.
(Intercept)	1.506	0.107	14.074	< 2e-16	***
Observer: SW	0.3615	0.1273	2.841	0.00451	**

R-sq.(adj) = 0.0532 Scale est. = 12.819 n = 6407

Northern Gannet

Supplementary Table 2.5: The significance of the parametric and smooth terms in the chosen model for northern gannet use of Billia Croo.

Model: gamm(NUMBER~s(Long,Lat)+s(JULIANDAY,bs="cc")+s(TIMEHOUR, bs="cs") +oGLAREEXTENT, correlation=corAR1(form=~1|DAYLAPSE), family=negative.binomial (theta=1.371285), gamma=1.4, data=northern gannet5k1)

Significance of parametric terms:

	df	F	p-value	Signif.
Glare Extent	3	2.748	0.0414	*

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value	Signif.
s(Long,Lat)	6.767	6.767	11.547	3.73E-14	***
s(Julian Day)	5.475	5.475	12.018	2.00E-12	***
s(Timehour)	2.815	2.815	7.665	6.58E-05	***

Supplementary Table 2.6: Parameter estimates, standard errors, probability values for the GAMM investigating northern gannet counts as a function of latitude and longitude, julian day, time of day and glare extent.

	Estimate	Std. error	Wald	Pr (>|W|)	Signif.
(Intercept)	0.84133	0.06367	13.215	<2e-16	***
Glare: Slight	-0.1514	0.09454	-1.602	0.109	
Glare: Moderate	0.17815	0.12084	1.474	0.141	
Glare: Severe	0.14506	0.14386	1.008	0.313	

R-sq.(adj) = 0.0436 Scale est. = 3.0006 n = 3140

European Shag

Supplementary Table 2.7: The significance of the parametric and smooth terms in the chosen model for European shag use of Billia Croo.

Model: gamm(NUMBER~s(Long,Lat)+s(JULIANDAY,bs="cc")+s(TimetolowHR2, bs="cc")+oGLAREEXTENT+Observer, correlation=corAR1 (form=~1|DAYLAPSE), family=negative.binomial (theta=1), gamma=1.4,data=shag5K1)

Significance of parametric terms:

	df	F	p-value	Signif.
Glare Extent	3	1.592	0.189	
Observer	1	17.928	2.32E-05	***

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value	Signif.
s(Long,Lat)	18.875	18.875	16.828	<2e-16	***
s(Julian Day)	6.484	6.484	25.769	<2e-16	***
s(Time to low tide)	1.652	1.652	3.617	0.0351	***

Supplementary Table 2.8: Parameter estimates, standard errors, probability values for the GAMM investigating European shag counts as a function of latitude and longitude, julian day, time to low tide and glare extent.

	Estimate	Std. error	Wald	Pr (>|W|)	Signif.
(Intercept)	0.86682	0.08829	9.818	< 2e-16	***
Glare: Slight	0.04418	0.09228	0.479	0.6321	
Glare: Moderate	0.03131	0.12476	0.251	0.8018	
Glare: Severe	-0.29794	0.13902	-2.143	0.0321	*
Observer: SW	0.42843	0.10118	4.234	2.32E-05	***

R-sq.(adj) = 0.0481 Scale est. = 5.1604 n = 7498

Great Skua

Supplementary Table 2.9: The significance of the parametric and smooth terms in the chosen model for great skua. use of Billia Croo.

Model: gamm(NUMBER~s(Long,Lat,by=FLOCK)+oGLAREEXTENT+Observer, correlation=corAR1(form=~1|DAYLAPSE), family=negative.binomial(theta=2.557), gamma=1.4,data=bonxie1)

Significance of parametric terms:

	df	F	p-value	Signif.
Glare Extent	3	0.95	0.4157	
Observer	1	6.013	0.0144	*

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value	Signif.
s(Long,Lat): Mixed sp.	21.85	21.85	4.064	1.66E-09	***
s(Long,Lat): Single Sp	2	2	2.341	0.0968	.

Supplementary Table 2.10: Parameter estimates, standard errors, probability values for the GAMM investigating great skua counts as a function of latitude and longitude by flock (mixed or single species), glare extent and observer.

	Estimate	Std. error	Wald	Pr (>|W|)	Signif.
(Intercept)	0.49132	0.08589	5.721	1.44E-08	***
Glare: Slight	-0.08218	0.10691	-0.769	0.4423	
Glare: Moderate	-0.04199	0.13433	-0.313	0.7546	
Glare: Severe	-0.18297	0.15718	-1.164	0.2447	
Observer: SW	-0.20868	0.0851	-2.452	0.0144	*

R-sq.(adj) = 0.211 Scale est. = 1.1054 n = 948

Larus spp.

Supplementary Table 2.11: The significance of the parametric and smooth terms in the chosen model for *Larus spp.* use of Billia Croo.

Model: gamm(NUMBER~s(Long,Lat)+s(JULIANDAY, bs="cc", by=SPECIES) +WIND.STRENGTH+oGLAREEXTENT, correlation=corAR1 (form=~1|DAYLAPSE), family=negative.binomial(theta=1.449),gamma=1.4,data=larus1)

Significance of parametric terms:

	df	F	p-value	Signif.
Wind Strength	1	10.057	0.00157	**
Glare Extent	3	1.301	0.27271	

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value	Signif.
s(Long,Lat)	7.627	7.627	2.425	0.015113	*
s(Julian Day): Common	2.556	2.556	6.517	0.000532	***
s(Julian Day): Great black-backed	4.768	4.768	7.406	1.31E-06	***
s(Julian Day): Herring	3.814	3.814	4.251	0.002446	**
s(Julian Day): Larus.sp	2.08	2.08	9.514	6.26E-05	***

Supplementary Table 2.12: Parameter estimates, standard errors, probability values for the GAMM investigating *Larus spp.* counts as a function of latitude and longitude, julian day by species, wind strength and glare extent.

	Estimate	Std. error	Wald	Pr (>|W|)	Signif.
(Intercept)	1.290865	0.137555	9.384	< 2e-16	***
Wind Strength	-0.131901	0.041593	-3.171	0.00157	**
Glare: Slight	0.001433	0.14669	0.01	0.99221	
Glare: Moderate	-0.308883	0.196107	-1.575	0.11557	
Glare: Severe	-0.240856	0.23575	-1.022	0.3072	

R-sq.(adj) = 0.167 Scale est. = 2.6817 n = 992

Black-legged Kittiwake

Supplementary Table 2.13: The significance of the parametric and smooth terms in the chosen model for black-legged kittiwakes use of Billia Croo.

Model: gamm(NUMBER~s(Long,Lat)+s(JULIANDAY,bs="cc")+ s(TimetolowHR2,bs="cc")+WINDIR2+oGLAREEXTENT+Observer, correlation= corAR1(form=~1|DAYLAPSE), family=negative.binomial (theta=1), gamma=1.4, data=blki1)

Significance of parametric terms:

	df	F	p-value	Signif.
Wind Direction	4	2.753	0.02811	*
Glare Extent	3	3.058	0.02847	*
Observer	1	7.63	0.00606	**

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value	Signif.
s(Long,Lat)	2	2	26.732	1.74E-11	***
s(Julian Day)	6.026	6.026	51.517	< 2e-16	***
s(Time to Low Tide)	3.639	3.639	3.015	0.0219	*

Supplementary Table 2.14: Parameter estimates, standard errors, probability values for the GAMM investigating black-legged kittiwake counts as a function of latitude and longitude, julian day, time from low tide, wind direction, glare extent and observer.

	Estimate	Std. error	Wald	Pr (>|W|)	Signif.
(Intercept)	1.61899	0.31459	5.146	4.58E-07	***
Wind Direction: North	0.09936	0.3108	0.32	0.7494	
Wind Direction: None	-0.02742	1.43873	-0.019	0.98481	
Wind Direction: South	-0.19124	0.30736	-0.622	0.53423	
Wind Direction: West	-0.71637	0.36811	-1.946	0.05249	.
Glare: Slight	-0.40584	0.20683	-1.962	0.05058	.
Glare: Moderate	-0.66525	0.23233	-2.863	0.00446	**
Glare: Severe	-0.13907	0.25877	-0.537	0.59135	
Observer: SW	0.49507	0.17923	2.762	0.00606	**

R-sq.(adj) = 0.0268 Scale est. = 1.2993 n = 350

Arctic Tern

Supplementary Table 2.15: The significance of the parametric and smooth terms in the chosen model for Arctic tern use of Billia Croo.

Model: gamm(NUMBER~s(JULIANDAY,bs="cs")+s(TIMEHOUR)+WINDIR2+
oGLAREEXTENT +Observer, correlation=corAR1(form=~1|DAYLAPSE),
family=negative.binomial(theta=1), gamma=1.4,data=atern1)

Significance of parametric terms:

	df	F	p-value	Signif.
Wind Direction	3	7.498	0.000172	***
Glare Extent	3	4.497	0.005709	**
Observer	1	22.179	1.01E-05	***

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value	Signif.
s(Julian Day)	1.866	1.866	29.53	5.88E-10	***
s(Time of Day)	5.103	5.103	6.768	2.25E-05	***

Supplementary Table 2.16: Parameter estimates, standard errors, probability values for the GAMM investigating Arctic tern counts as a function of julian day, time of day, wind direction, glare extent and observer.

	Estimate	Std. error	Wald	Pr (>|W|)	Signif.
(Intercept)	1.262311	0.800656	1.577	0.118785	
Wind Direction: North	-0.002141	0.844206	-0.003	0.997983	
Wind Direction: South	0.81901	0.809466	1.012	0.31465	
Wind Direction: West	-0.806692	0.875962	-0.921	0.359825	
Glare: Slight	-1.385732	0.382923	-3.619	0.000514	***
Glare: Moderate	-0.274654	0.375514	-0.731	0.466639	
Glare: Severe	-0.146863	0.361627	-0.406	0.685727	
Observer: SW	1.460506	0.310121	4.709	1.01E-05	***

R-sq.(adj) = -0.133 Scale est. = 0.9389 n = 96

Common Guillemot

Supplementary Table 2.17: The significance of the parametric and smooth terms in the chosen model for common guillemot use of Billia Croo.

Model: gamm(NUMBER~s(JULIANDAY,bs="cc")+oGLAREEXTENT, correlation=corAR1(form=~1|DAYLAPSE),family=negative.binomial(theta=1),gamma=1.4,data =guill5k1)

Significance of parametric terms:

	df	F	p-value	Signif.
Glare Extent	3	2.545	0.0543	.

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value	Signif.
s(Julian Day)	4.614	4.614	7.553	1.10E-06	***

Supplementary Table 2.18: Parameter estimates, standard errors, probability values for the GAMM investigating common guillemot counts as a function of julian day and glare extent.

	Estimate	Std. error	Wald	Pr (>|W|)	Signif.
(Intercept)	0.7728	0.0739	10.457	<2e-16	***
Glare: Slight	-0.0544	0.1184	-0.459	0.6459	
Glare: Moderate	-0.1908	0.1474	-1.294	0.1957	
Glare: Severe	-0.3893	0.1667	-2.335	0.0196	*

R-sq.(adj) = 0.00748 Scale est. = 5.9826 n = 4472

Razorbill

Supplementary Table 2.19: The significance of the parametric and smooth terms in the chosen model for razorbill use of Billia Croo.

Model: gamm(NUMBER~s(Long,Lat)+s(JULIANDAY,bs="cc"), correlation=corAR1 (form=~1|DAYLAPSE), family=negative.binomial(theta=5.55), gamma=1.4, data=razor1)

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value	Signif.
s(Long,Lat)	2	2	7.249	0.00094	***
s(Julian Day)	2.344	2.344	4.176	0.01235	*

Supplementary Table 2.20: Parameter estimates, standard errors, probability values for the GAMM investigating razorbill counts as a function of latitude and longitude and julian day.

	Estimate	Std. Error	t	value	Pr(>|t|)	Signif.
(Intercept)	0.64343	0.07895		8.15	6.14E-14	***

R-sq.(adj) = 0.104 Scale est. = 1.2898 n = 184

Black Guillemot

Supplementary Table 2.21: The significance of the parametric and smooth terms in the chosen model for black guillemot use of Billia Croo.

Model: gamm(NUMBER~s(Long,Lat)+s(JULIANDAY,bs="cc")+s(TIMEHOUR, bs="cs")+WIND.STRENGTH, correlation=corAR1(form=~1|DAYLAPSE), family=negative.binomial(theta=1), gamma=9.99,data=tystie1)

Significance of parametric terms:

	df	F	p-value	Signif.
Wind Strength	1	3.68	0.0552	.

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value	Signif.
s(Long,Lat)	16.849	16.849	3.048	2.82E-05	***
s(Julian Day)	5.14	5.14	13.555	2.59E-13	***
s(Timehour)	4.926	4.926	15.137	1.99E-14	***

Supplementary Table 2.22: Parameter estimates, standard errors, probability values for the GAMM investigating black guillemot counts as a function of latitude and longitude, julian day, time of day and glare extent.

	Estimate	Std. error	Wald	Pr (>|W|)	Signif.
(Intercept)	0.40158	0.03598	11.162	<2e-16	***
Wind Strength	-0.02171	0.01132	-1.918	0.0552	.

R-sq.(adj) = 0.11 Scale est. = 0.25557 n = 1922

Atlantic Puffin

Supplementary Table 2.23: The significance of the parametric and smooth terms in the chosen model for Atlantic puffin use of Billia Croo.

Model: gamm(NUMBER~s(Long,Lat)+s(JULIANDAY,bs="cc")+s(TIMEHOUR, bs="cs")+s(TimetolowHR2,bs="cc")+oGLAREEXTENT+Observer, correlation= corAR1(form=~1|DAYLAPSE), family=negative.binomial(theta=9.99), gamma=1.4, data=Atlantic puffin1)

Significance of parametric terms:

	df	F	p-value	Signif.
Glare Extent	3	2.748	0.0414	*

Approximate significance of smooth terms:

	edf	Ref.df	F	p-value	Signif.
s(Long,Lat)	4.241	4.241	2.172	0.0673	.
s(Julian Day)	2.03	2.03	3.758	0.0235	*
s(Timehour)	1.622	1.622	4.639	0.0156	*
S(TimetoLowTide)	1.259	1.259	1.887	0.1673	

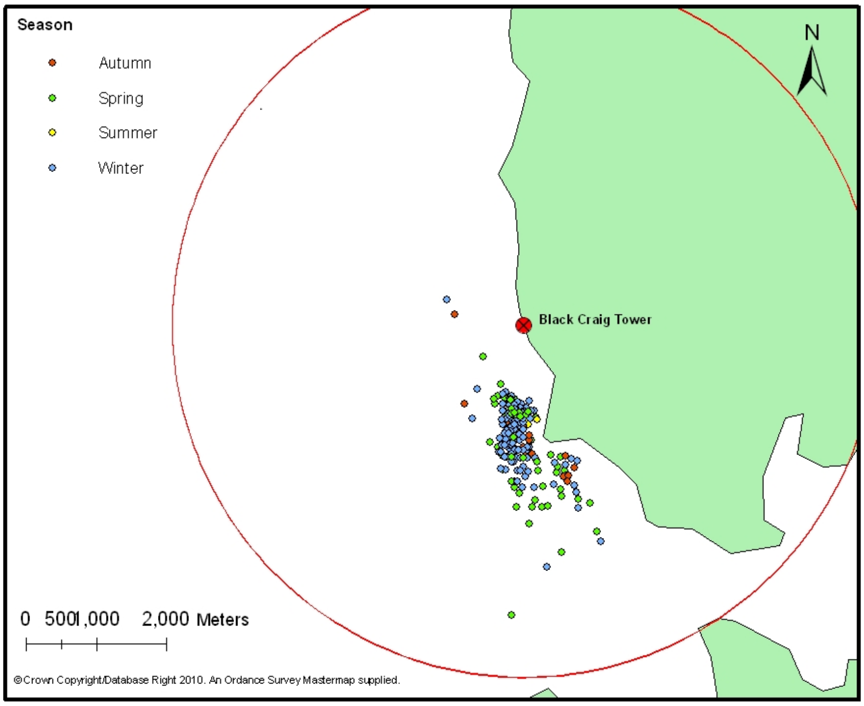
Supplementary Table 2.24: Parameter estimates, standard errors, probability values for the GAMM investigating Atlantic puffin counts as a function of latitude and longitude, julian day, time of day, time to low tide, glare extent and observer.

	Estimate	Std. error	Wald	Pr (>|W|)	Signif.
(Intercept)	0.47285	0.0849	5.569	4.45E-08	***
Glare: Slight	0.12458	0.12174	1.023	0.30673	
Glare: Moderate	0.38075	0.14265	2.669	0.00789	**
Glare: Severe	0.19346	0.15378	1.258	0.20904	
Observer: SW	-0.21008	0.09127	-2.302	0.02182	*

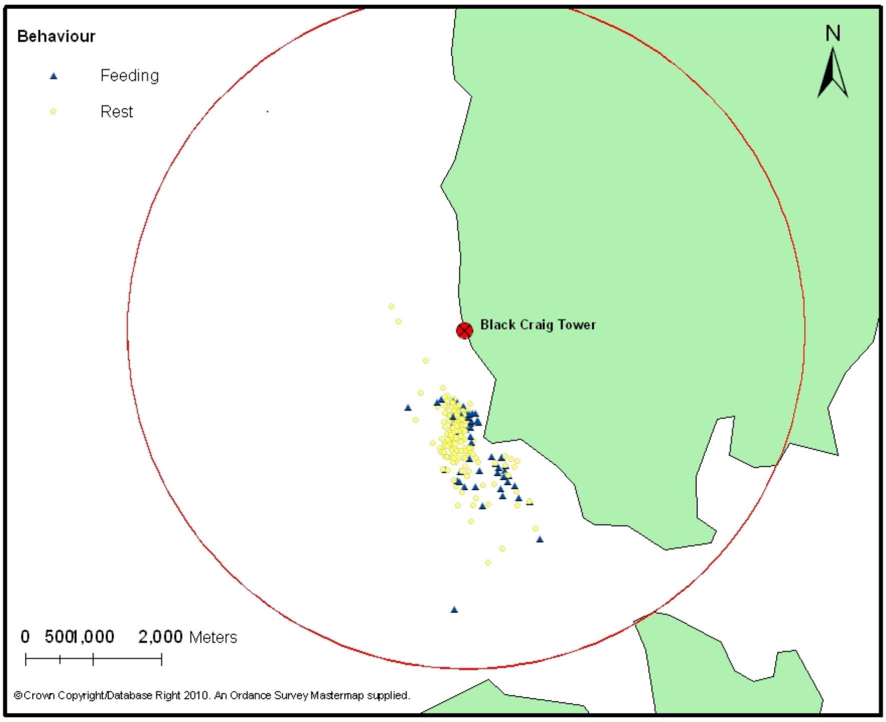
R-sq.(adj) = 0.0697 Scale est. = 0.78303 n = 455

Distribution Maps

Common Eider

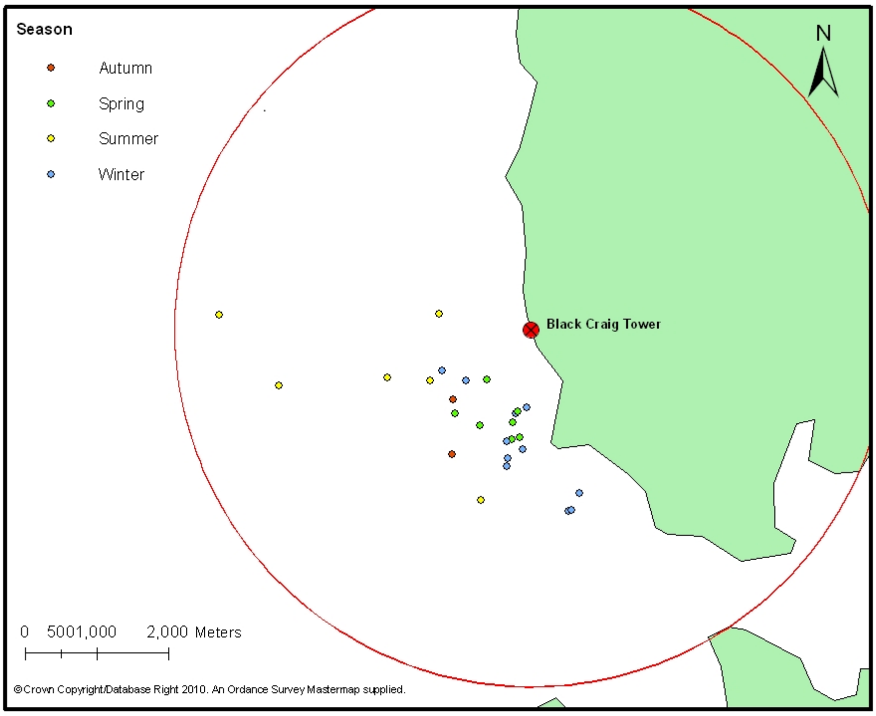


Supplementary Figure 2.1: Map showing the seasonal distribution of Common Eider observations at Billia Croo



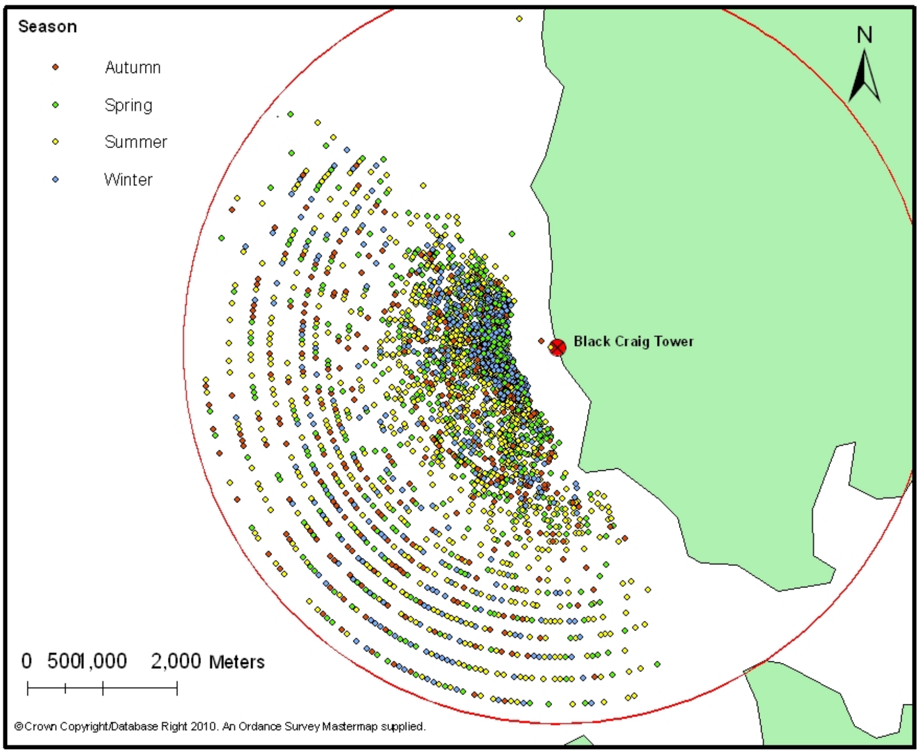
Supplementary Figure 2.2: Map showing the distribution of observed feeding and resting Common Eider at Billia Croo

Red-throated Diver

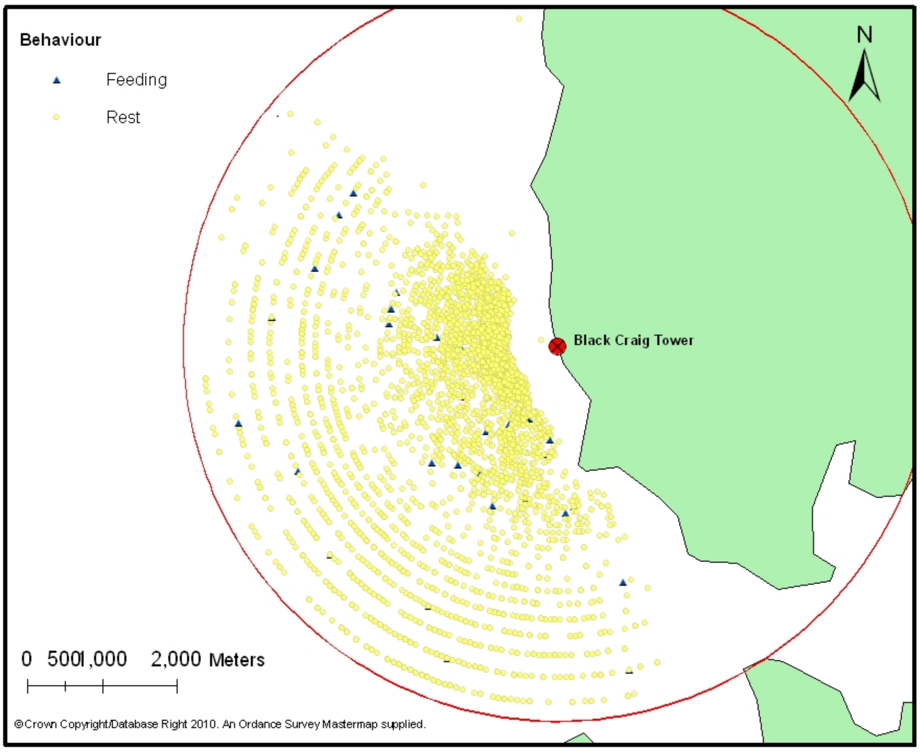


Supplementary Figure 2.3: Map showing the seasonal distribution of Red-throated Diver observations at Billia Croo

Northern Fulmar

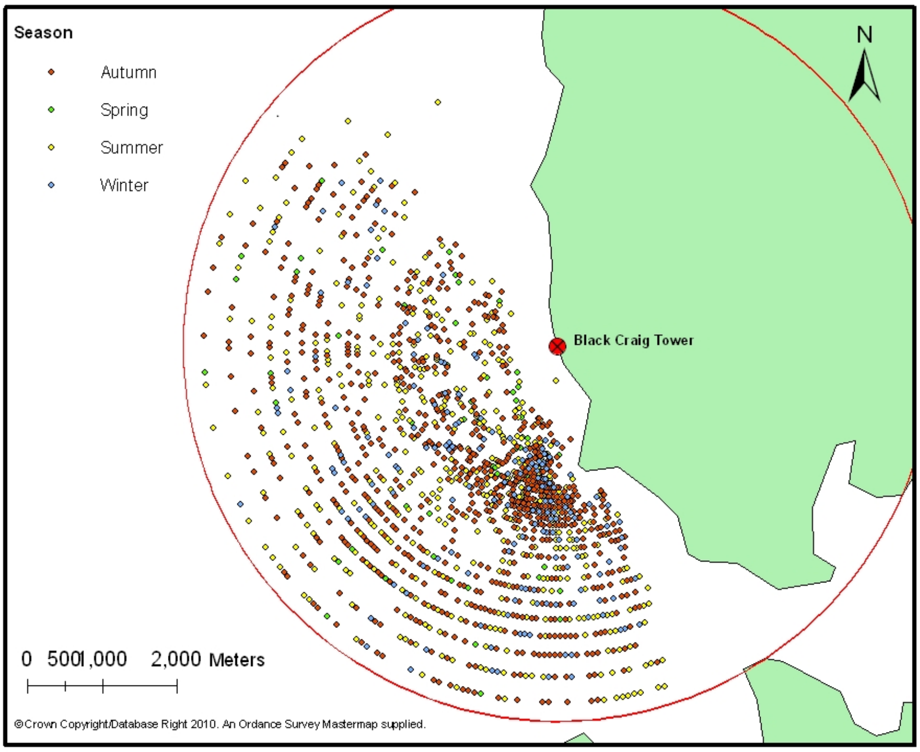


Supplementary Figure 2.4: Map showing the seasonal distribution of Northern Fulmar observations at Billia Croo

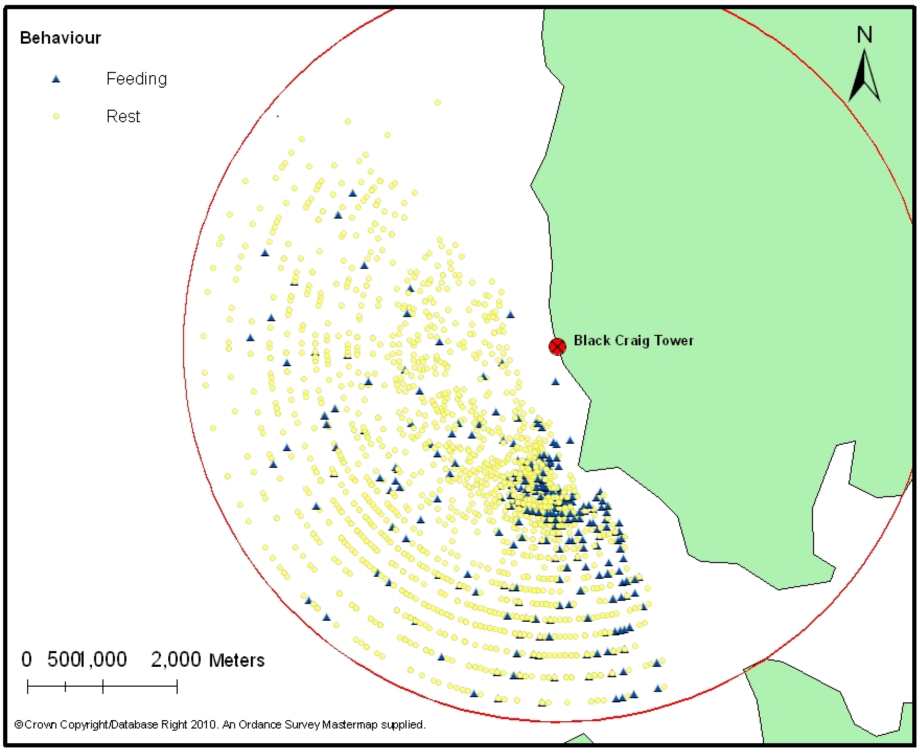


Supplementary Figure 2.5: Map showing the distribution of observed feeding and resting Fulmar at Billia Croo

Northern Gannet

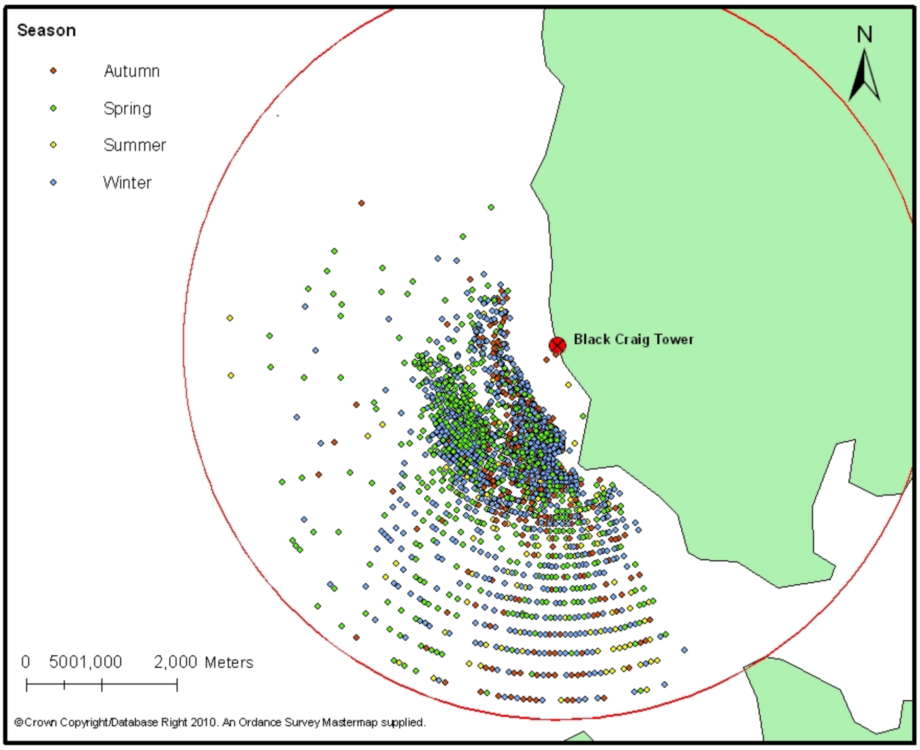


Supplementary Figure 2.6: Map showing the seasonal distribution of Northern Gannet observations at Billia Croo

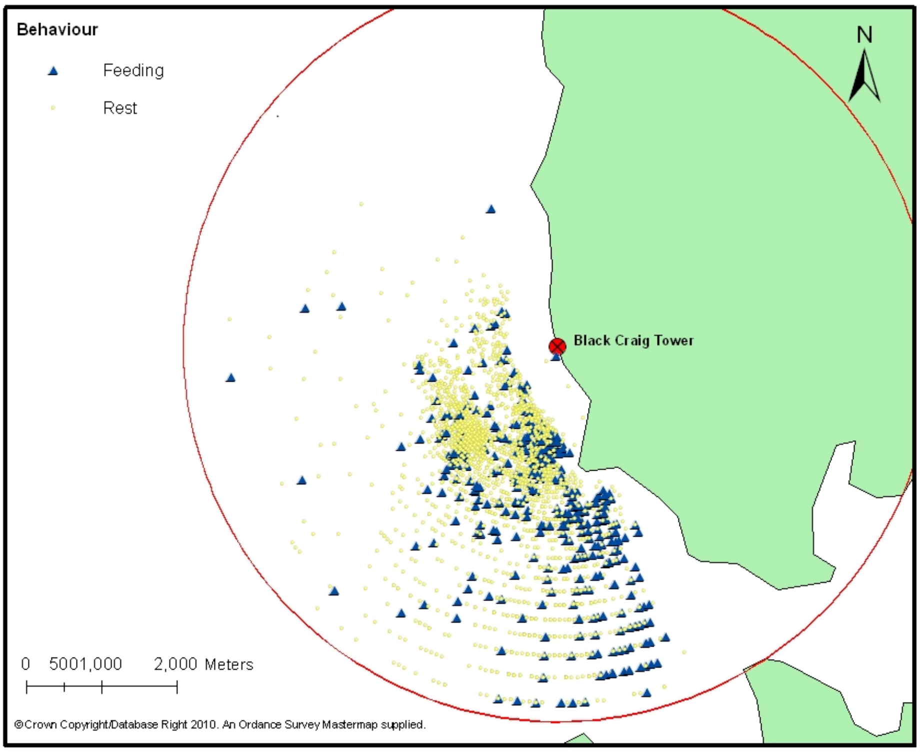


Supplementary Figure 2.7: Map showing the distribution of observed feeding and resting Northern Gannet at Billia Croo

European Shag

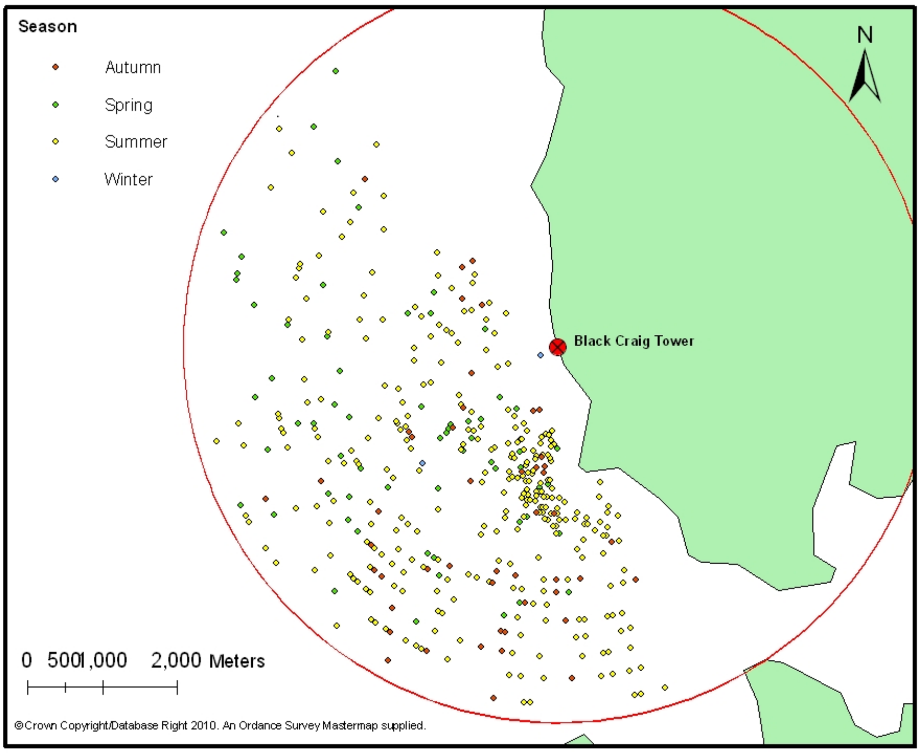


Supplementary Figure 2.8: Map showing the seasonal distribution of European Shag observations at Billia Croo

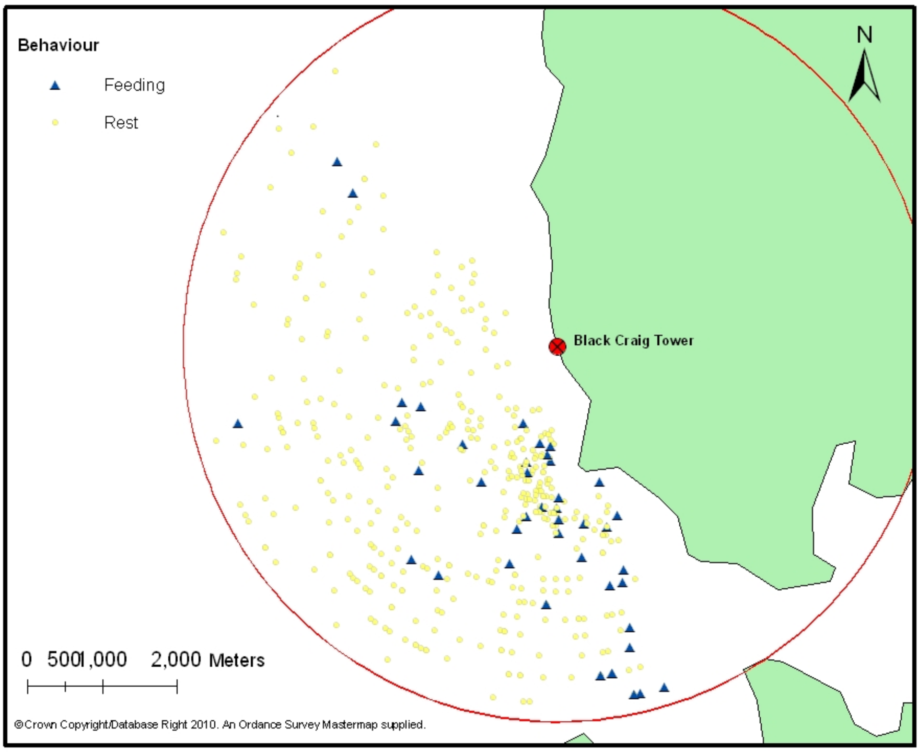


Supplementary Figure 2.9: Map showing the distribution of observed feeding and resting European Shag at Billia Croo

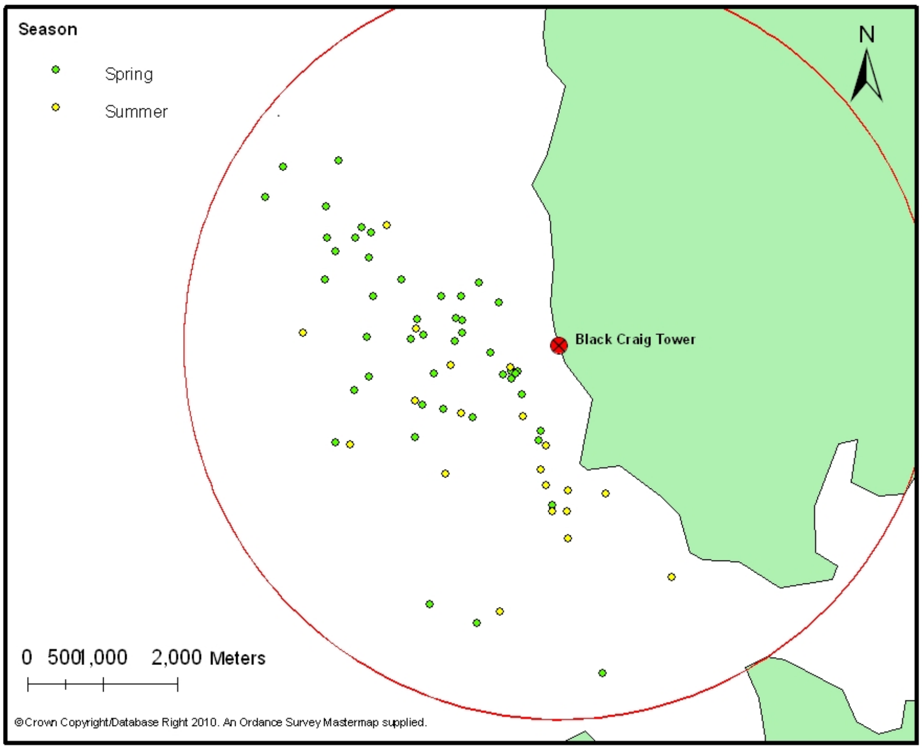
Skua *spp.*



Supplementary Figure 2.10: Map showing the seasonal distribution of Great Skua observations at Billia Croo

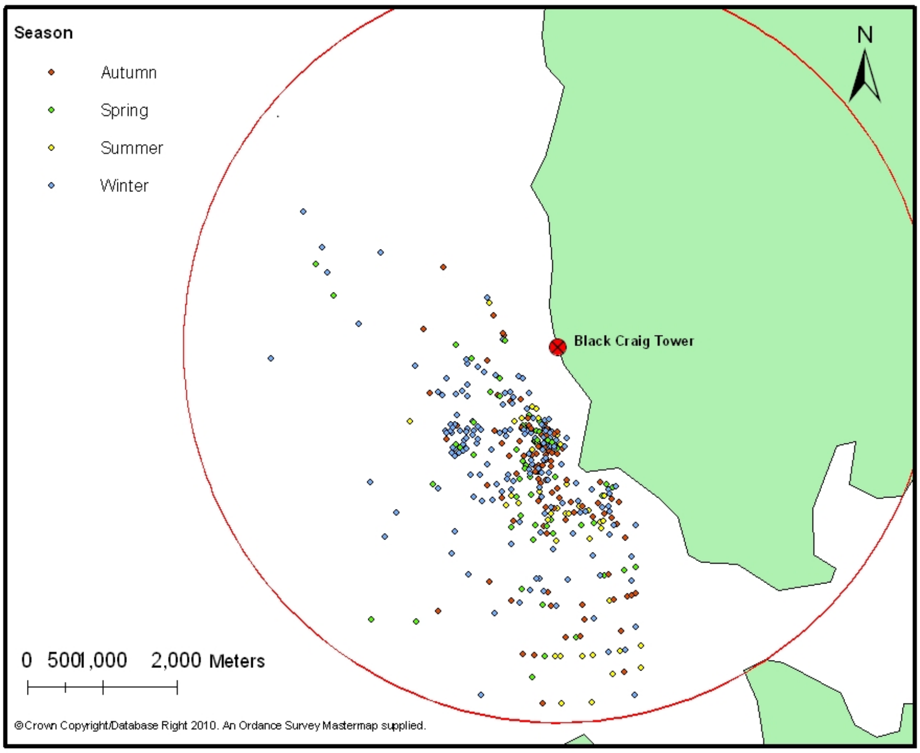


Supplementary Figure 2.11: Map showing the distribution of observed feeding and resting Great Skua at Billia Croo

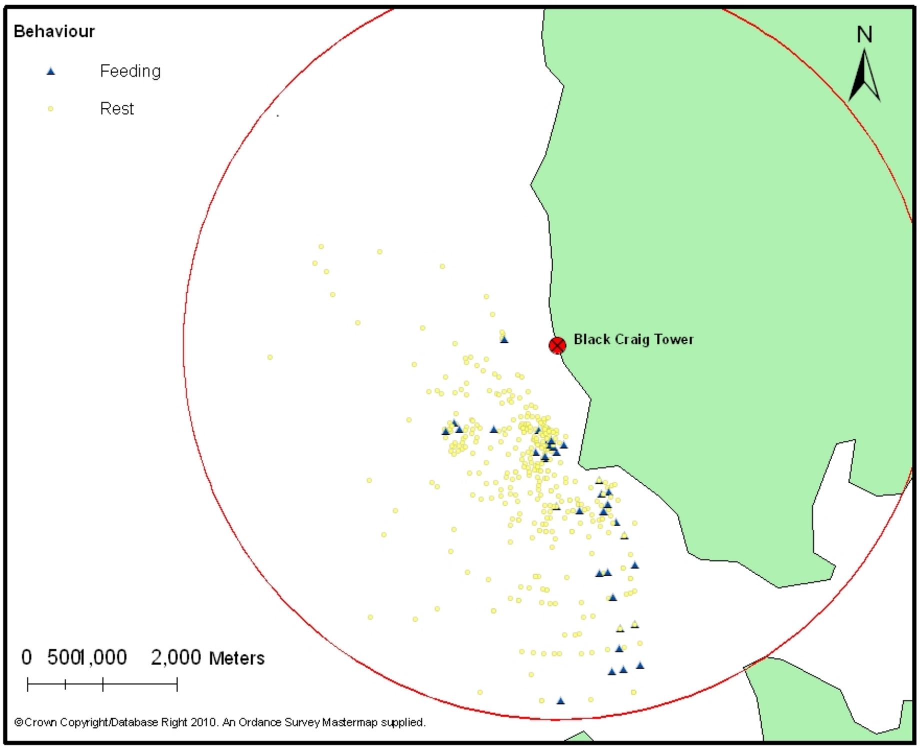


Supplementary Figure 2.12: Map showing the seasonal distribution of Arctic Skua observations at Billia Croo

Gulls *Larus spp*.

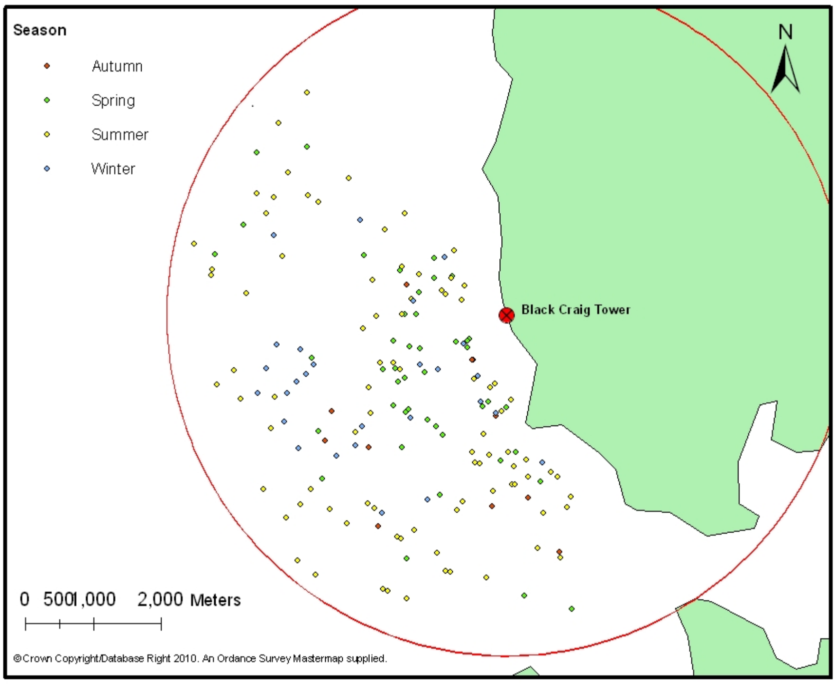


Supplementary Figure 2.13: Map showing the seasonal distribution of *Larus spp*. observations at Billia Croo



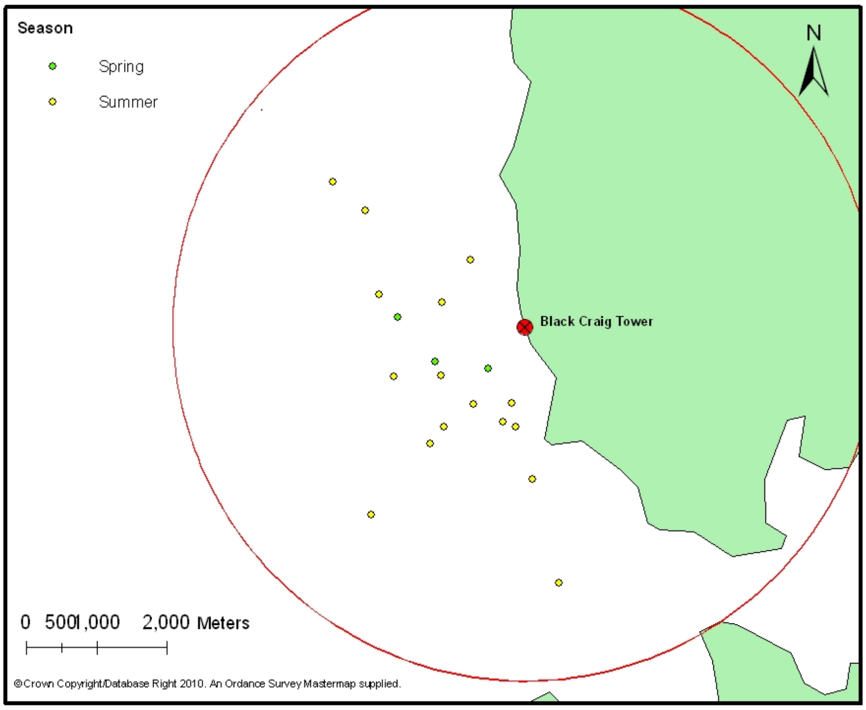
Supplementary Figure 2.14: Map showing the distribution of observed feeding and resting *Larus spp.* at Billia Croo

Black-legged Kittiwake



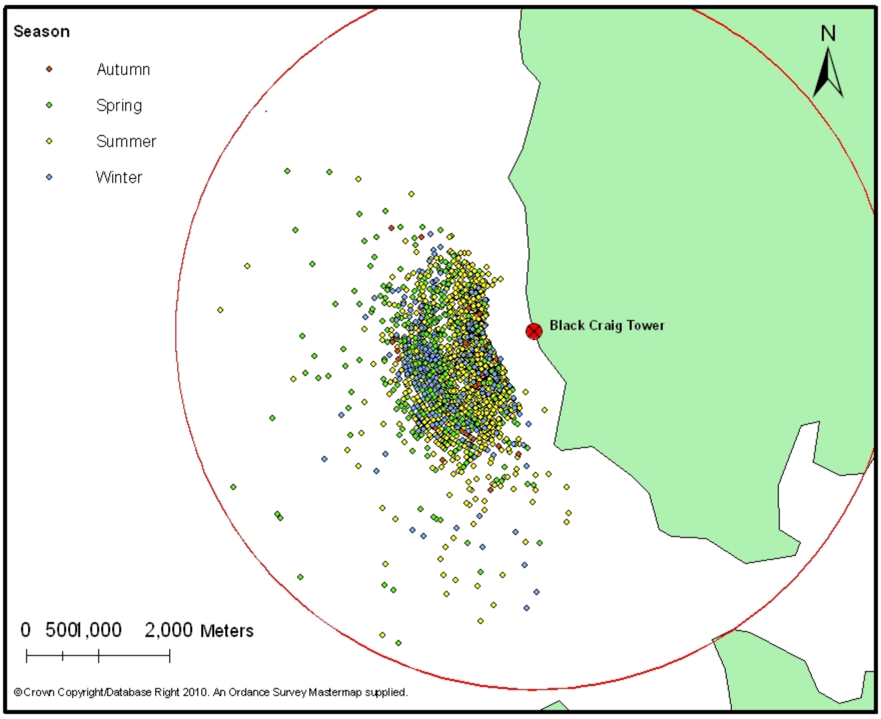
Supplementary Figure 2.15: Map showing the seasonal distribution of Black-legged Kittiwake observations at Billia Croo

Arctic Tern

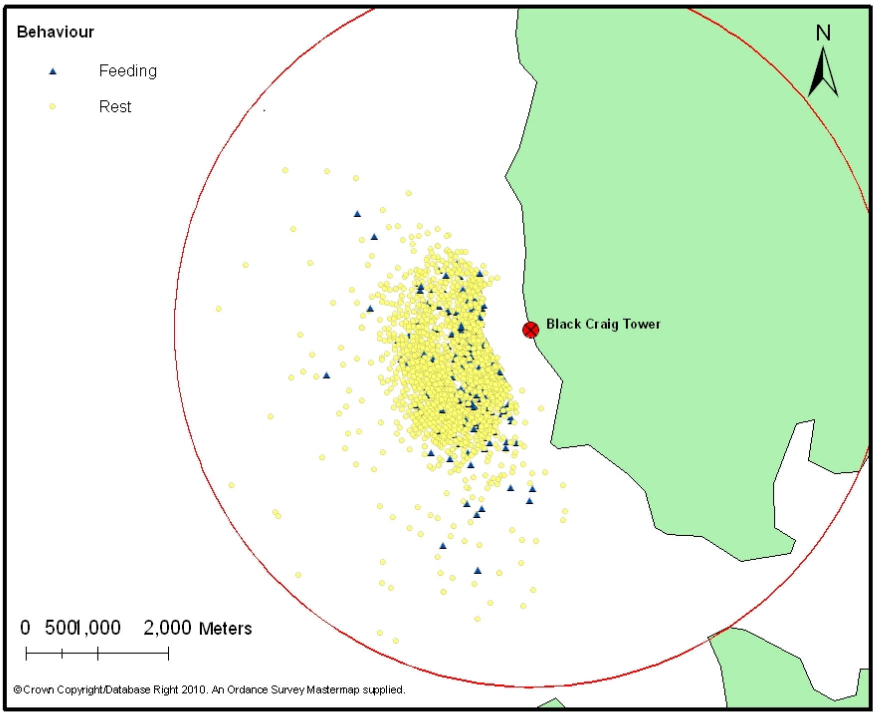


Supplementary Figure 2.16: Map showing the seasonal distribution of Arctic Tern observations at Billia Croo

Common Guillemot

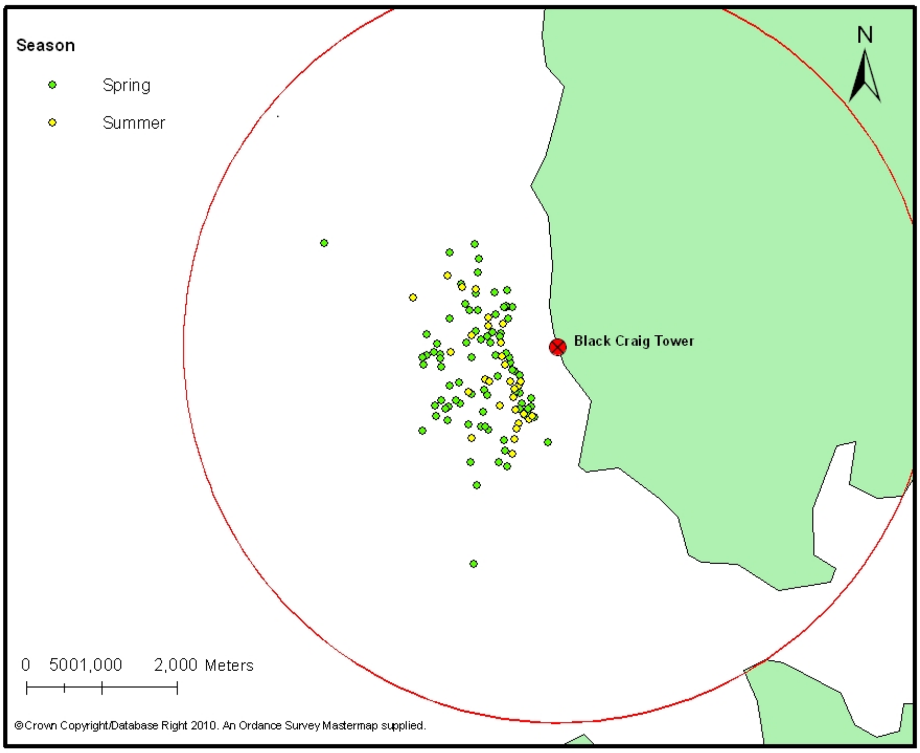


Supplementary Figure 2.17: Map showing the seasonal distribution of Common Guillemot observations at Billia Croo

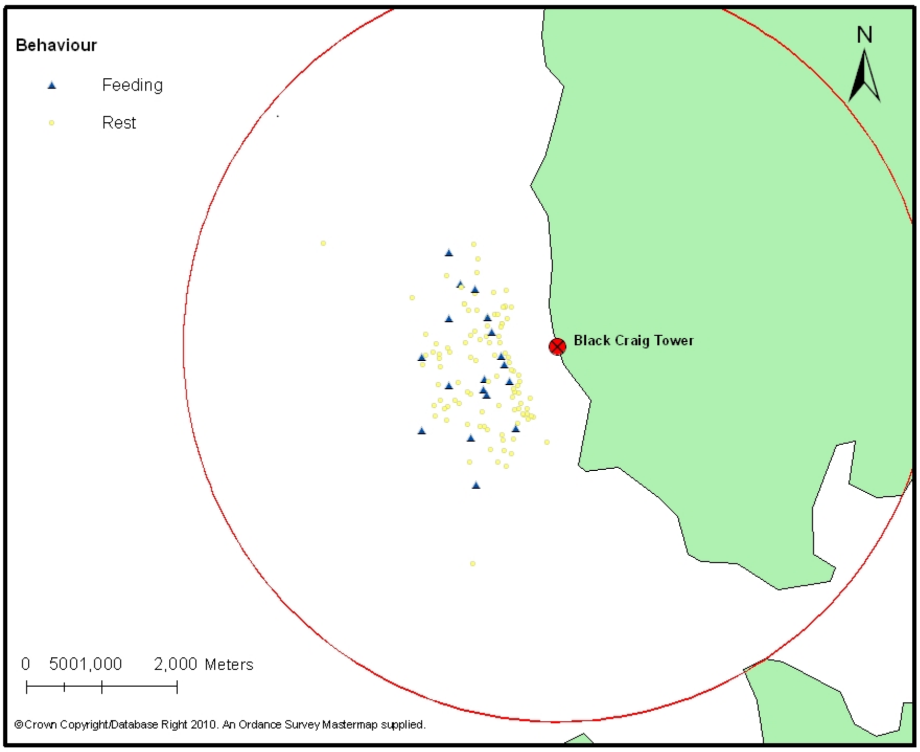


Supplementary Figure 2.18: Map showing the distribution of observed feeding and resting Common Guillemot at Billia Croo

Razorbill

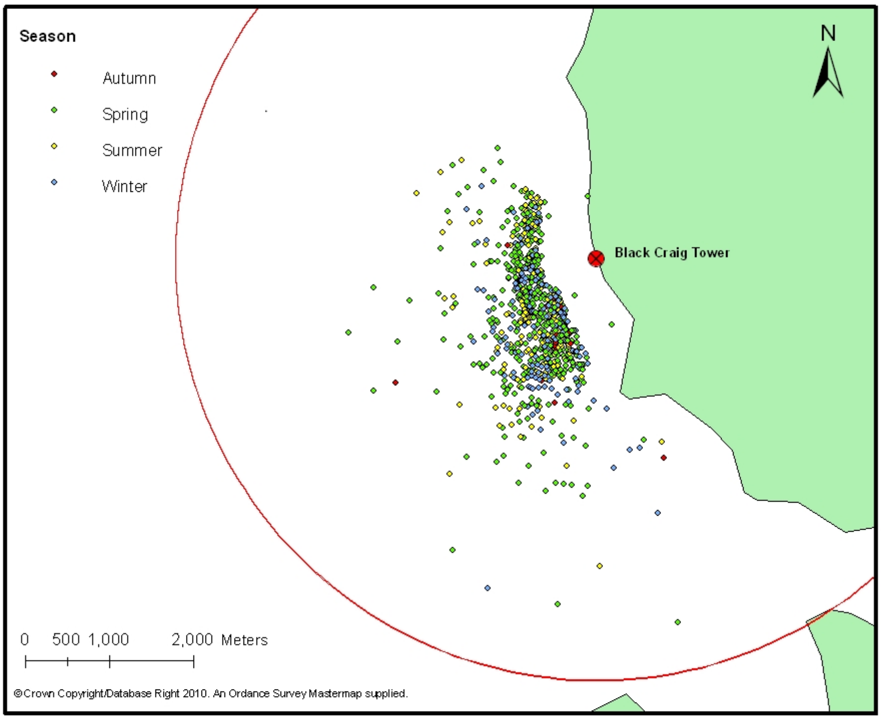


Supplementary Figure 2.19: Map showing the seasonal distribution of Razorbill observations at Billia Croo

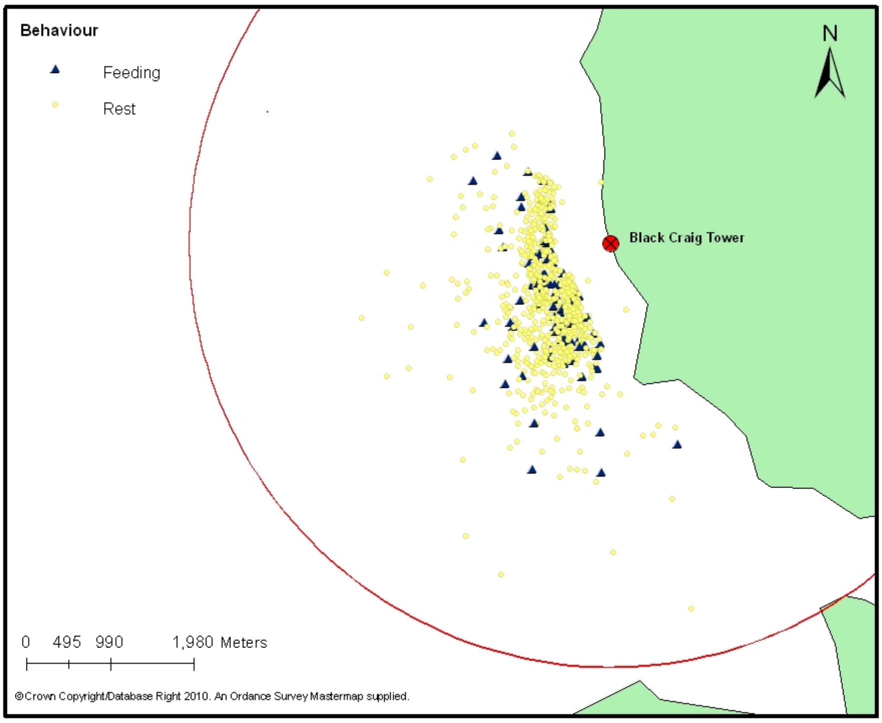


Supplementary Figure 2.20: Map showing the distribution of observed feeding and resting Razorbill at Billia Croo

Black Guillemot

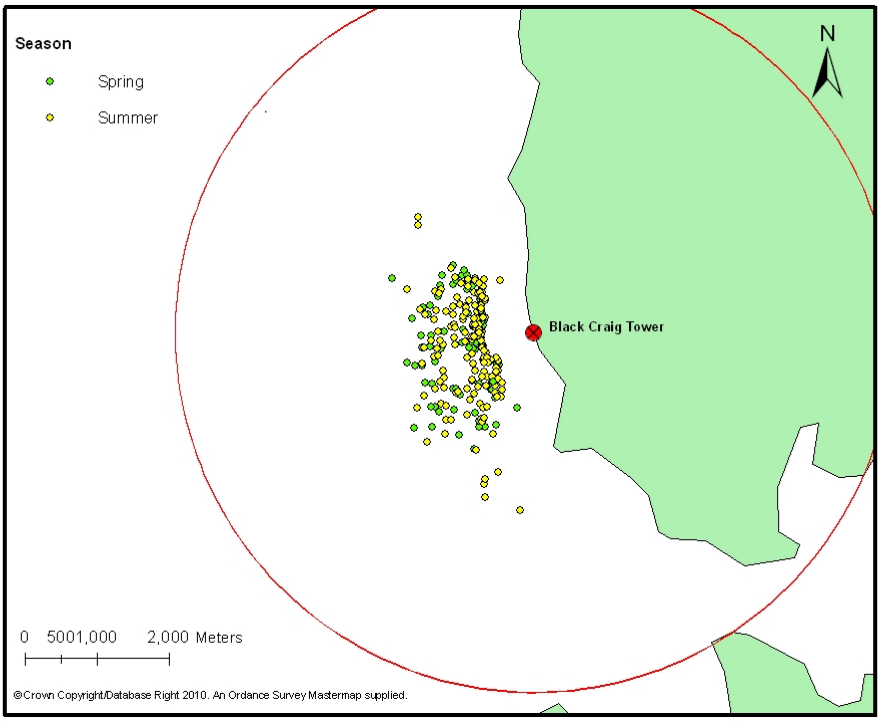


Supplementary Figure 2.21: Map showing the seasonal distribution of Black Guillemot observations at Billia Croo

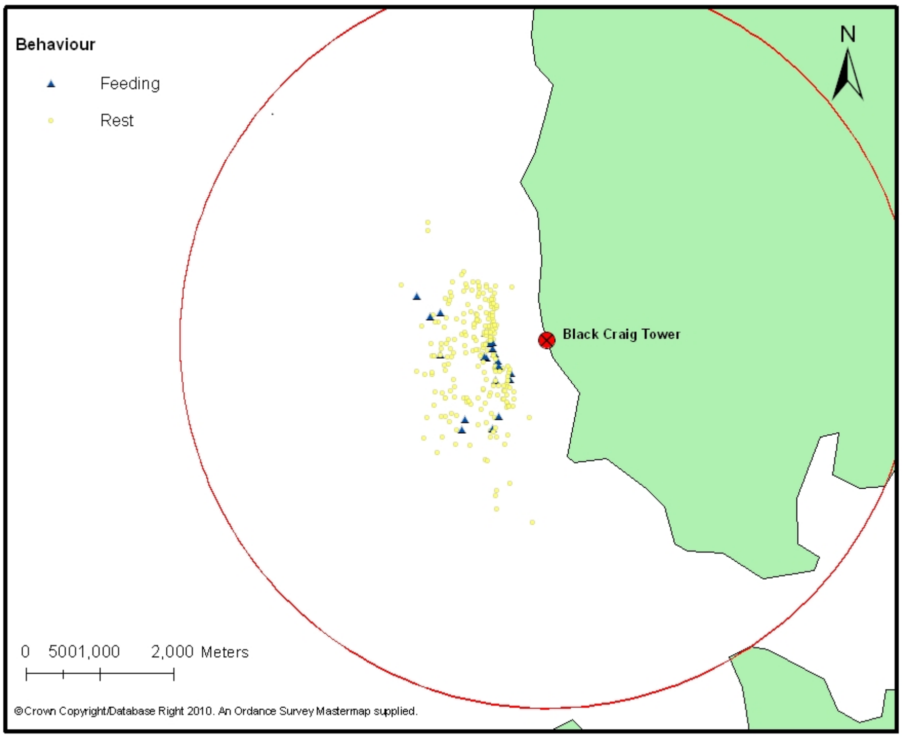


Supplementary Figure 2.22: Map showing the distribution of observed feeding and resting Black Guillemot at Billia Croo

Atlantic Puffin



Supplementary Figure 2.23: Map showing the seasonal distribution of Atlantic Puffin observations at Billia Croo



Supplementary Figure 2.24: Map showing the distribution of observed feeding and resting Atlantic Puffin at Billia Croo

Appendix 3: Chapter 4 Supplementary Material

Contents:

Model Outputs

Black Guillemot

```
Family: quasipoisson
Link function: log

Formula:
No ~ s(CSpd.min, k = 5) + s(fScanID, bs = "re")

Parametric coefficients:
                    Estimate     Std. Error  t value    Pr(>|t|)
(Intercept)         -0.32976   0.07473      -4.413     1.11e-05 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
                    Edf          Ref.       df         F  p-value
s(CSpd.min)         1.005        1.005      6.05       6.42e-05 ***
s(fScanID)          215.990      230.000    17.73      < 2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.117   Deviance explained =   79%
GCV = 0.36403  Scale est. = 0.3871    n = 1526
Cross Validation: R = 0.09190482
```

European Shag

```
Family: quasipoisson
Link function: log

Formula:
No ~ s(Month, k = 5) + oSpeed + Beh + Slope + Bathy + s(CSpd.min,
   k = 5) + s(fScanID, bs = "re")

Parametric coefficients:
                    Estimate      Std. Error  t value   Pr(>|t|)
(Intercept)         -0.022706   0.166780     -0.136    0.89176
oSpeed.L            -0.719016   0.256099     -2.808    0.00518 **
oSpeed.Q            -0.149564   0.220468     -0.678    0.49783
oSpeed.C            -0.288061   0.155877     -1.848    0.06518 .
oSpeed^4             0.012632   0.098294     0.129     0.89779
BehLoafing          -0.397346   0.079038     -5.027    6.88e-07 ***
BehTravel           -0.449368   0.111629     -4.026    6.54e-05 ***
SlopeSlope          -0.255705   0.100283     -2.550     0.01107 *
Bathy               -0.006478   0.003705     -1.748    0.08098 .
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
```

```
Approximate significance of smooth terms:
                  edf       Ref.df    F        p-value
s(Month)          1.899     1.991     9.159    0.000173 ***
s(CSpd.min)       3.682     3.746     6.995    7.75e-05 ***
s(fScanID)        135.730   208.000   3.036    < 2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.259   Deviance explained = 69.8%
GCV = 1.0136  Scale est. = 0.83336   n = 665
Cross Validation R = 0.22328539
```

Atlantic Puffin

```
Family: quasipoisson
Link function: log

Formula:
No ~ s(Month, k = 5) + oSpeed + s(CSpd.min, k = 5) + s(fScanID,
   bs = "re")

Parametric coefficients:
             Estimate   Std. Error   t value   Pr(>|t|)
(Intercept)       -2.60072  0.20343    -12.784  < 2e-16 ***
oSpeed.L          0.03075      0.27918  0.110    0.91242
oSpeed.Q          -0.06012  0.24270    -0.248   0.80463
oSpeed.C          0.51001      0.15481  3.294    0.00117 **
oSpeed^4          -0.09678  0.09573    -1.011   0.31328
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
                  edf       Ref.df    F        p-value
s(Month)          1.001     1.001     8.451    0.00406 **
s(CSpd.min)       1.000     1.000     22.395   4.17e-06 ***
s(fScanID)        101.773   111.000   32.674   < 2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.172   Deviance explained = 97.9%
GCV = 0.77833  Scale est. = 0.54194   n = 302
Cross Validation R = 0.07653993
```

Northern Gannet

```
Family: quasipoisson
Link function: log

Formula:
No ~ oLinear + Beh + s(Bathy, k = 5) + s(CSpd.min, k = 5) + s(fScanID,
   bs = "re")

Parametric coefficients:
                     Estimate    Std. Error  t value   Pr(>|t|)
(Intercept)          -0.47826    0.27475     -1.741    0.083686 .
oLinear.L            2.14875     0.56225     3.822     0.000190 ***
oLinear.Q            1.55636     0.40078     3.883     0.000151 ***
oLinear.C            -0.17486    0.58824     -0.297    0.766660
oLinear^4            -0.08142    0.67503     -0.121    0.904146
oLinear^5            -0.60508    0.53875     -1.123    0.263102
oLinear^6            0.22505     0.45097     0.499     0.618451
BehLoafing           0.53203     0.18869     2.820     0.005427 **
BehOther             0.36503     0.40805     0.895     0.372389
BehTravel            0.18397     0.27714     0.664     0.507779
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Approximate significance of smooth terms:
                     edf         Ref.df      F         p-value
s(Bathy)             2.958       3.339       2.772     0.0392 *
s(CSpd.min)          1.000       1.000       5.373     0.0217 *
s(fScanID)           54.673      90.000      2.921     <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  0.214   Deviance explained = 68.4%
GCV = 1.4453  Scale est. = 1.0325    n = 226
Cross Validation R = 0.2882558
```

Common Guillemot

```
Family: quasipoisson
Link function: log

Formula:
No ~ oSpeed + Beh + s(Month, k = 5) + s(CSpd.min, k = 5) + s(ToD,
   k = 5) + s(fScanID, bs = "re")

Parametric coefficients:
                     Estimate    Std. Error  t value   Pr(>|t|)
(Intercept)          -0.4340     0.1595      -2.721    0.00794 **
oSpeed.L             0.3183      0.1937      1.643     0.10415
oSpeed.Q             0.2251      0.1519      1.482     0.14224
oSpeed.C             0.2427      0.1315      1.846     0.06839 .
BehLoafing           -0.3316     0.1540      -2.153    0.03419 *
BehTravel            -0.4423     0.1936      -2.285    0.02489 *
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
```

```
Approximate significance of smooth terms:
                      edf        Ref.df    F         p-value
s(Month)              1.00       1         8.796     0.003918 **
s(CSpd.min)           1.00       1         13.667    0.000384 ***
s(ToD)                1.00       1         6.547     0.012280 *
s(fScanID)            46.03      70        3.186     < 2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

R-sq.(adj) =  -0.175   Deviance explained = 82.4%
GCV = 0.61426  Scale est. = 0.39858   n = 138
Cross Validation R = 0.1476131
```

Appendix 4: Chapter 5 Supplementary Material

Contents:

Supplementary Table 4.1. Summary of dive depth values (m), including maximum, mean maximum and global mean; error presented is ±1SD and the sample sizes are given in parentheses.

Supplementary Table 4.2. Summary of dive duration values (sec), including maximum, mean maximum and global mean; error presented is ±1SD and the sample sizes are given in parentheses.

Supplementary Table 4.3. Summary of pause duration values (sec), for the global mean; error presented is ±1SD and the sample sizes are given in parentheses.

Supplementary Figure 4.1 The global mean for dive depth (a); dive duration (b); and pause duration (c) for common guillemot using direct studies and all studies; error presented is ±1SD refer to supplementary tables 1-3 for sample sizes.

Full Reference List

Comparison of results using direct only and all studies

I compared the results for dive depth, dive duration and pause duration for common guillemot. The results are outlined in the following three tables and the global mean ±1SD is also presented in the figure below.

Supplementary Table 4.1. Summary of dive depth values (m), including maximum, mean maximum and global mean; error presented is ±1SD and the sample sizes are given in parentheses. Sample sizes refer to the number of pieces of information used after averaging across multiple estimates for individual studies and/or colonies – see full description in methods. The category of studies used is either only or all data; a measure of confidence is given for each species – refer to table 5.1.

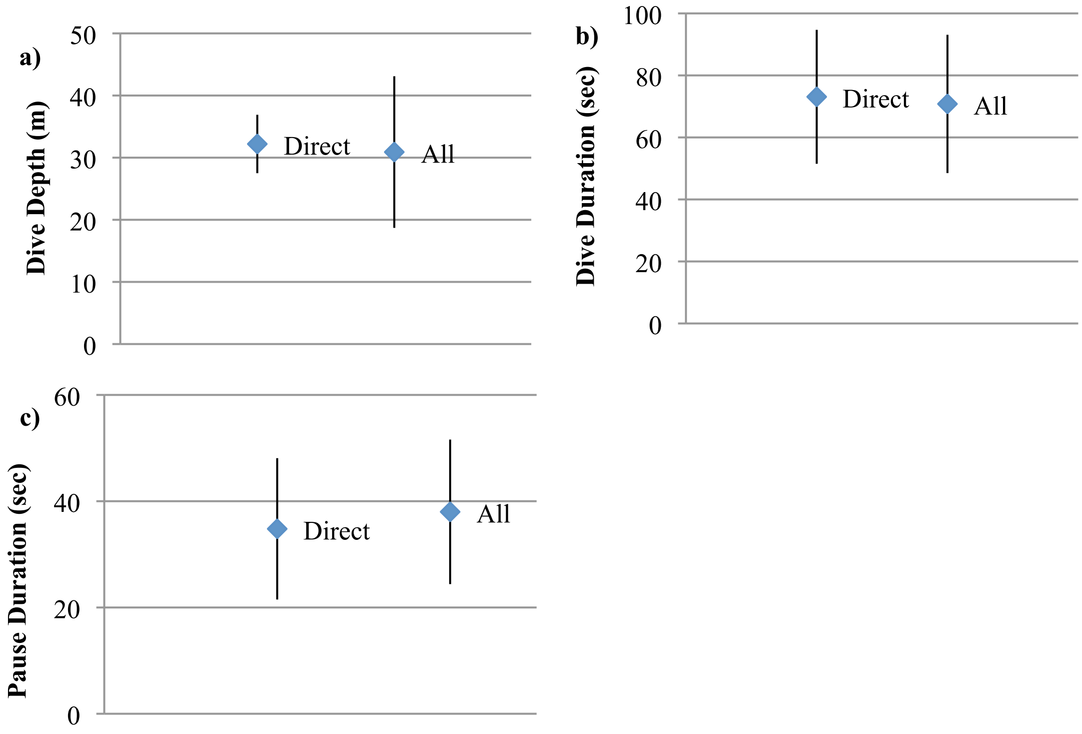
Species	Dive Depth Max	Mean max.	Mean	Category used	Confidence in data
Common Guillemot	200	101.7±58.2 (9)	30.9±12.2 (7)	All	High
Common Guillemot	177	98.0±59.5 (5)	32.2±4.7 (5)	Direct Only	High

Supplementary Table 4.2. Summary of dive duration values (sec), including maximum, mean maximum and global mean; error presented is ±1SD and the sample sizes are given in parentheses. Sample sizes refer to the number of pieces of information used after averaging across multiple estimates for individual studies and/or colonies – see full description in methods. The category of studies used is either only or all data; a measure of confidence is given for each species – refer to table 5.1.

Species	Dive Duration Max	Mean max.	Mean	Category used	Confidence in data
Common Guillemot	249	142.4±70.7 (11)	70.8±22.3 (10)	All	High
Common Guillemot	249	200.8±50.3 (5)	73.1±21.6 (6)	Direct Only	High

Supplementary Table 4.3. Summary of pause duration values (sec), for the global mean; error presented is ±1SD and the sample sizes are given in parentheses. Sample sizes refer to the number of pieces of information used after averaging across multiple estimates for individual studies and/or colonies – see full description in methods. The category of studies used is either only or all data; a measure of confidence is given for each species – refer to table 5.1.

Species	Pause Duration Mean	Category used	Confidence in data
Common Guillemot	38.0±13.6 (5)	All	High
Common Guillemot	34.8±13.3 (4)	Direct Only	Moderate



Supplementary Figure 4.1 The global mean for dive depth (a); dive duration (b); and pause duration (c) for common guillemot using direct studies and all studies; error presented is ±1SD refer to supplementary tables 1-3 for sample sizes.

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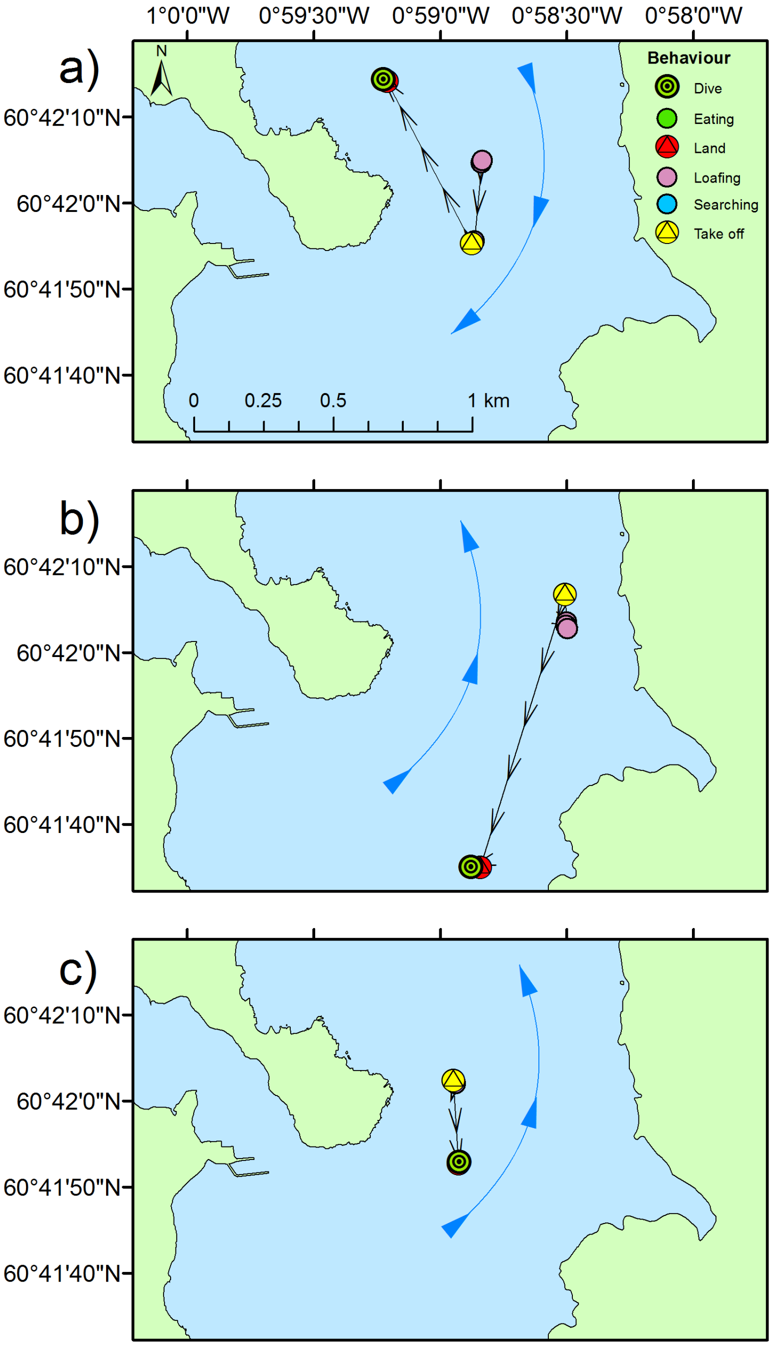
Appendix 5: Additional Analysis

I have plotted three example focal tracks using ArcGIS to demonstrate how they can be used to provide additional information on movement and behavioural usage, beyond data collected from traditional site scans.



Supplementary Figure 5.1: Three example tracks of focal observations undertaken at Bluemull Sound, Shetland in summer 2011-12 for a) black guillemot; b) black guillemot and c) European shag; where the blue arrows represents the tidal current direction and black arrows represents the individual birds direction.

Appendix 6: Scientific Contributions

Article 1:

Furness, R.W., Wade, H.M., **Robbins, A.M.C.**, Masden, E.A. 2012. Assessing the sensitivity of seabird populations to adverse effects from tidal stream turbines and wave energy devices. *ICES Journal of Marine Science*, 69: 1466-1479.

Article 2:

Waggitt, J.J., **Robbins, A.M.C.**, Wade, H.M., Masden, E.A., Furness, R.W., Jackson, A.C., Scott, B.E. 2017. Comparative studies reveal variability in the use of tidal stream environments by seabirds. *Marine Policy,* 81: 143-152.