Marine top predator responses to offshore windfarm developments

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A thesis presented for the degree of
Doctor of Philosophy in Ecology
at the University of Aberdeen

December 2021

THESIS'S CONTENTS

AUTHOR'S DECLARATION

I, hereby, declare that the work presented in this thesis is of my own composition, was undertaken by me, or principally by myself in collaboration with others as acknowledged below.

Chapter 2 was published in a peer-reviewed journal and therefore includes suggestions to the text from co-authors. The reference for this publication appears as a footnote at the beginning of the chapter. In addition, my supervisors, Paul Thompson and Isla Graham, provided suggestions to the core manuscript for all chapters. Nathan Merchant analysed the broadband noise data for Chapters 2 and 3. The Marine Laboratory of Marine Scotland Science analysed the chlorophyl α samples and Robert Watret carried out otolith extraction and reading. Oihane Fernandez and Joshua Lawrence provided advice on fisheries acoustic data processing for Chapter 5.

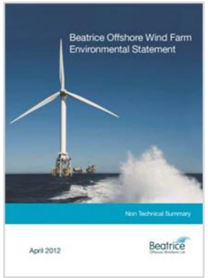
No part of this work has been accepted in any previous application for a degree. All quotations have been distinguished by quotation marks and the sources of information specifically acknowledged.

December 2021

ABSTRACT

The offshore wind energy sector has rapidly expanded over the last two decades and has an important role to play to meet nations' ambitious decarbonization targets. To minimize the potential impacts of these developments on protected species, such as seabirds and cetaceans, robust environment impact assessments are required. However, uncertainty over the effects, particularly cumulative, of developments may delay the consenting process. The overall aim of this thesis was to assess the behavioural responses of harbour porpoises and seabirds to specific construction and operation activities at offshore windfarm sites in the Moray Firth, Scotland. This thesis was the first study to characterise the overall vessel activity at offshore windfarm development sites. Results from a large array of passive acoustic monitoring devices deployed over three years during windfarm construction and operation revealed that porpoise occurrence and foraging activity decreased in relation to increased levels of vessel intensity and broadband noise during both preparatory activities prior to pile-driving and pile-driving itself. Furthermore, using digital aerial imagery data, this thesis provided empirical evidence that large gull species exhibited a strong attraction to jacket foundations, on which they perched, during construction before the final installation of wind turbines. This roosting behaviour was not observed after final construction, but high numbers of large gulls (N = 297) were observed within 250 m of turbines. Lastly, boat-based surveys were conducted to investigate the underlying processes influencing predator-prey interactions within and around an operational windfarm. Fisheries acoustic data highlighted that prey patchiness and depth were the two most important measures of prey availability influencing seabird distribution and abundance. This thesis provides evidence-based estimates of marine top predator behavioural responses to various disturbance sources during windfarm construction and operation, which can now be used to inform policy and management decisions.

Baseline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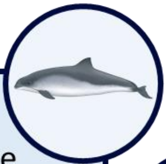

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ACKNOWLEDGEMENTS

I would like to express my sincere gratitude to both Prof Paul Thompson and Dr Isla Graham for supervising my PhD and their incredible support throughout this journey. First, thank you, Paul, for believing in my potential and giving me the opportunities to carry out my own research and fieldwork, network, be involved in other projects and for your guidance and very useful advice. To Isla, thank you for always being there for me, listening to my struggles, offering me unconditional support in the most stressful times, and providing me with fruitful comments and most importantly chatting about our Munros' adventures.

A massive thank you to the Lighthouse crew for their cheerful support, sharing their hands-on knowledge, their multiple competences and skills and for so much fun during our team-building rowing sessions. It is fair to say that this PhD would never have happened without you, and I feel very privileged and honoured to be part of the crew. Thank you to Tim Barton for preparing and supervising the deployment and recovery of passive acoustic monitoring (PAM) devices, for your problem-solving skills and always having the equipment I needed. Equally, a huge thanks to Dr Barbara Cheney for being involved in planning the PAM deployment and processing some of the PAM data, and allowing me to assist you during your dolphin photo-ID fieldwork. Most importantly thanks for your positivity, enthusiasm and brilliant human skills. A massive thank you to Becky Hewitt for your amazing editing skills that always make my presentations and figures look professional and colourful! An additional thanks for welcoming me on your seal photo-ID fieldwork, organising rowing sessions and for your kindness. Special thank you to Willow for bringing joy and happiness at coffee breaks! In addition, I would like to thank you Ruth Molloy for being the best human being! Thank you for helping with the PAM setting-up, deployment and recovery and for your amazing moral support and ornithological skills during my fieldwork, and most importantly big thanks for always bringing joy and happiness wherever you went. Another massive thank you to Dr Oihane Fernandez for your valuable help throughout my whole PhD, for taking the time to answer my technical questions, listen to my struggles with R, statistical analyses and

fisheries acoustic data, and for always being there when I needed it. I would also like to acknowledge Saliza Bono and Lea Brandes for their support in the first half of my PhD journey. May our paths meet again! And last but not least, a huge thank you to (soon-to-be Dr) Virginia Iorio for being such a good friend, an amazing office-mate, and the best assistant on the Alba na Mara cruise. I will always remember you duct taping GoPros all over the boat to get the best videos of each activity. Thank you for being there in my lowest moments, for cheering me up and always find the right words to make me feel better. You are a very promising researcher, a friend for life, and I am truly looking forward to working together in the near future!

Thank you to Bill Ruck for all your hard work to successfully deploy and retrieve all the PAM equipment, welcoming me on your boat for fieldwork. Thank you to Nathan Merchant who analysed the noise measurements, took time to answer my questions and provided me with useful comments for our manuscript. I would like to acknowledge Dr Mark Trinder, Dr Kate Brookes and Dr Jared Wilson for your fruitful feedback and help when needed. I am grateful to have been able to collaborate with the Beatrice and Moray East offshore windfarm developers who gave me access to digital aerial survey datasets and engineering records etc., allowed us to collect data at construction sites and provided valuable technical information during workshop and meetings.

Special thanks for the fieldwork conducted in Chapter 5:

I would like to thank Marine Scotland Science for giving me the opportunity to do boat-based surveys onboard the RV Alba na Mara. Thank you to the Alba na Mara's captain and crew for their hard work, their good mood and support. Most importantly, I would like to thank Robert Watret, the Scientist in Charge, who helped me plan, organise, gather equipment, and successfully conduct these surveys. Thank you for making this project possible! Thank you to Danny Copland for conducting the 38kHz transducer calibration, collecting and storing the fisheries acoustic data and providing me with useful information. Thank you to Virginia, Ruth, Rufus Danby, Kathy MacDonald and Dr Rory O'hara Murray for taking part in the surveys, helping collect, process and store data onboard, and having taken pictures and videos of all the activities conducted during the surveys. Your hard work and positivity made these surveys very enjoyable! I would also like to thank you Steven O'Connell from Marine Scotland Science, for enabling me to process the fisheries acoustic data. Thank you to the Marine Laboratory for letting me borrow equipment, providing me with consumables and most importantly for analysing chlorophyl samples. Thank you to Dr Margarita Machairopoulou and Dr Seóna Wells for letting me process the zooplankton samples at the lab. Thank you to Dr Hedda Weitz from the University of Aberdeen for preparing the 1% neutral red solution used to stain alive zooplankton. A massive thank you goes to Oihane, Rufus and Dr Joshua Lawrence for taking the time to answer all my questions on processing fisheries acoustic data and extracting fish schools echotraces. Thank you to Isla and Paul for helping me throughout the study, from the survey design to analysis and reporting and all the challenging steps in between.

To my family:

Un grand merci à ma famille pour son soutien pendant ces douze années d'études ! Merci d'avoir cru en moi et de m'avoir laissé voler de mes propres ailes ! Je ne serai certainement pas là sans vous et j'espère que peu importe les milliers de kilomètres qui nous séparent, nous serons toujours là les uns pour les autres ! Merci Sœurette, d'être venue me remonter le moral en Écosse pendant mes derniers mois de thèse, nos aventures m'ont toujours reboosté ! À maman, merci de m'avoir transmis ta détermination infaillible, j'espère que tu es fière de moi !

Finally, an infinite thank you goes to Graeme Bruce who has provided me with endless emotional support throughout my PhD journey, despite me being tired and stressed in the last few months. Thank you for your patience, and for our crazy adventures in the hills. Our good moments of laughs are priceless. To Yana, the best office-companion ever; set up like a clock, she has always reminded me when it is time for a well-deserved and energising walk.

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GENERAL INTRODUCTION

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GENERAL INTRODUCTION

Given the present Global Climate Emergency, the need to transition to sustainable low carbon energy systems has been widely recognized and is a component of the United Nations' Sustainable Development Goals (United Nations 2015). This requires a paradigm shift to address climate change by moving away from our reliance of fossil fuels, using renewable energy sources instead (York and Bell 2019). To achieve global decarbonization, whilst ensuring a "just transition" (McCauley and Heffron 2018), policies must align to deliver net-zero greenhouse gas emissions, while promoting fairness and equity and supporting energy justice (McCauley et al. 2019). In Europe, the European Union (EU) aims to reduce their greenhouse gas emissions by 55% by 2030 (compared to 1990 levels) and be net-zero by 2050 (European Parliament 2020). Further, the Renewable Energy Directive (2018/2001/EU) aims to generate at least 40% of energy from renewable sources by 2030 (European Parliament 2018; European Commission 2021). Similarly, a net-zero by 2050 target has been enshrined into the United Kingdom (UK) legislation. Efforts to meet these ambitious net-zero carbon emission targets have led to the rapid growth of the renewable energy industry sector. As Europe has extensive marine space with strong wind potential, offshore wind power offers a valuable source of renewable energy. Thus, offshore windfarms are considered to have an essential role in global renewable energy policies, providing an effective tool for nations to deliver their renewable energy production targets.

Emergence of the offshore windfarm sector

Historically, windfarms began to move offshore in the early 1990s, through small near-shore demonstrator projects (Jameson et al. 2019). It was only in 2002 that Horns Rev 1 was commissioned in Danish waters, becoming the first large-scale (160 MW, 80 2-MW turbines) commercial offshore windfarm in the world (Bailey et al. 2014). As technologies improve, the ability to deploy higher capacity turbines in greater numbers has led to larger windfarms being installed further offshore in deeper waters (Bailey et al. 2014). In 2020, 162 offshore windfarms were generating electricity globally with a further 26 under construction. The offshore wind energy industry has expanded rapidly over the last two decades, with a cumulative installed offshore capacity of 32.5 GW worldwide (World Forum Offshore Wind 2021). Of this, 25 GW (77%) can be found in European waters, with 5,402 grid-connected wind turbines across 12 countries by the end of 2020 (GWEC 2021; WindEurope 2021). The majority of commissioned windfarms are currently located in European coastal waters, primarily in the North and Baltic Seas (Figure 1.1). The UK is committed to generate 40 GW of offshore wind energy by 2030 (BEIS 2021).

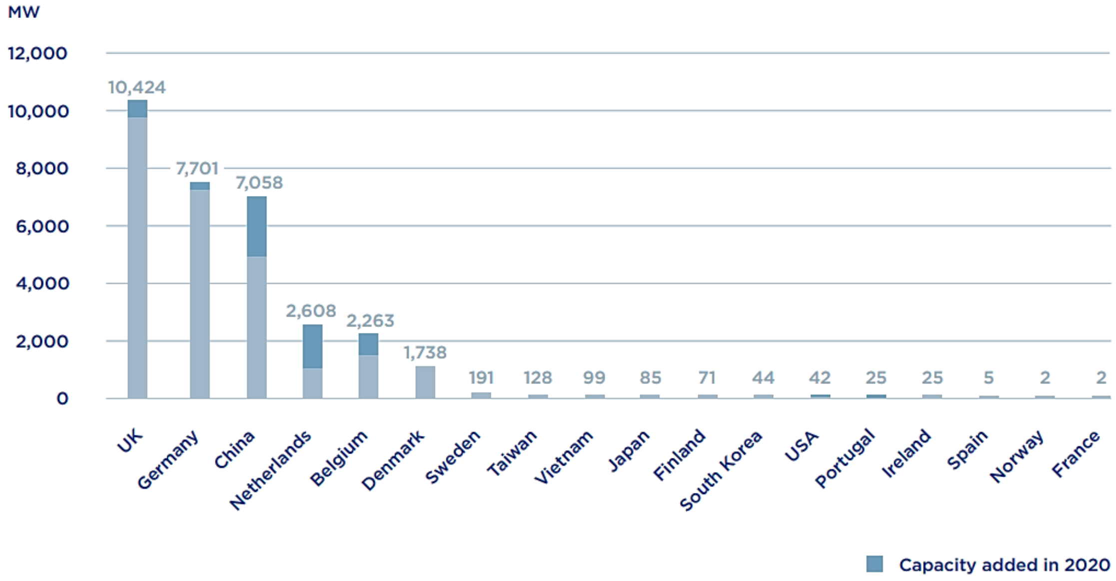

Figure 1.1 Global offshore wind capacity in operation by country in 2020 (©Global Forum Offshore Wind)

Within the UK, offshore wind developments require a lease from the Crown Estate or the Crown Estate Scotland, who own the seabed up to the 12 nmi territorial boundary and the rights to generate offshore renewable electricity beyond 12 nmi. Since 2000, there have been several leasing rounds, to encourage the expansion of large-scale commercial offshore windfarm developments (RenewableUK 2016). The Round 1 (2001), intended as a pilot phase, limited windfarms to 30 turbines. Thirteen Round 1 windfarm projects were developed reaching 1.2 GW capacity (Broadbent and Nixon 2019) (Figure 1.2). The second Round, launched in 2003, involved larger windfarm projects with greater turbine capacity up to 7.2 GW. For the third Round, in 2008, nine additional zones, including the Moray Firth zone, were identified for lease with the aim to deliver 25 GW by 2020. For this Round, the Crown Estate worked closely with developers at the earliest stages of the development to facilitate the consenting process (Jameson et al. 2019), with the Moray East project being consented in 2014 and the Moray West project consented in 2019. In 2009, the Crown Estate also awarded seabed exclusivity to 10 sites in Scottish territorial waters, which included the Beatrice offshore windfarm site studied in this thesis.

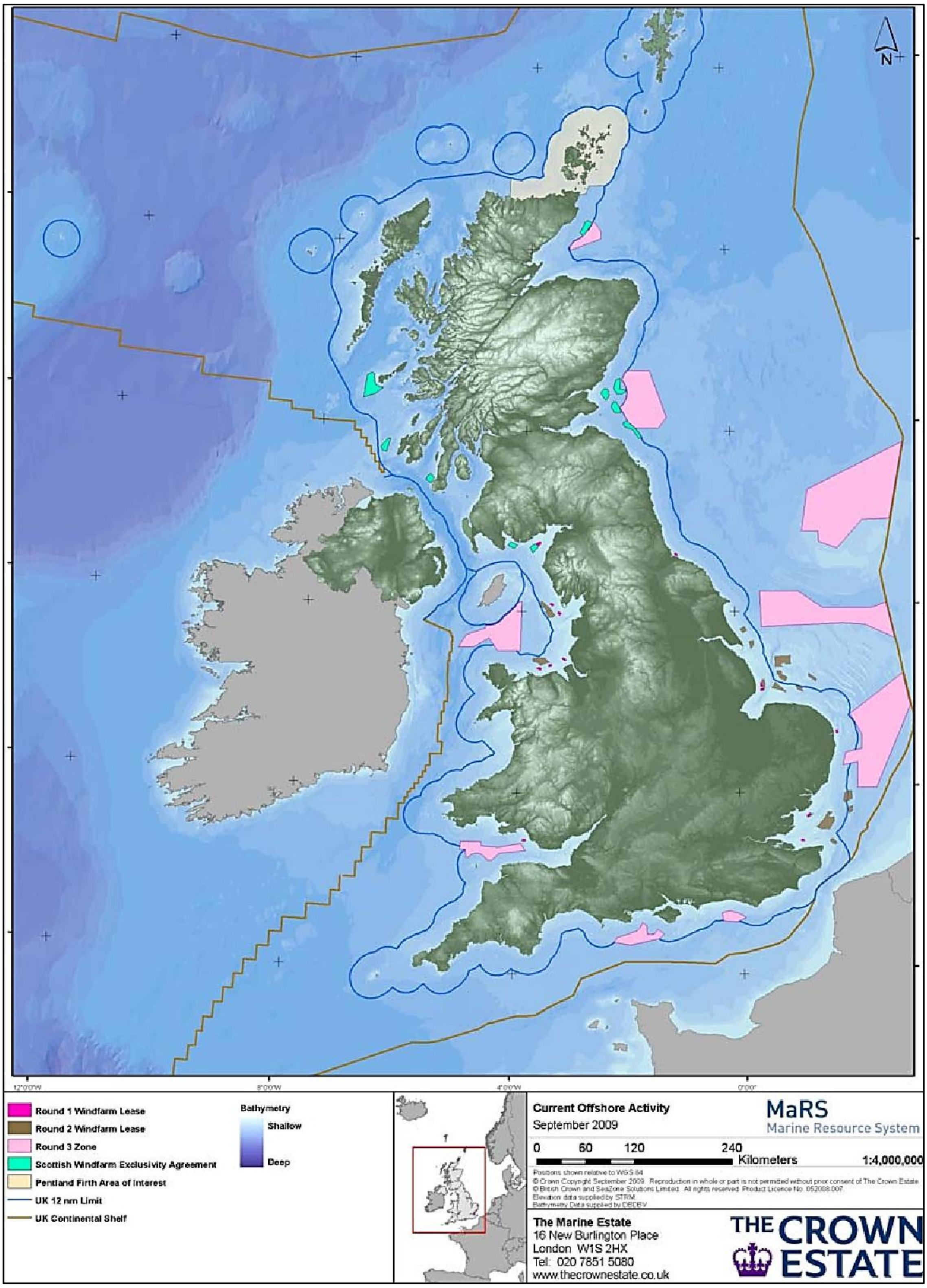

Figure 1.2 Existing Round 1 and 2 areas, potential Round 3 development zones and Scottish Territorial Waters proposed sites as of September 2008. (©The Crown Estate)

Legislation

International Conventions for the conservation and protection of biodiversity have played an important role in shaping the legal frameworks and regulation of these UK offshore wind developments. European Union (EU) legislation transposed the requirements of these International Conventions into European Directives, several of which are key for offshore windfarm developments. The amended Environmental Impact Assessment (EIA) 2014/52/EU Directive requires the assessment of (public or private) projects likely to have significant impacts on the environment. The EU Habitats (92/43/EEC) and Birds (2009/147/EC) Directives have been implemented through the designation of Natura 2000 (UK) sites, such as Special Area of Conservation (SACs) and Special Protection Areas (SPAs) for habitats and birds respectively (European Commission 2010). Under these two directives, the potential impacts of individual offshore wind projects and the cumulative impact of several development projects on protected populations must be assessed (Broadbent and Nixon 2019). As a result of leaving the EU, these EU Directives have been transposed into the national legislation. In Scotland, for example, the Habitats Regulations were amended in 2019 (Scottish Government 2020) to ensure effort on nature protection and conservation is maintained.

Due to the potential environmental impacts of offshore windfarm developments, these large projects require appropriate consent from the relevant governing authorities. The UK consenting process differs between countries. In England and Wales, any offshore wind projects with capacity > 50 MW fall under the Planning Act 2008 and the Secretary of State for Energy and Climate Change has the authority to grant a Development Consent Order, based upon fulfilment of pre-application requirements (Jameson et al. 2019). In Scotland, developers must obtain consent under the Section 36 of the Electricity Act 1989 and a Marine Licence must be awarded under the Marine (Scotland) Act 2010 (Jameson et al. 2019). In both cases, as part of their pre-application requirement, developers must submit an Environmental Statement (ES) which summarises the detailed Environmental Impact Assessment (EIA) undertaken to estimate the biological significance of potential impacts of these developments on the marine environment.

Overview of the negative and positive effects of offshore windfarm construction and operation

Offshore windfarm developments have the potential to be both detrimental and beneficial to the marine environment (Inger et al. 2009). The nature, magnitude, and duration of these ecological and environmental effects may be context-dependent, site- and species-specific. They may also vary spatially and temporally, and differ between development phases (Gill 2005). To date, effort has focused on assessing and mitigating the adverse effects of specific stressors on protected ecosystem components (receptors) during offshore windfarm developments. Key environmental concerns related to offshore windfarm developments are the adverse effects of increased underwater noise on acoustically sensitive species (marine mammals and fish) and the risk of collision with physical structures for aerial species (birds and bats). Developments also have the potential to change the natural structure and functioning of the benthic (seafloor) and pelagic ecosystems and alter the food webs. Finally, concerns have been raised over changes to hydrodynamic conditions and sediment transport, the release of chemical contaminants, and the effects of electro-magnetic fields emitted from cables during operation (Bailey et al. 2014).

During the operational phase, changes in local environmental conditions associated with the introduction of new hard substrates may have positive impacts on key receptors through habitat gain (Bergström et al. 2014). Further, in some windfarm sites, commercial fishing activities can be restricted, which in return may provide shelter for mobile species and enhance local biomass. Thus, the in-combination positive and negative effects of individual windfarm development and the cumulative impacts of one windfarm with existing and planned windfarm developments and/or with other marine activities should be considered during the Environmental Impact Assessment process (Perrow 2019).

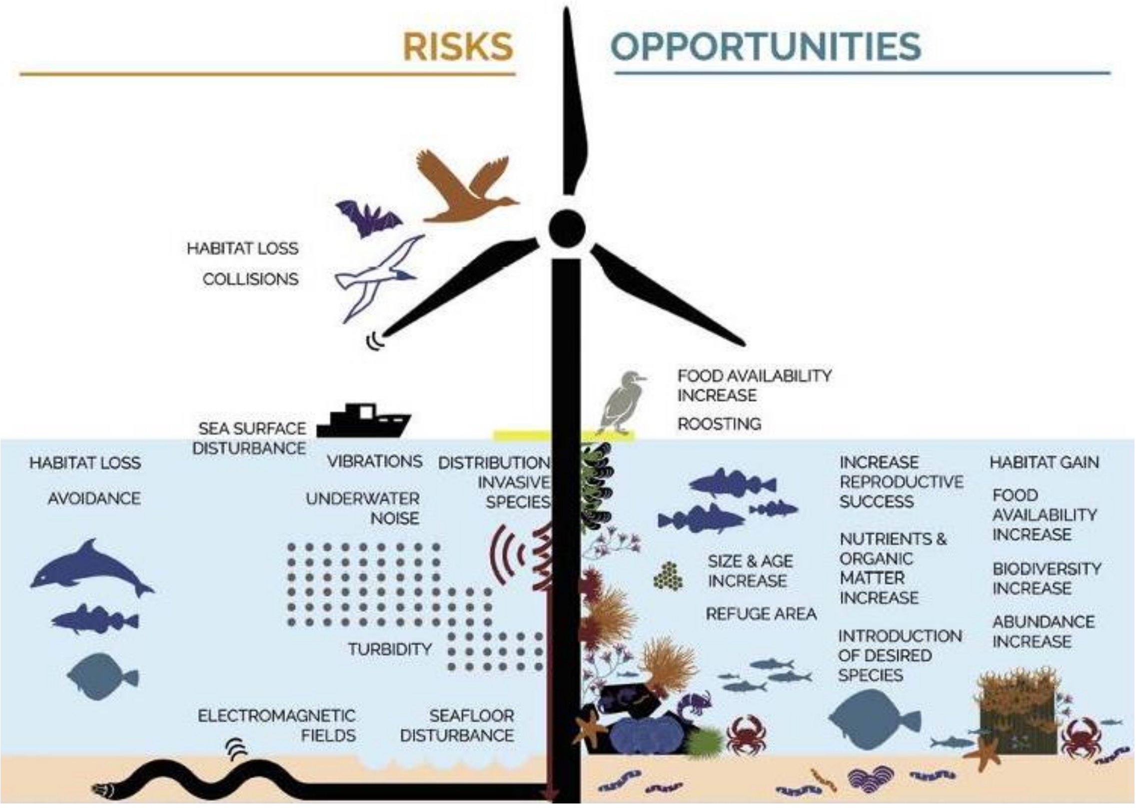

Figure 1.3 Overview of the risks and opportunities for the wildlife from offshore windfarms. Source: Stichting De Noordzee, 2018

In this thesis, I focus on the potential effects of offshore wind development on two key receptors: marine mammals and seabirds.

Marine mammals

The construction phase is considered to have the greatest potential for impacts on marine mammals, due to increased underwater noise levels from pile-driving activities and vessel traffic (Bailey et al. 2014). Offshore wind turbines are usually mounted on monopiles or steel lattice-type structures (also called jacket foundation) that are fixed to the seabed. The installation of either monopile or jacket foundations typically requires pile-driving activities. Impact pile-driving involves driving a large-diameter pile tens of meters into the seabed with a hammer (Thomsen and Verfuss 2019). For acoustically sensitive species, such as marine mammals, the impulsive noise generated during pile driving activities can cause instantaneous death if animals are at very close range to the noise source, or cause permanent or temporary, physical or auditory injury at greater ranges (Southall et al. 2008; Southall et al. 2019). Ships mostly produce low-frequency noise (10-100 Hz) but can also emit significant energy at higher frequencies, potentially overlapping with frequency bands of biological relevance for both mysticetes (i.e. baleen whales) (Cranford and Krysl 2015) and odontocetes (i.e. toothed whales) (Hermannsen et al. 2014). Consequently, both piling and vessel noise from offshore windfarm developments can alter the local soundscape and cause sub-lethal behavioural effects. For instance, elevated anthropogenic noise can mask signals from conspecifics or environmental cues which may alter social cohesion, reduce the communication space (Putland et al. 2018; Erbe et al. 2019), alter migration routes and navigation (Moore and Clarke 2002), and affect the detection of prey and predator (Erbe et al. 2016). Behavioural responses to disturbance can be displayed as avoidance through displacement, where animals leave an ensonified area, either temporarily or permanently (Tougaard et al. 2012). Animals can also modify their acoustic behaviour, ceasing vocalising (Wisniewska et al. 2018) or modulating the spectral features of acoustic signals in response to noise (*cf.* Lombard effect) (Branstetter et al. 2018; Erbe et al. 2019). Behavioural reactions, resulting from chronic exposure to anthropogenic disturbance may have fitness costs on individuals, e.g. due to increased energy expenditure and/or reduced feeding success. In the longer term, this may impact survival and reproductive success, which may have population-level consequences (King et al. 2015; Booth et al. 2017; Nabe-Nielsen et al. 2018).

Finally, during offshore windfarm operation, low-frequency continuous noise generated by operational wind turbines can be detected by marine mammals. The zone of audibility of < 2 MW wind turbines was estimated to be up to 2.5 - 10 km for harbour seals and 63 m for harbour porpoises (Tougaard et al. 2009b). However, based on published measurements and simulations, Stöber and Thomsen (2021) esimated that larger turbines such as a 10 MW direct drive turbine, may cause behavioural responses in marine mammals up to 1.4 km from turbines, which is greater than the distance between turbines. As a result, the authors suggested that the impact area of individual turbines could overlap and the whole windfarm should be considered an impact area (Stöber and Thomsen 2021). Maintenance activities may also induce some level of disturbance, through increased vessel traffic (e.g. Mendel et al. (2019)). In some operational offshore windfarms, however, vessel traffic is prohibited or restricted, which should reduce fishing pressure and overall disturbance at these sites. Marine mammals may use operational windfarms such as these as shelter to avoid neighbouring ensonified areas (Scheidat et al. 2011). Additionally, the introduction of hard substrates on soft sandy bottom can change fish assemblages, diversity and biomass in the vicinity of offshore structures, and thus may enhance foraging opportunities for top predators (Scheidat et al. 2011). For instance, Russell et al. (2014) found strong evidence that both harbour and grey seals were attracted to offshore windfarms, as they foraged at subsea structures. Harbour porpoises may also display this type of behaviour as increased foraging activity has been recorded in the vicinity of offshore oil and gas platforms in the North Sea despite elevated noise levels (Todd et al. 2009; Clausen et al. 2021).

Seabirds

For seabirds, it is the operational phase that is likely to have the greatest impact on their populations. The three key effects of offshore windfarms on seabird populations are barrier effects, displacement or attraction, and collision mortality (Cook et al. 2018). Seabirds may avoid travelling within a windfarm, which acts as a barrier to movement, and consequently may result in increased travel times to foraging areas or during migration (Desholm and Kahlert 2005), thus increasing energy expenditure (Masden et al. 2010; Cook et al. 2018). Displacement can be defined as the decrease in seabird occurrence around and within the windfarm footprint due to functional habitat loss and/or modification (May et al. 2015). Breeding seabirds, as central place foragers, are constrained to forage within a certain range from the breeding colony (Thaxter et al. 2012) and thus may be particularly vulnerable to loss of key foraging areas through displacement. Consequently, effective habitat loss or increased competition in alternative foraging areas may lead to changes in food intake and energy budgets, with potential consequences on both individual and population fitness (through reduced survival and reproductive success) (Fox et al. 2006). Attraction may result from increases in food availability in the vicinity of wind turbine foundations and underwater infrastructures, which could act as artificial reefs or fish aggregating devices (Wilhelmsson et al. 2006; Inger et al. 2009). Wind turbine structures can also serve as roosting platforms for seabirds (Dierschke et al. 2016), and birds attracted to windfarm sites may face higher collision risks (Vanermen and Stienen 2019). The magnitude of spatial changes in (at-sea) seabird distribution due to offshore windfarms can be divided into three scales; macro-, meso- and micro-avoidance (May et al. 2015; Cook et al. 2018). Macro-avoidance includes the barrier effect, displacement through functional habitat loss and attraction. Meso-avoidance refers to the anticipatory and impulsive evasion (of rows) of wind turbines due to perceived potential predation or collision risks. Micro-avoidance is considered as the last-second escape response to avoid colliding with moving wind turbine blades (May et al. 2015; Cook et al. 2018).

Seabirds are also susceptible to direct mortality caused by collision with wind turbine blades and other structures, which may have population consequences due to reduced survival rates (Fox et al. 2006; Lane et al. 2020). Population vulnerability to collision risk is species-specific due to the high variability in behavioural response, flight altitude, speed and manoeuvrability, seasonal and nocturnal flight activity, habitat specialisation, and regional population status between species (Furness et al. 2013; Bradbury et al. 2014; Bowgen and Cook 2018; Furness et al. 2018; Harwood et al. 2018; Skov et al. 2018; Masden et al. 2021). Other factors influencing collision risk can be windfarm- and site-specific, based on the layout, density, height, status, design of wind turbines, and the relative importance of the area for seabirds for transiting, foraging or even breeding (Furness et al. 2013; Bradbury et al. 2014). To date, a key area of uncertainty concerns the population-level impacts of additional mortality caused by collisions, and estimating the number of additional mortalities a population could sustain remains challenging (O'Brien et al. 2017; Miller et al. 2019; Lane et al. 2020).

ENVIRONMENTAL IMPACT ASSESSMENTS OF OFFSHORE WINDFARM DEVELOPMENTS

Given the potential risk on protected populations, regulators must ensure that developers undertake assessments of the potential effects of proposed projects and ensure that the scale of the impact on the environment is considered “acceptable” (Broadbent and Nixon 2019).

These EIAs provide information on the project seeking consent and use the best available knowledge to characterise the baseline conditions of the site and the key environmental receptors that may be affected by the development. Within the EIA, direct and indirect effects of the project on key receptors, throughout the offshore windfarm lifespan, must be identified, assessed, and when necessary, mitigated. Further, the in-combination synergistic and antagonistic effects of the same development and the cumulative effects with other marine activities should be considered (Caine 2019; Perrow 2019).

Thus, the EIA consists of four key steps. First, understanding the baseline spatio-temporal distribution and abundance of highly mobile receptors, such as marine mammals and seabirds. This step is essential to estimate the numbers of individuals likely to be affected by the planned offshore windfarm development. The second step involves assessing, with standardised approaches, the potential behavioural responses of receptors to the development stage(s) likely to significantly disturb or kill individuals (e.g. avoidance of marine mammals during pile-driving activities or collision of seabirds with turbine blades). The third step aims to predict the consequences of behavioural changes on the survival and productivity of affected individuals. The last step evaluates the population-scale impacts of the development on protected species (Searle et al. 2014; King et al. 2015; Booth et al. 2017; Nabe-Nielsen et al. 2018; Searle et al. 2018). Further, the cumulative effects of one windfarm both with other windfarms (either consented or planned) and with other human activities must be assessed through cumulative impact assessments. Currently, uncertainties over the cumulative effects of developments makes impact assessment particularly challenging, delaying or, in some cases, halting the consenting process. For instance, after eight years of development, the Docking Shoal windfarm project has been refused planning consent due to its potential impacts on seabird populations and the potential cumulative impacts associated with neighbouring offshore windfarm developments (Broadbent and Nixon 2019). With the rapid expansion of offshore windfarm developments, cumulative ornithological impact is a growing concern and could potentially become a barrier to further development in UK waters (Jameson et al. 2019). For example, several future developments areas, identified during the Scotwind leasing round, are subject to a moratorium until better evidence is available on cumulative collision risks for seabirds (Marine Scotland 2019). The expansion of the industry to meet net-zero carbon emission targets therefore depends on more research to understand nature and scale of these impacts.

STUDY AREA

The Moray Firth, a 5,230 km^2 coastal embayment in north-east Scotland, has experienced two large-scale offshore windfarm developments over the last 10-15 years (Figure 1.4). These developments have been the focus of strategic research studies (the Marine Mammal Monitoring Programme - MMMP) developed, in accordance with key stakeholders, to understand the potential impacts of offshore windfarm construction and operation on key protected marine mammal populations (Graham et al. 2017a).

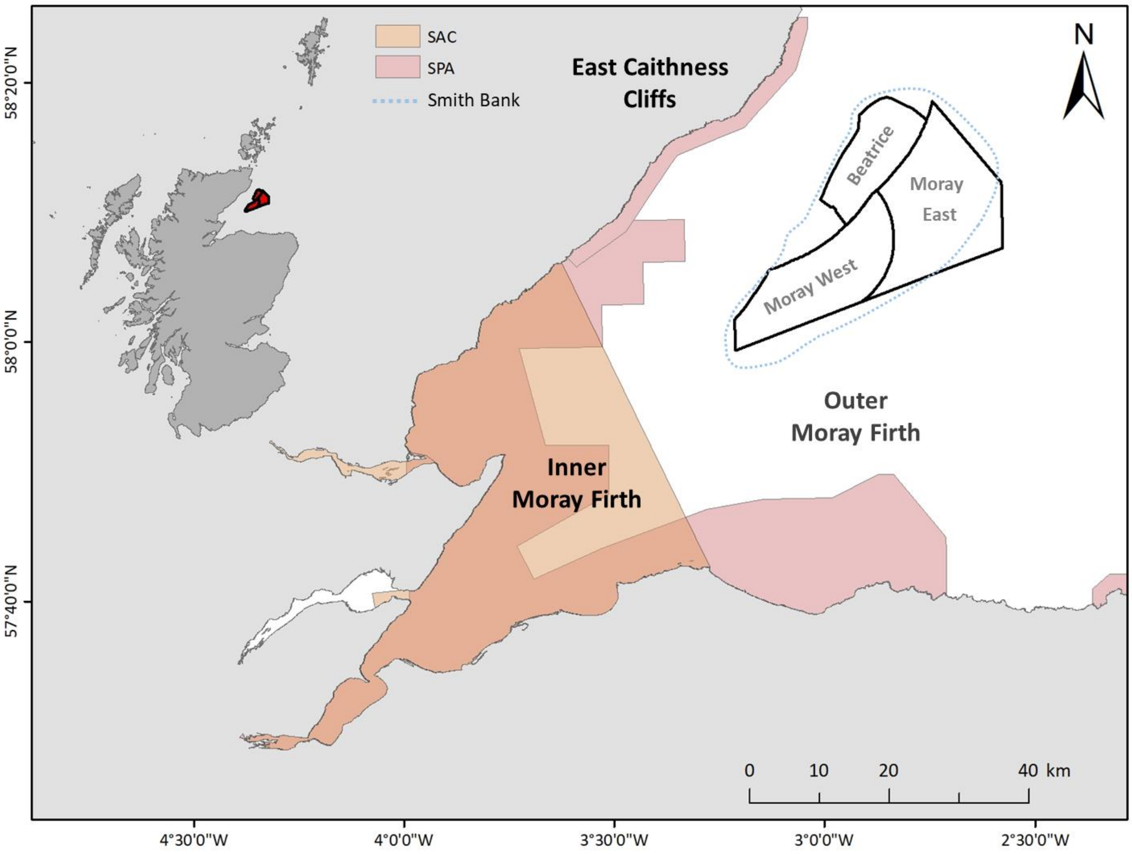

Figure 1.4 Map of the Moray Firth (NE Scotland, UK) with the designated Special Areas of Conservation (SAC) in orange and Special Protection Areas (SPA) in pink; the Smith Bank approximate boundary is delimited with a dashed blue line and the three offshore windfarm developments are delimited with black lines. As of December 2021, Beatrice is operational, Moray East is under construction and Moray West under consent.

The Smith Bank is an offshore sandbank in the outer Moray Firth, approximately 15 km from land, located at the centre of offshore windfarm developments. The bank covers an area of approximately 35 km by 20 km and is mainly composed of Holocene gravelly sand and sand, at water depths ranging between 35 to 68 m below the Lowest Astronomical Tide (Holmes et al. 2004). The outer Moray Firth is characterised by seasonal thermal stratification with a thermocline at 10 – 15 m depth in summer, but also weak local thermal fronts and very low concentrations of suspended sediment (< 5 $mg.L^{-1}$) (Beatrice Offshore Wind Farm Limited 2012). Further, the Smith Bank, and more broadly the Moray Firth, supports a wide variety of protected marine top predators, commercial fish and shellfish species, diadromous migratory species and species of high importance in the food web.

The Moray Firth is recognised for its ecological interests and accommodates internationally important populations of marine mammals and numerous breeding seabirds and over-wintering waterbirds. The Moray Firth supports the only known resident bottlenose dolphin (*Tursiops truncatus*) population in the North Sea (Wilson et al. 1997) and thus was designated SAC, in 2005, to protect the core range of this species (Cheney et al. 2014). The resident population of harbour seals (*Phoca vitulina*) breed and haul out on the inter-tidal sandbanks in the inner Moray Firth (Thompson et al. 1996) and thus the Dornoch Firth and Morrich More SAC was designated in 2005 (Figure 1.4) (Cordes et al. 2011). Other protected species such as grey seal (*Halichoerus grypus*), harbour porpoises (*Phocoena phocoena*) (Brookes et al. 2013; Williamson et al. 2017; Williamson et al. 2021) and minke whales (*Balaenoptera acutorostrata*) (Robinson and Tetley 2007; Robinson et al. 2009; Risch et al. 2019) are common in the Moray Firth, with other cetacean species occasionally recorded (Table 1.1). Harbour porpoises, one of the species studied in this thesis, is the most common cetacean species detected at offshore development sites in the North Sea. In the Moray Firth, harbour porpoise occurrence is highest at the Smith Bank and relatively low in coastal areas (Williamson et al. 2021). Porpoise distribution and activity vary seasonally, diurnally (Williamson et al. 2017; Williamson et al. 2021), with habitat type (Brookes et al. 2013; Williamson et al. 2017) and in response to perceived predation risk by bottlenose dolphins (Williamson et al. 2021), seismic surveys (Pirotta et al. 2014a) and pile-driving activities (Graham et al. 2019).

Table 1.1 Marine mammal species known to occur in the Moray Firth, NE Scotland (Reid et al. 2003; Robinson et al. 2007; Thompson et al. 2015)

Common name	Scientific name	Occurrence
Harbour seal	*Phoca vitulina*	Common, all year
Grey seal	*Halichoerus grypus*	Common, seasonal
Harbour porpoise	*Phocoena phocoena*	Common, all year
Bottlenose dolphin	*Tursiops truncatus*	Common, all year
Common dolphin	*Delphinus delphis*	Common, seasonal
White-beaked dolphin	*Lagenorhynchus albirostris*	Common, seasonal
Minke whale	*Balaenoptera acutorostrata*	Common, seasonal
Risso's dolphin	*Grampus griseus*	Occasional
White-sided dolphin	*Lagenorhynchus acutus*	Occasional
Striped dolphin	*Stenella coeruleoalba*	Occasional
Killer whale	*Orcinus orca*	Occasional
Pilot whale	*Globicephala melas*	Rare
Humpback whale	*Megaptera novaeangliae*	Rare
Fin whale	*Balaenoptera physalus*	Rare
Sperm whale	*Physeter macrocephalus*	Rare
Northern bottlenose whale	*Hyperoodon ampullatus*	Rare
Beluga whale	*Delphinapterus leucas*	Rare

The East Caithness Cliffs Special Protection Area (SPA) and Site of Special Scientific Interest (SSSI) are designated for their internationally important seabird assemblage (79/409/EEC) (Figure 1.4). During the breeding season, the area accommodates large breeding colonies and supports 300,000 individual seabirds including Northern fulmar (*Fulmarus glacialis*), herring gull (*Larus argentatus*), great black-backed gull (*Larus marinus*), black-legged kittiwake (*Rissa tridactyla*), common guillemot (*Uria aalge*), razorbill (*Alca torda*), great cormorant (*Phalacrocorax carbo*) and European shag (*Phalacrocorax aristotelis*) (JNCC 2018) (Table 1.2).

Table 1.2 Breeding seabird species occurring at the East Caithness Cliffs Special Protection Area (SPA), with total number of breeding individuals (*) or pairs and the percentage of the biogeographic population it represents (JNCC 2018)

Species	Scientific name	Number of individuals (*) or pairs breeding		% of biogeographic population
Northern fulmar	*Fulmarus glacialis*	15,000		unknown
Herring gull	*Larus argentatus*	9,400		1
Great black-backed gull	*Larus marinus*	800		unknown
Black-legged kittiwake	*Rissa tridactyla*	32,500		1
Common guillemot	*Uria aalge*	106,700	*	3.1
Razorbill	*Alca torda*	15,800	*	1.8
Great cormorant	*Phalacrocorax carbo*	230		unknown
European shag	*Phalacrocorax aristotelis*	2,300		1.8

During the breeding season, seabirds, as central place foragers, rely on high quality and abundance of accessible prey within a constrained range from their nest for chick provisioning (Burke and Montevecchi 2009; Boyd et al. 2017). The mean foraging distance of the seabird species breeding at the East Caithness Cliffs SPA ranges from 4 to 50 km (see Thaxter et al. (2012)). Due to its close proximity to breeding colonies (approx. 20 km), the Smith Bank is known to be an important foraging area for seabirds (Mudge et al. 1984; Mudge and Crooke 1986).

Historically, the Moray Firth was an important fishing area, exploiting demersal gadoid species such as cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), and whiting (*Merlangus merlangius*), and flatfish species such as lemon sole (*Microstomus kitt*) and plaice (*Pleuronectes platessa*). Pelagic clupeid fish species such as sprat (*Sprattus sprattus*) and herring (*Clupea harengus*) were exploited in winter from the early 1960s, but the fishery was closed in the 1980s to prevent by-catch of juvenile herring (Hopkins 1986). The Moray Firth is the most important spawning ground in the North Sea for plaice and is an important nursery area for plaice and herring (Hopkins 1986). Further, although the sandeel (*Ammodytes spp*) fishery in the Moray Firth was sporadic (Hopkins 1986), the Smith Bank is a high intensity sandeel spawning ground and low intensity nursery ground (Ellis et al. 2010). Sediment grab sampling, benthic trawl and dredge surveys conducted at the Smith Bank before offshore windfarm developments confirmed that the sandbank is a suitable habitat for sandeels, with a high proportion of medium and coarse sediment and low silt content (Holland et al. 2005; Greenstreet et al. 2010; Langton et al. 2021). The overwintering sandeel distribution and density was variable across the Smith Bank (Beatrice Offshore Windfarm Limited 2012b). Sandeel, the mid-trophic link between zooplankton and piscivorous predators, is an important component of the North Sea food web (Wanless et al. 2004). In fact, sandeels are one of the key prey species for piscivorous fish (Greenstreet 1998), seabirds (Lewis et al. 2001; Wilson et al. 2004), seals (Wilson and Hammond 2019) and cetaceans (Santos and Pierce 2003; Booth 2019). Other species exploited in the Moray Firth were great scallops (*Pecten maximus*), Norway lobster (*Nephrops norvegicus*), crabs (e.g. *Cancer pagurus*) and veined squid (*Loligo forbesi*) (Hopkins 1986; Young et al. 2006). Finally, the 18 major rivers that flow into the Moray Firth have supported, historically, an annual run of up to 270,000 adult Atlantic salmon (*Salmo salar*) (Cox et al. 2020). Although salmon populations have declined, it still supports in-river rod fisheries (Butler et al. 2008; Fisheries Management Scotland 2021).

The Moray Firth has also supported a wide variety of anthropogenic activities including commercial and recreational shipping traffic (Merchant et al. 2014), ecotourism (Pirotta et al. 2015) and fishing (Kafas et al. 2014). In the outer Moray Firth, anthropogenic activities include oil and gas exploration (Thompson et al. 2013; Pirotta et al. 2014a) and extraction (Linsley et al. 1980), and offshore wind developments (Bailey et al. 2010; Graham et al. 2019).

RESEARCH AIMS AND OBJECTIVES

Part of the strategic research programme (MMMP), this thesis focuses on key questions that address uncertainties currently constraining efforts to assess overall impacts of offshore windfarm developments on marine top predators and identify the underlying drivers of their responses.

The effect of pile-driving during windfarm construction has been the focus of assessments of impacts on marine mammals. Despite this, other construction activities have the potential to change acoustic habitats through increased shipping activity, but the scale of impact from these activities remains uncertain. In Chapter 2, I aim to assess the extent to which animals are displaced throughout the whole windfarm construction period. To do so, I investigate the broad-scale spatio-temporal variation in harbour porpoise acoustic occurrence and foraging activity in relation to different windfarm construction phases. Combining the Before-After Control-Impact (BACI) and Impact gradient assessment techniques, I estimate the scale and magnitude of porpoise behavioural responses to different construction activities. Further, I describe the spatio-temporal distribution, density, and intensity of vessels to highlight the levels of vessel activity within and around offshore windfarm development sites.

Mitigating the cumulative effects of offshore windfarm development on marine mammals requires a good understanding and quantification of adverse effects of multiple stressors. For instance, it is not known to what extent preparation work, prior to pile-driving activities, may displace harbour porpoises and thus reduce risk of injury. In Chapter 3, I investigate how pre-piling activities affect local soundscapes and quantify changes in levels of vessel activity and harbour porpoise occurrence in the vicinity of construction sites. Following the same method as in Chapter 2, I use passive acoustic monitoring, broadband noise measurements, Automatic Identification System (AIS) vessel-tracking data and daily engineering records to characterise and compare levels of disturbance between two piling campaigns. Further, I estimate when and to what extent harbour porpoise occurrence may decline before the start of pile-driving activities. Finally, I discuss the management implications of the results and provide further recommendations to optimize construction work, while minimising far-field disturbance.

Once offshore windfarms are commissioned and operational, one of the main concerns is the risk of seabirds colliding with wind turbine blades. While the ecological impacts of offshore windfarm developments on seabirds have been extensively studied, estimating the population consequences of these developments remains challenging (Green et al. 2016; O'Brien et al. 2017; Searle et al. 2018). Ornithology consent monitoring surveys are usually conducted before and after windfarm construction, in order to assess changes in the broad-scale spatial distribution and density of seabirds in the vicinity of the windfarm site. Changes in the baseline distribution of seabirds in the early phases of a windfarm development may affect assumptions over baseline distributions used in Collision Risk Models. However, there is limited information on variation in seabird habitat use and behaviour during construction. In Chapter 4, I integrate data from different industry surveys to investigate variation in seabird distribution in relation to offshore structures throughout offshore windfarm development. Using digital aerial imagery data, I compare the horizontal meso-scale (< 1 km) distribution and behaviour of seabirds from three species groups (with different taxonomic and foraging constraints), before, during (i.e. jacket foundation installation) and after an offshore windfarm construction. The ultimate objective of this chapter is to highlight the importance of accounting for potential changes in seabird baseline distribution to predict, more accurately, the collision risk probability of some focal species for future developments.

Most impact assessment studies focus on the potential effects of windfarm developments on key receptors with conservation relevance, such as marine mammals and seabirds, without considering the pelagic effects of these developments on their prey. As one of the main drivers of predator occurrence is prey availability, understanding the underlying environmental and ecological processes driving predator and prey distribution and interactions is required to predict long-term cumulative effects of commissioned windfarms. In Chapter 5, I investigate the concomitant distribution of pelagic schooling fish and seabirds in the vicinity of an operational windfarm, during the breeding season. To achieve this goal, I conducted two boat-based fisheries acoustic and visual seabird surveys. Alongside this work, I also characterised variation in the primary and secondary production, and water column structure, providing new empirical data on the pelagic ecosystem within and around an operational windfarm.

Finally, Chapter 6 provides an overview of the main findings of the thesis (Chapters 2 to 5), discusses the wider implications of those findings on conservation and management, and highlights the requirements for further research.

CHAPTER 2

BROAD-SCALE RESPONSES OF HARBOUR PORPOISES TO PILE-DRIVING AND VESSEL ACTIVITIES DURING OFFSHORE WINDFARM CONSTRUCTION

CHAPTER 2

BROAD-SCALE RESPONSES OF HARBOUR PORPOISES TO PILE-DRIVING AND VESSEL ACTIVITIES DURING OFFSHORE WINDFARM CONSTRUCTION[1]

ABSTRACT

Offshore windfarm developments are expanding, requiring assessment and mitigation of impacts on protected species. Typically, assessments of impacts on marine mammals have focussed on pile-driving, as impulsive noise elicits adverse behavioural responses. However, other construction activities such as jacket and turbine installation also change acoustic habitats through increased vessel activity. To date, the contribution of construction-related vessel activity in shaping marine mammal behavioural responses at windfarm construction sites has been overlooked and no guidelines or mitigation measures have been implemented. I compared broad-scale spatio-temporal variation in harbour porpoise occurrence and foraging activity between baseline periods and different construction phases at two Scottish offshore windfarms. Following a Before-After Control-Impact design, arrays of echolocation click detectors (CPODs) were deployed in 25 km by 25 km impact and reference blocks throughout the 2017-2019 construction. Echolocation clicks and buzzes were used to investigate porpoise occurrence and foraging activity respectively. In parallel, I characterised broadband noise levels using calibrated noise recorders (SoundTraps and SM2Ms) and vessel activities using AIS data integrated with engineering records. Following an impact gradient design, I then quantified the magnitude of porpoise responses in relation to changes in the acoustic environment and vessel activity. Compared to baseline, an 8-17% decline in porpoise occurrence was observed in the impact block during pile-driving and other construction activities. The probability of detecting porpoises and buzzing activity was positively related to increasing distance from vessel and construction activities, and negatively related to increasing levels of vessel intensity and background

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[1] This chapter has been published as: Benhemma-Le Gall, A., Graham, I.M., Merchant, N.D., Thompson, P.M., 2021. Broad-Scale Responses of Harbor Porpoises to Pile-Driving and Vessel Activities During Offshore Windfarm Construction. Frontiers in Marine Science 8.

noise. Porpoise displacement was observed at up to 12 km from pile-driving activities and up to 4 km from construction vessels. This evidence of broad-scale behavioural responses of harbour porpoises to these different construction activities highlights the importance of assessing and managing all vessel activities at offshore windfarm sites to minimise potential impacts of anthropogenic noise.

INTRODUCTION

Offshore windfarm developments are currently expanding in response to global efforts to meet decarbonisation targets. Many countries aim to generate significant proportions of electricity from offshore wind sources by 2030 (BEIS 2019), but these developments must be in line with international conservation agreements such as the EU Habitats Directive 92/43/EEC and Environmental Impact Assessment 2014/52/EU Directives (Le Lièvre 2019). This requires assessment and mitigation of construction, operation and decommissioning activities to reduce potential impacts on marine wildlife. In particular, there have been concerns over the effect of high levels of underwater noise from different anthropogenic activities on cetaceans, with potential to cause either injury or behavioural disturbance (Richardson et al. 1995; Dolman and Simmonds 2010; Bailey et al. 2014)

Harbour porpoises (*Phocoena phocoena*), protected under the EU Habitats and Species Directive, are the most common cetacean species in offshore energy development sites within the North Sea (Thomsen et al. 2011; Hammond et al. 2013; Waggitt et al. 2020). Due to their high metabolic requirements, harbour porpoises are vulnerable to starvation (Wisniewska et al. 2016; Wisniewska et al. 2018; Kastelein et al. 2019a; Booth 2020) and, as a consequence, could be especially vulnerable to anthropogenic disturbance. In exposed areas, such as offshore windfarm sites, individuals have to make trade-offs between using energy to leave the area or remaining in exposed areas and tolerating higher levels and/or rates of disturbance (Frid and Dill 2002). These decisions are likely to be individual-based, context-dependent and site-specific, impacting individual activity budgets and fitness through reduced foraging performance (Booth 2020). As such, animals may be responding to natural environmental variation, and a variety of different anthropogenic stressors such as fisheries, shipping noise and construction activity. In turn, either individually or in combination, this may have significant long-term biological consequences at a population level (Pirotta et al. 2014a).

The construction and operation of offshore windfarms involves a variety of vessels and activities that could each generate many types of anthropogenic noise that potentially disturb harbour porpoises or other marine mammals. Previously, however, impact assessments have focussed on the loudest of these sources; impulsive noise from the pile-driving hammers used to install turbine foundations at most offshore windfarms (Madsen et al. 2006). Where these piling methods are used, mitigation typically involves either minimising the likelihood that animals are within the injury zone when piling is initiated (Thompson et al. 2020), or using noise abatement techniques such as bubble curtains (Dähne et al. 2017). Extensive research conducted around North Sea windfarm sites has demonstrated that harbour porpoises may be displaced at distances of up to 26 km from piling (e.g. Tougaard et al. (2009a); Brandt et al. (2011); Scheidat et al. (2011); Dähne et al. (2013); Haelters et al. (2015); Brandt et al. (2018)). However, porpoises are also known to be displaced by vessel noise at distances of up to 7 km (Hermannsen et al. 2014; Dyndo et al. 2015; Wisniewska et al. 2018), with the level of response dependent upon vessel type and behaviour (e.g. heading, speed) (Oakley et al. 2017; Hermannsen et al. 2019). Furthermore, even where animals are not displaced, porpoise foraging efficiency may be temporarily affected by exposure both to impulsive noise (Pirotta et al. 2014a; Sarnocińska et al. 2020) and vessel noise (Wisniewska et al. 2018).

Whilst previous studies recognised that construction vessel activity influenced porpoise displacement around pile-driving activities (Brandt et al. 2018; Graham et al. 2019), there remains uncertainty over the cumulative effects of different windfarm construction activities on displacement, foraging efficiency, and population fitness. From a management perspective, this constrains efforts to assess and mitigate potential disturbance from windfarm construction activities other than pile-driving. For example, the installation of jackets, turbines and cables may also disturb animals by altering acoustic habitats through intense vessel activity (Merchant et al. 2012; Merchant et al. 2014). Consequently, there may be opportunities to better manage vessels throughout construction and operation to minimise cumulative impacts of shipping movements that could affect harbour porpoise occurrence and behaviour. Furthermore, efforts to reduce impulsive noise levels during intermittent periods of pile-driving may result in longer-term noise from additional vessels. Better data on how harbour porpoises respond to different construction and operational phases of windfarm construction is therefore required to understand how different conservation interventions could affect broad-scale habitat displacement and foraging success, particularly within harbour porpoise Special Areas of Conservation (SAC) (JNCC and Natural England 2019).

In this study, I aimed to compare broad-scale spatio-temporal variation in harbour porpoise occurrence and foraging activity between baseline periods and different phases of construction at two offshore windfarms in the Moray Firth, NE Scotland. The Beatrice offshore windfarm (commissioned in 2019) is composed of 84 (7 MW) turbines and two substations mounted on quadrapod jackets, while the Moray East offshore windfarm (to be commissioned in 2021) will have 100 (9.5 MW) turbines and three substations mounted on tripod jackets. Previous studies in this area used two complementary approaches to assess harbour porpoise responses to impulsive noise from seismic surveys (Thompson et al. 2013; Pirotta et al. 2014a) and pile-driving (Graham et al. 2019). First, Before-After Control-Impact (BACI) designs (Underwood 1992; Smith 2002) were used to determine whether variations in porpoise occurrence (Thompson et al. 2013) and activity (Pirotta et al. 2014a) were related to these anthropogenic disturbances. Second, impact gradient sampling designs (Ellis and Schneider 1997) were also applied in each of these studies to estimate the spatial scale of effects (Graham et al. 2019). Here, I build on these studies, using a BACI design to determine how porpoise occurrence and activity were impacted during different construction phases, and a gradient design to explore how responses varied in relation to the distance from piling vessels as they undertook different activities. Finally, I characterised finer-scale variation in vessel activity and noise levels during different phases of construction and explored how these influenced spatio-temporal variation in porpoise occurrence and activity within the construction sites.

MATERIAL AND METHODS

Study area and context

The study was carried out in 2017, 2018 and 2019 during the construction of the Beatrice Offshore Windfarm and the first phase of construction at Moray East Offshore Windfarm (Figure 2.1). Beatrice was constructed between March 2017 and May 2019. From April 2017 to December 2017, an anchored piling vessel used impulsive pile-driving to install four 2.2 m diameter steel piles at 86 locations (Graham et al. 2019). Jackets were then installed onto each set of foundation piles between August 2017 and August 2018 using a jack-up vessel. This vessel was also used to install towers, nacelles and blades on each jacket, and the windfarm was fully operational in May 2019 (see Figure 2.1). Other activities such as boulder removal, inter-array and export cable installation and protection took place during the windfarm construction phase but were not investigated specifically in this study. Construction at Moray East started in May 2019 and the windfarm is anticipated to be fully operational in 2021. Between May and December 2019, a jack-up vessel used impulsive pile-driving to install three 2.5 m diameter steel piles at the first 90 Moray East locations. There was no overlap between the piling campaign and the jacket foundation installation phase at Moray East (see Table S 2.1).

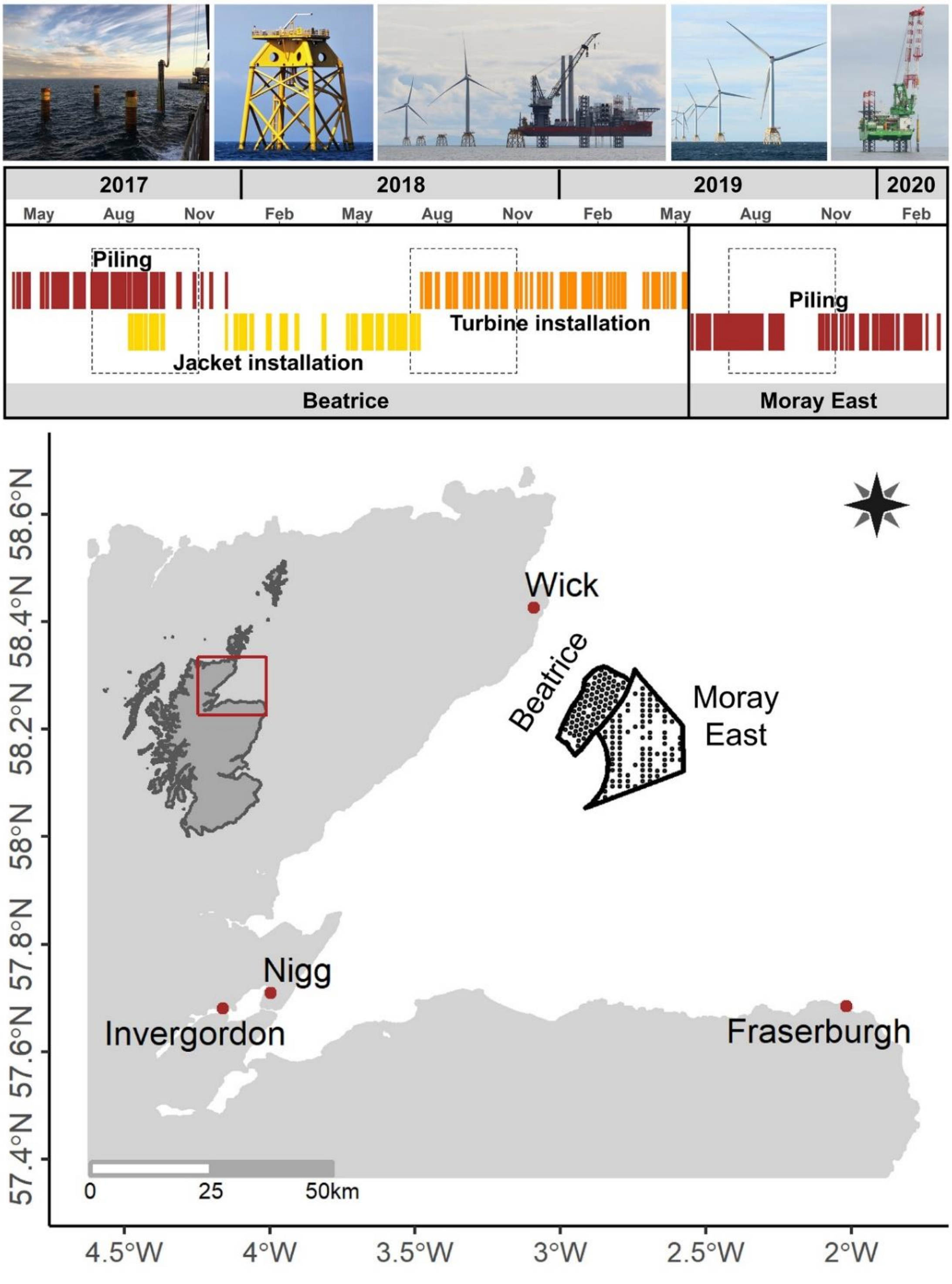


Figure 2.1 A) Timeline of key construction activities at two offshore windfarms, between 2017 and 2019, i.e. pile-driving at Beatrice and Moray East (in red), jacket foundation installation at Beatrice (in yellow), wind turbine installation at Beatrice (in orange); the time periods used in the Before-After Control-Impact models to compare baseline periods with key construction activities are represented with dashed rectangles. B) Map showing the location of the Moray Firth in Scotland, the Beatrice and Moray East windfarm boundaries (black line) and turbine locations (black dots), and the harbours used as construction bases (red dots); Source (first picture): Beatrice Offshore Wind Limited.

Passive Acoustic Monitoring

Sampling design and data collection

Following the sampling design used by Thompson et al. (2013), I investigated variation in harbour porpoise occurrence and buzzing activity using arrays of echolocation click detectors (V.0 and V.1 CPODs (www.chelonia.co.uk)). These devices were deployed a) in a 25 km by 25 km impact and reference block throughout construction in 2017, 2018 and 2019 and b) along a gradient of exposure from construction activities within the two windfarm sites (Figure 2.2). These data were also compared to baseline data that had been collected in 2010 and 2011 to support Environmental Impact Assessments (EIA) (Beatrice Offshore Windfarm Limited 2012a; Moray Offshore Renewables Limited 2012b). In parallel, calibrated noise recorders (Ocean Instruments SoundTrap and Wildlife Acoustics SM2M) were deployed at three locations to characterise variation in underwater noise levels (Figure 2.3, Figure S 2.1A and Figure S 2.1B).

Measuring variation in harbour porpoise occurrence

Echolocation click characteristics (e.g. time of occurrence, duration, centre frequency, bandwidth) logged by the CPODs were processed and extracted with the manufacturer's software CPOD.exe (v2.044). The standard built-in "KERNO" classifier allocates click trains into one of four signal classes (Narrow Band High Frequency "NBHF", "Other cetaceans", "Boat Sonars" and "Unclassified") and one of four quality categories (high "Hi", moderate "Mod", low "Lo" and doubtful "?"). No information on the design of the classifier is currently available, but based on the manufacturer's CPOD manual (Tregenza 2014), the classification algorithm searches for specific click parameters and inter-click intervals within trains (Clausen et al. 2019). High and moderate quality NBHF echolocation click trains of porpoise origin were extracted as Detection Positive Minutes per hour and then converted into presence-absence of porpoise detections per hour to assess hourly porpoise occurrence.

To extend endurance, CPODs are typically set up to log a maximum of 4,096 clicks per minute. This means that high levels of background noise can quickly saturate the

CPODs and prevent any further data logging until the start of the next minute (Wilson et al. 2013). Additionally, the probability of detecting acoustic signals can be affected by the acoustic environment (Clausen et al. 2019). To estimate the distance at which CPODs were unlikely to saturate because of piling or construction vessel noise around each turbine site, I first extracted the number of unfiltered clicks (*Nall*) logged per minute by each device during the 10 months of piling activity in 2017. I then summarised these data in relation to distance from the piling vessel (Figure S 2.2) and took a conservative approach to prevent false-negative detections; discarding all data from CPODs within 2 km of the piling vessel and all hours with less than 60 minutes logged.

Estimating variation in harbour porpoise foraging activity

I used variation in inter-click intervals (ICIs) to identify buzzes and provide a proxy for foraging activity (Pirotta et al. (2014b). The ICIs of logged NBHF click trains were calculated, normalized by natural log-transformation, and categorized into three groups representing specific biological processes. The first group represents the high repetition rate click trains called buzzes that may be used for both foraging activity and social communication (Sorensen et al. 2018; Sarnocińska et al. 2020). Currently, it is not possible to distinguish between these two behaviours but, as in earlier studies (Pirotta et al. 2014a; Pirotta et al. 2014b; Sarnocińska et al. 2020), I assumed that buzzes can be used as a proxy for foraging. The second group includes regular click trains and the third group represents the time between different click trains (Pirotta et al. 2014b). To identify the multimodal distribution of ICIs and allocate each ICI to one of the processes, Gaussian mixture models were fitted to the time series of ICIs, using the package *mixtools* (Benaglia et al. 2009) in R (v 3.6.0) (R Core Team 2019). The number of component distributions *k* was initially set equal to three. However, at some locations, the low number of ICIs prevented the model from identifying the distribution centred on the buzz ICIs and so the number of components (*k*) was increased to four. If the model still did not discriminate the buzz ICI distribution using four components, data were pooled, so that datasets with higher proportions of buzz ICIs helped identify the buzz ICIs in datasets with overall lower numbers of detections. Additionally, when models did not converge after 1000 iterations, I increased the number of iterations to 2000 (and on one occasion reduced the convergence precision (epsilon) to 0.0001). Mixture models with 3 or 4 components were compared, choosing the model with the maximum loglikelihood. Results from the best model were then used to categorise each ICI into one of the three processes (e.g. Figure S 2.3) and the number of buzzes, regular and inter-train interval clicks were summarised per hour. The number of buzzes was converted into binary presence-absence of buzzes per hour, reducing the potential bias due to differences in sensitivity and detection range between acoustic devices and locations, respectively.

Before-After Control-Impact analyses of variation in porpoise occurrence and foraging activity in relation to different phases of windfarm construction

For the BACI models, variation in both porpoise occurrence and foraging activity within each 25 km by 25 km block (Figure 2.2) were compared between the baseline and each monitoring phase. These analyses focused on data collected between July and October when comparable data were available in all years (see Table 2.1). Baseline data used in both windfarm EIAs were collected in 2010 and 2011 (Moray Offshore Renewables Limited 2012b). As seismic surveys were conducted in the current study's reference block between 1st and 11th September 2011, I excluded these data from the analyses. For the BACI modelling, the Beatrice piling phase was from July to October 2017, during which 221 piling events occurred at 52 turbine locations and 24 jackets were installed. The turbine installation phase was from July to October 2018, during which 32 turbines and the last 6 jackets were installed. The Moray East piling phase, from July to October 2019, included 165 piling events at 47 turbine locations. No further construction work occurred at Beatrice during this time period, but operations and maintenance vessels visited the site regularly once it became fully operational in May 2019 (Figure 2.2).

Table 2.1 Sampling effort used for the Before-After Control-Impact models

Monitoring phases		**Reference Block**		**Impact Block**	
		# CPOD sites	**# days**	**# CPOD sites**	**# days**
Baseline	July-October 2010	18	110	18	122
	July-October 2011	15	110	9	123
Beatrice piling	July-October 2017	9	122	24	123
Beatrice turbine installation	July-October 2018	4	123	4	123
Beatrice operation – Moray East piling	July-October 2019	4	122	16	123

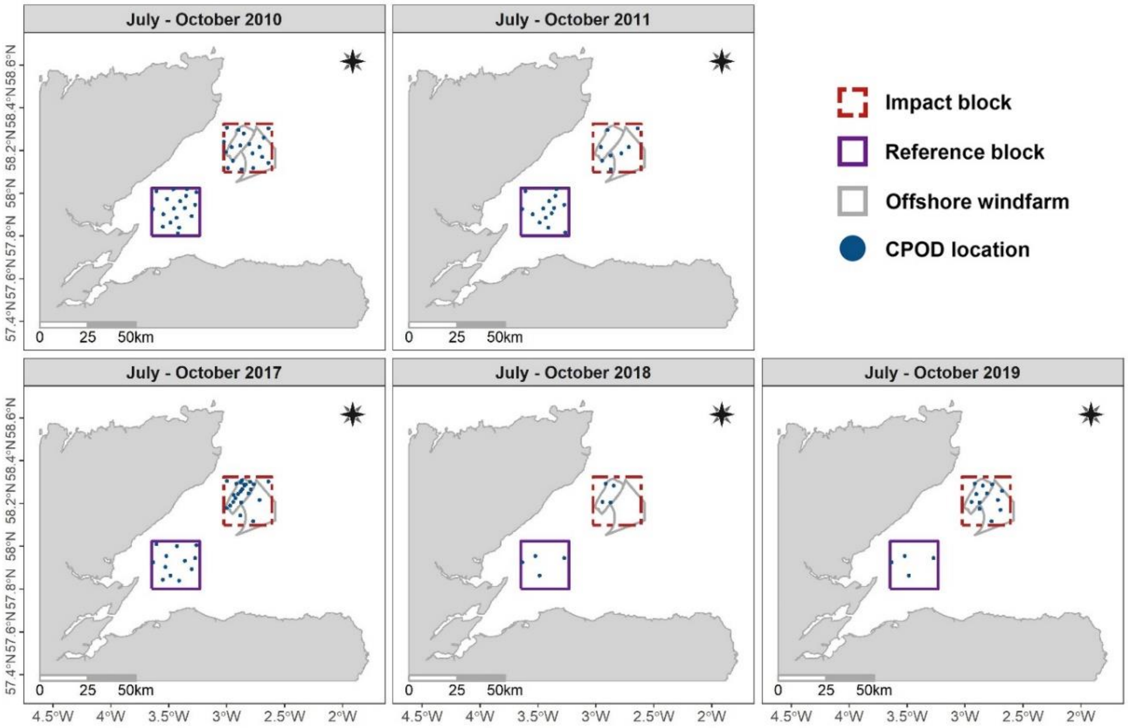


Figure 2.2 Spatio-temporal distribution of echolocation click detectors (CPODs) within the reference and impact blocks and the offshore windfarms between 2010 and 2019.

Spatial scale of porpoise responses to different piling vessel activities

An impact gradient approach was used to assess finer scale variation in porpoise occurrence and buzzing activity in relation to distance from the piling vessels at both Beatrice and Moray East as they undertook different construction activities in 2017 and 2019. The position of each piling vessel was extracted from an Automatic Identification System (AIS) vessel-tracking dataset for the Moray Firth region. The mean and minimum distance between each CPOD and the piling vessel were then calculated for each monitoring hour, using the *sf* package (Pebesma 2018). Information on the activity of the vessels was extracted from the developers' daily construction reports, and the factor *piling* or *no piling* was allocated to each hour monitored. Distance from the piling vessel was used as a proxy for the distance from construction activities (i.e. the noise/disturbance source), as the piling vessel was supported by two pilot vessels for anchoring, at Beatrice, and a tug bringing the piles on site. Hourly porpoise occurrence and buzzing activity were each modelled as a function of distance from the piling vessel in interaction with the vessel's activity (*piling* or *no piling*) (Table 2.2).

Table 2.2 Sampling effort used for the gradient models, within the impact block, to assess harbour porpoise responses to pile-driving activities at Beatrice between March and December 2017 and Moray East between May and December 2019; mean harbour porpoise occurrence and mean foraging activity when porpoises were detected during and outside piling hours

Windfarm (year)	**Piling phase**	**Porpoise occurrence**				**Buzzing activity**			
		# site	**# day**	**# hour**	**mean per hour**	**# site**	**# day**	**# hour**	**mean per hour**
Beatrice (2017)	no piling	34	204	54,467	0.319	34	204	17,387	0.279
	piling	34	102	4,816	0.195	31	89	939	0.285
Moray East (2019)	no piling	29	219	83,841	0.458	23	219	29,166	0.348
	piling	29	104	7,773	0.321	23	103	1,883	0.363

Characterising vessel activity

To characterise variation in the extent to which harbour porpoises were exposed to both piling vessels and other construction vessels, I integrated data from the developers' engineering records with AIS vessel-tracking data (Wright et al. 2019). AIS data for the entire Moray Firth were sourced at 5 min (2017) or 1 min (2018 and 2019) resolution from Astra Paging Ltd. (www.astrapaging.com) and Anatec Ltd. (www.anatec.com).

A 4 km by 4 km grid was created across the Moray Firth and the area of each grid cell calculated after any grid cells overlapping coastlines were cropped. AIS data were projected into WGS84 UTM 30N and then processed to produce hourly summaries a) within each of these grid cells and b) within a 5 km buffer around each of the passive acoustic monitoring sites. AIS data were interpolated every 5 min to calculate the time that each individual vessel stayed in a grid cell or buffer area. These data were then used to estimate measures of both vessel density and vessel intensity for each hour within each grid cell or buffer area. These two metrics provide complementary information highlighting variation in vessel behaviour and distribution across the Moray Firth. At windfarm sites, construction-related vessels are often stationary for several hours, while other vessels (not involved in the construction) are likely to be transiting and consequently contribute less to the overall vessel intensity. Here, I defined vessel density as the number of individual vessels present in that hour per kilometre squared, and vessel intensity as the sum of residence times for all vessels present in that hour per kilometre squared. The minimum and mean distance from each CPOD or noise recorder to all vessels within each buffer area were also calculated and summarised for each hour and location.

Information on the vessels involved in the windfarm construction was extracted from the developers' weekly construction reports and used to filter AIS data to provide separate measures of vessel density and intensity a) for construction vessels and b) for other third-party marine traffic. To estimate the vessel density and intensity within each of the windfarm construction sites, these AIS data were filtered by location, and vessels were categorised following Table 1 in Metcalfe et al. (2018).

Variation in background noise levels at the construction site

Underwater broadband noise recorders were deployed for periods of 2-6 months at three sites within the impact block to characterise noise levels in different phases of construction (Table 2.3, Figure 2.4, Figure S 2.1). Recorders collected data at sampling rates of either 48 kHz or 96 kHz, with duty cycle rates varying depending upon device and sampling rates (Table S 2.2). Data were processed in MATLAB following Merchant et al. (2015). Broadband noise levels were quantified between 25Hz and 24 kHz to provide hourly root-mean-square (RMS) averaged sound pressure levels (SPL) in decibels (dB) relative to a reference pressure of 1 µPa (Kinsler et al. 1999; Merchant et al. 2015).

Table 2.3 Sampling effort to investigate harbour porpoise responses to vessel activity and underwater broadband noise levels

PAM devices	**2017**		**2018**		**2019**	
	# site	**# day**	**# site**	**# day**	**# site**	**# day**
CPOD	40	317	24	342	30	275
Noise recorder	2	128	2	252	1	97

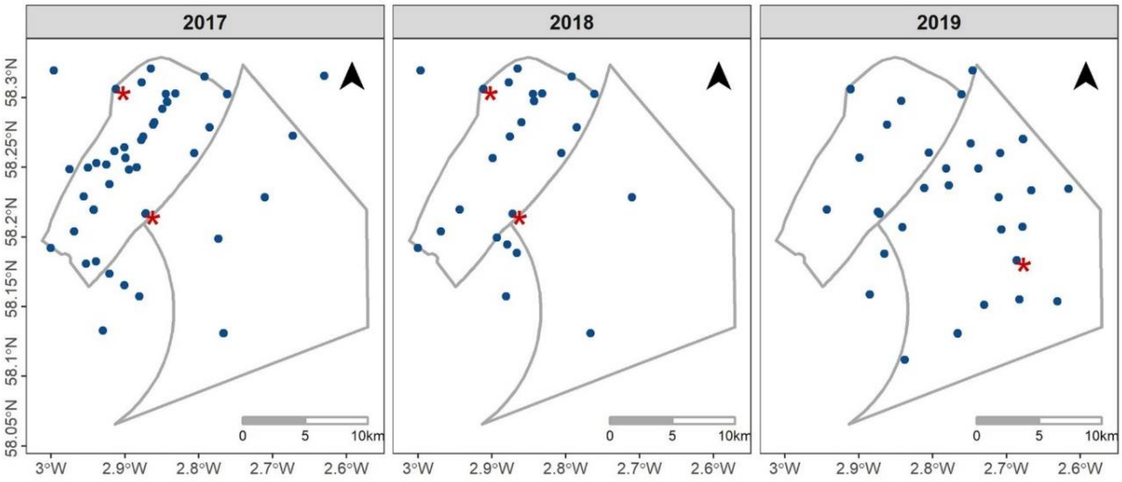


Figure 2.3 Spatio-temporal distribution of the Passive Acoustic Monitoring array, CPODs (blue circles) and noise recorders (red stars), within and around the two offshore windfarms between 2017 and 2019.

Modelling

BACI models

To compare between the baseline and each construction phase, the hourly occurrence of porpoise detections and buzz detections were modelled as binomial response variables using six Generalised Linear Mixed-effects Models (GLMM). For each model, the link function was chosen based on the lowest Akaike Information Criterion (AIC), using either the *cloglog* or *probit* link function. The interaction between block and construction phase was used as the explanatory variable in all models.

Based on previous studies and preliminary data analyses, it was known that porpoises display diel and seasonal patterns in occurrence and foraging activity in this study area (Williamson et al. 2017; Graham et al. 2019). To focus on changes associated with the windfarm construction, month and diel phase were used as random factors in the BACI model. Diel phase (i.e. sunrise, day, sunset or night) was allocated based on local sunrise and sunset times. Additionally, the CPOD location was also used as random effect to control for any site-specific environmental differences. To assess the significance of fixed effects and their interactions, a sequential analysis of deviance table (Type II Wald chi-square tests) was computed using the R package *car* (Fox and Weisberg 2019). For each model, the response variable was predicted, and the uncertainty (95% confidence intervals, CI) calculated using a bootstrapping approach (100 simulations) with the *bootMer* function of the *lme4* package (Bates et al. 2015). To validate models, I checked for temporal autocorrelation in model residuals using the partial auto-correlation function. The *DHARMa* package was used to verify the uniformity, dispersion, spatial and temporal autocorrelation of residuals (Hartig 2020).

Impact gradient models

To investigate the spatial scale of the effects of pile-driving, vessel activity and underwater noise on harbour porpoise occurrence and foraging activity, hourly occurrence of porpoise detections and buzz ICIs were modelled as a function of: 1) the interaction between the distance from construction activities and piling occurrence (*piling effect* models); 2) the interaction between the vessel intensity per hour and the mean distance from vessels (*vessel effect* models); 3) the averaged broadband sound pressure levels and piling occurrence (*noise effect* models).

For the *piling effect* models, I only considered sites within the impact block to investigate meso-scale response to construction activities. Additionally, I filtered the dataset to explore the magnitude of porpoise responses between 2 and 30 km from the piling vessel as it has been shown in other studies that a response to piling activities was apparent up to 26 km from piling (Tougaard et al. 2009a; Dähne et al. 2013; Haelters et al. 2015; Brandt et al. 2016; Brandt et al. 2018). Similarly, to investigate porpoise occurrence and activity in relation to broadband noise levels and the presence or absence of piling activity, I used data from the CPODs that were deployed at the same location as the noise recorders (Figure 2.3B). Hours in which piling activities occurred within 2 km of these sites were not used in the *noise effect* models.

For the *vessel effect* models, I excluded hours in which piling activities occurred to focus only on the effect of vessels. To prevent masking effects of vessel noise on porpoise echolocation clicks, I followed Pirotta et al. (2014a) and excluded hours in which vessels were within 1 km of CPOD locations.

For the six models, binary generalised linear models with either a *probit* or *cloglog* link function were fitted using generalised estimating equations (GEE-GLMs), to account for temporal autocorrelation. The correlation structure was selected based on the lowest Quasi Information Criterion (QICr), resulting in using an "independence" correlation structure for all models. To determine whether the two-way interaction term should be retained, I used the *dredge* function of the *MuMIn* package, which ranks all model possibilities according to QIC (Barton 2020). Year, Julian day and site ID were used to define a blocking variable in the GEE, allowing model residuals from each site within each day to be autocorrelated. Wald's tests were used to assess significance and bootstrapped coefficients from the GEE-GLM were used to estimate uncertainty (95% CI) and plot relationships between response and explanatory variables.

RESULTS

Variability in porpoise occurrence and foraging activity between different phases of windfarm construction

In the BACI comparison, variation in harbour porpoise occurrence and foraging activity was best explained by the interaction between block and construction phase (Figure 2.4). The baseline probability of occurrence prior to any construction was around 0.42 in the reference block and 0.55 in the impact block, while the baseline probability of detecting buzz ICIs, when porpoises were present, was around 0.3 for both blocks. In comparison with the baseline, harbour porpoise occurrence significantly decreased by 14.3% in the impact block during the Beatrice piling phase (Wald test: $\chi^2 = 725.267$, $p < 0.001$) and by 8% during the Moray East piling phase (Wald test: $\chi^2 = 126.024$, $p < 0.001$). A decrease in porpoise occurrence (-16.7%) was also observed between the baseline and the Beatrice turbine installation phase (Wald test: $\chi^2 = 6.269$, $p = 0.012$). Despite these significant decreases, harbour porpoises were regularly detected within these construction sites throughout the monitoring period (Figure S 2.4).

When porpoises were present, the probability of detecting buzz ICIs also significantly decreased by 4.2% between the baseline and piling phase at Beatrice (Wald test: $\chi^2 = 14.216$, $p < 0.001$), although no significant change in buzzing activity was observed during turbine installation (Wald test: $\chi^2 = 0.009$, $p = 0.923$). In contrast, during the Moray East piling phase, the probability of detecting buzzing ICIs increased by 11% in the impact block but decreased by 5% in the reference block (Wald test: $\chi^2 = 176.517$, $p < 0.001$) (Figure 2.4).

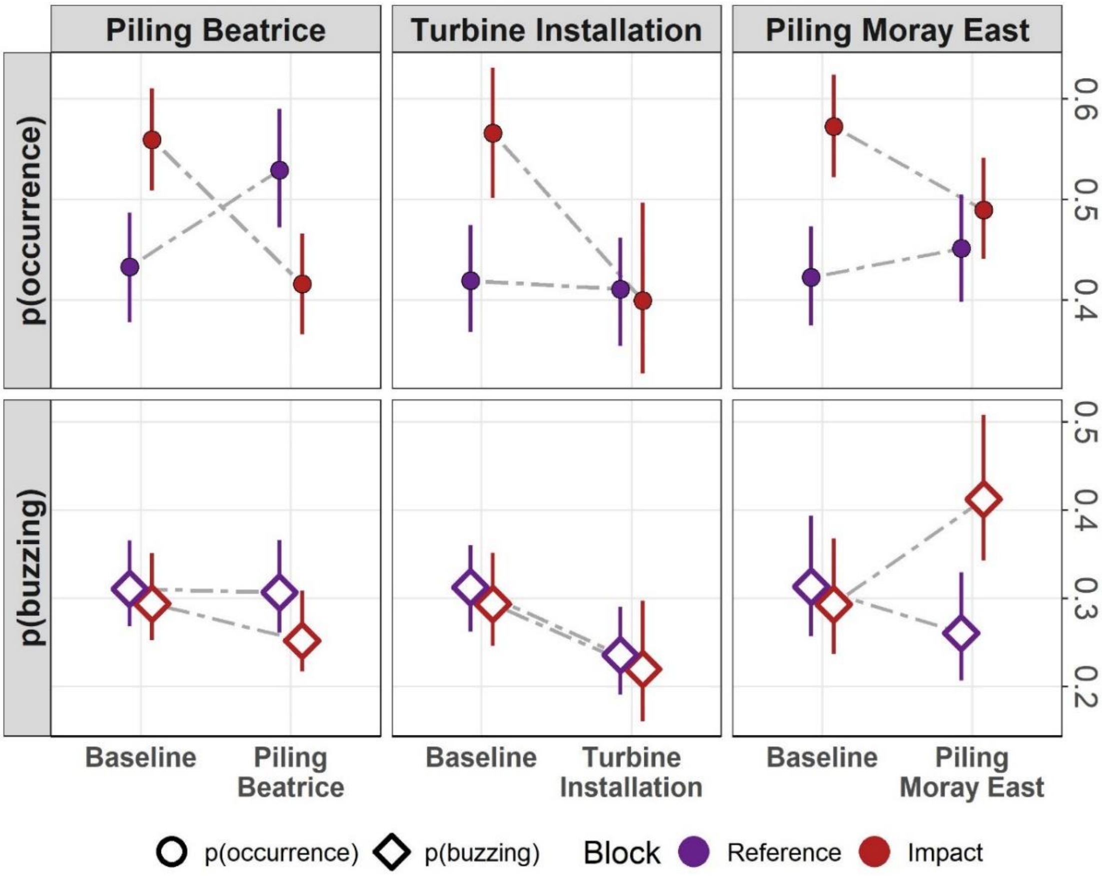


Figure 2.4 The probability of harbour porpoise occurrence (circle) and buzzing activity (diamond) per hour between the reference (in purple) and impact (in red) blocks and between the baseline monitoring period and key construction activities (i.e. pile-driving and wind turbine installation at Beatrice offshore windfarm and pile-driving at Moray East offshore windfarm).Spatial scale of porpoise responses to different piling vessel activities

During the construction period, finer-scale variation in harbour porpoise occurrence and foraging activity were best explained in the *piling effect* model by the interaction between the distance from the piling vessel and the presence or absence of piling (Beatrice: porpoise occurrence – Wald test: $\chi^2 = 482$, $p < 0.001$; buzzing activity –

Wald test: $\chi^2 = 18.3$, $p < 0.001$. Moray East: porpoise occurrence – Wald test: $\chi^2 = 421$, $p < 0.001$; buzzing activity – Wald test: $\chi^2 = 6$, $p < 0.014$). During piling activity, the probability of porpoise occurrence increased significantly with distance from the source vessel in a similar fashion at both sites. When there was no piling activity, occurrence still decreased slightly closer to the vessel (-9.3% at Beatrice; -20.9% at Moray East) (Figure 2.5). During piling activities, when harbour porpoises were acoustically detected, buzzing activity at Beatrice decreased by 54% close to the piling vessel, but this effect was not as strong at Moray East (-40%). Again, when there was no piling activity, the probability of buzzing ICIs at both sites was slightly lower closer to the piling vessel (-19.6% at Beatrice; -22.7% at Moray East) (Figure 2.5).

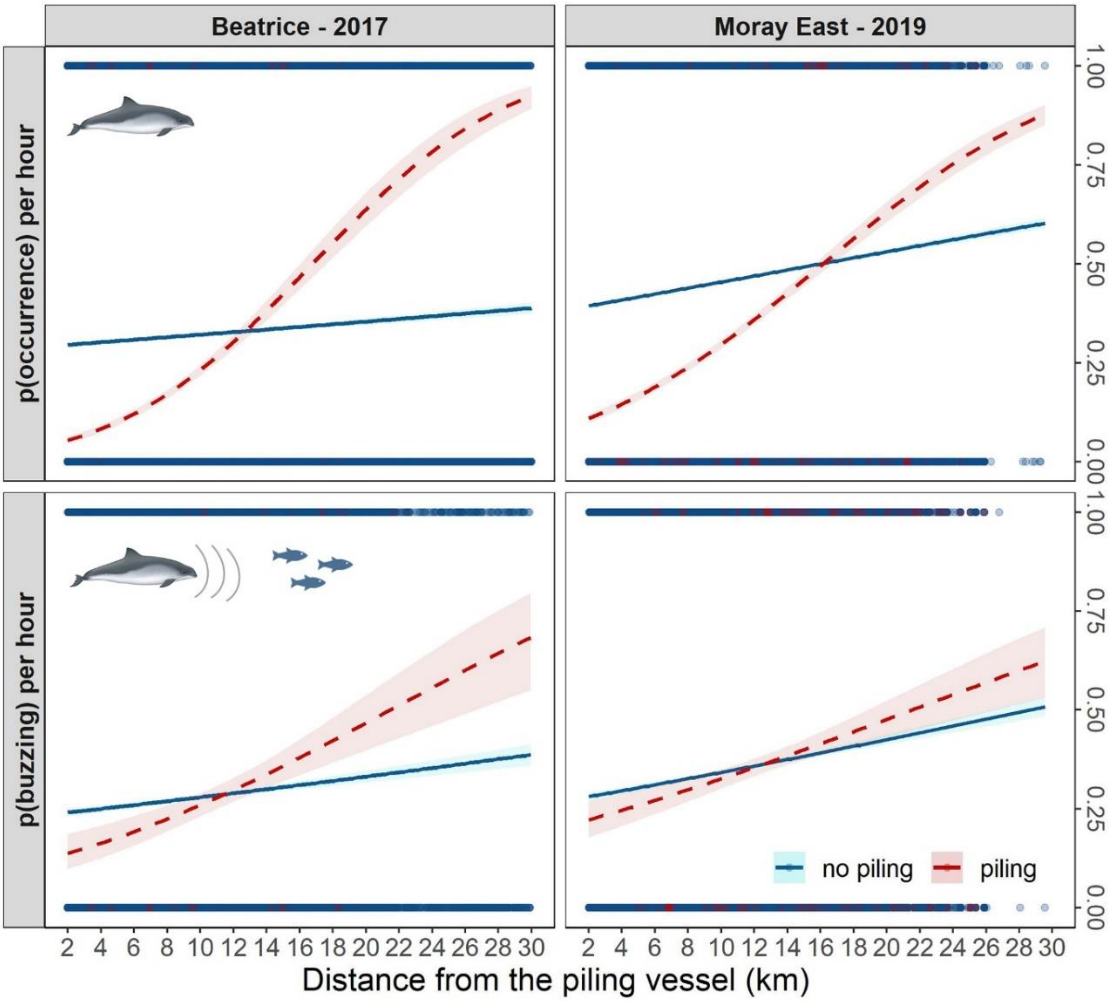


Figure 2.5 The probability of harbour porpoise occurrence and buzzing activity per hour during (dashed red line) and out with (blue line) pile-driving hours, in relation to distance from the pile-driving vessel at Beatrice (left) and Moray East (right); confidence intervals (shaded areas) estimated for uncertainty in fixed effects only; points represent the raw data distribution along the distance gradient during piling (red) and no piling (blue) activities; see the raw data frequency distribution in Figure S 2.6.

Response of porpoises to vessel activity at windfarm construction sites

Figure 2.6A and Figure 2.6B summarise the broad-scale spatial variation in the intensity and density of construction-related vessels across the Moray Firth between 2017 and 2019 (for further information on the overall vessel density and intensity across the Moray Firth between 2017 and 2019, see Figure S 2.5A and Figure S 2.5B). Over this period, median construction-related vessel density was 1.4 vessels.km^{-2} (range 0.06 to 64.8 vessels.km^{-2}) across the Moray Firth. Vessel density was highest in 2019 when both windfarms were under construction. Similarly, the median construction-related vessel intensity, across the Moray Firth, was 2.2 h.km^{-2} (range 0 to 29,006 h.km^{-2}). Most vessels occurred over the windfarm sites, but construction-related vessels also worked along export cable routes and between local ports and harbours, including Wick, Invergordon and Fraserburgh (Figure 2.1B).

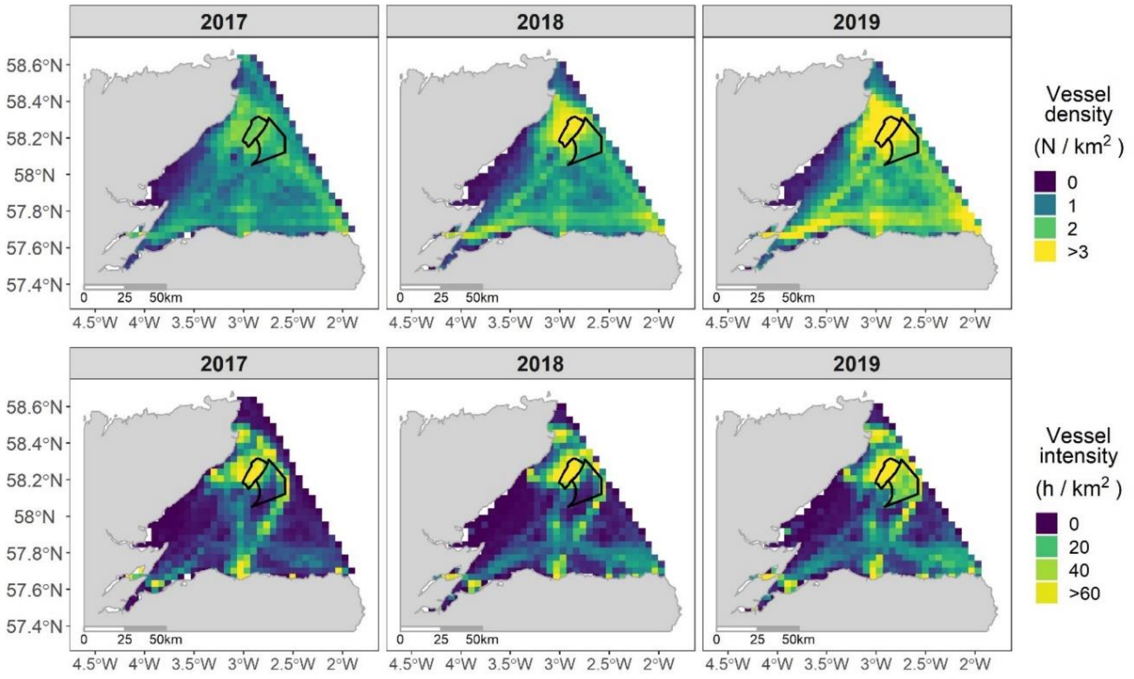


Figure 2.6 A) Construction-related vessel density (number of vessels/km^2) and B) intensity (h/km^2) in the Moray Firth (4 × 4 km grid) between 2017 and 2019; Black lines are the boundaries of the two offshore windfarms in development; the upper limit of both the vessel density and intensity colour scales is greater than 95th percentile.

In addition to the key offshore service vessels used for pile-driving and jacket or turbine installation, construction-related vessel traffic included fishing vessels working as guard vessels, passenger vessels for crew-transfers and some port service craft or unassigned vessels. Details of vessels used in each development are included in Table S 2.1.

Safety zones of 500-1500 m were maintained around structures under construction, and of 50 m around installed structures waiting to be commissioned. Nevertheless, both the Beatrice and Moray East sites remained accessible to fishermen and other third-party vessel traffic throughout their construction. Within the two windfarm sites, the density of fishing, bulk carrier, cargo, and unassigned vessels that were not involved in the construction ranged from 0.15 to 0.21 vessels.km^{-2} between 2017 and 2019. Fishing vessel density decreased at Beatrice in 2018 and 2019 and at Moray East in 2019. However, parallel increases in the intensity of fishing vessels suggested that the fishing vessels that were present spent more time in the area (Table 2.4).

During the 245- and 284-day pile-driving campaigns, the piling vessel was within the windfarm footprint for around 4,090 h (69.5% of the time) at Beatrice and 6,525 h (95.7% of the time) at Moray East. However, it should be noted that the piling vessel at Moray East was jacked-up for most of this time. The total number of hours in which piling occurred was around 437 h (7.4% of the time) at Beatrice and 773 h (11.3% of the time) at Moray East.

Estimates of vessel intensity around each of the passive acoustic monitoring sites were similar in 2017 and 2019 (Figure 2.7A), with the third quantile around 1.21-1.28 min.km^{-2} and a peak between 0.6 and 0.9 min.km^{-2}. Although the shape of the distribution of vessel intensities in 2018 was similar, the third quantile was around 1.78 min.km^{-2}, highlighting that vessel activity at Beatrice was higher during the installation of jackets and turbines. There was also spatial variability in vessel density and intensity between years and sites (Figure 2.7B). In 2017, the higher levels of vessel intensity occurred across Beatrice but in 2018 was more localised around the south-east boundary of the windfarm. In 2019, vessel intensity was spread across the two windfarm sites, but levels of vessel intensity remained highest at Beatrice (Figure 2.7B).

Table 2.4 A) Density and B) intensity of vessels involved or not involved in the construction at Beatrice and Moray East offshore windfarms between 2017 and 2019, grouped by vessel category. A white to red gradient was used with the highest values in vessel density and intensity represented in red.

A	**Construction-related vessels**						**Other vessels**					
Vessel density (N boat / km^2)	**Beatrice**			**Moray East**			**Beatrice**			**Moray East**		
Vessel category	**2017**	**2018**	**2019**	**2017**	**2018**	**2019**	**2017**	**2018**	**2019**	**2017**	**2018**	**2019**
Bulk carrier and cargo vessels	0.02	0.03	0.02	0.01	0.01	0.01	0.07	0.02	0.01	0.17	0.17	0.06
Fishing vessels	0.11	0.06	0.02	0.04	0.03	0.02	0.18	0.05	0.08	0.17	0.19	0.12
Military and law enforcement							0.01			0.01	0.00	
Non-port service craft	0.02	0.02		0.01	0.01		0.02	0.02	0.01	0.03	0.05	0.01
Offshore service vessels	0.08	0.15	0.06	0.04	0.07	0.03	0.03	0.02		0.05	0.03	0.01
Passenger vessels	0.01	0.11	0.10	0.00	0.03	0.04	0.04	0.04	0.02	0.12	0.13	0.02
Port service craft	0.15	0.09		0.05	0.05	0.01	0.02	0.02	0.02	0.05	0.04	0.04
Recreation vessels							0.09	0.06	0.06	0.08	0.09	0.13
Research vessels		0.01	0.02		0.00	0.00						
Tankers							0.02			0.04	0.04	0.02
Unassigned	0.09	0.11	0.05	0.06	0.07	0.04	0.15	0.15	0.02	0.16	0.21	0.08
Total	**0.47**	**0.58**	**0.26**	**0.21**	**0.28**	**0.15**	**0.63**	**0.37**	**0.21**	**0.88**	**0.96**	**0.48**

B	**Construction-related vessels**						**Other vessels**					
Vessel intensity (hour / km^2)	**Beatrice**			**Moray East**			**Beatrice**			**Moray East**		
Vessel category	**2017**	**2018**	**2019**	**2017**	**2018**	**2019**	**2017**	**2018**	**2019**	**2017**	**2018**	**2019**
Bulk carrier and cargo vessels	5.73	11.55	14.82	0.17	2.02	0.76	0.06	10.08	0.36	0.26	1.91	0.06
Fishing vessels	62.14	8.73	0.01	4.67	1.25	4.18	0.09	0.72	3.39	1.72	2.93	13.38
Military and law enforcement							0.00			0.01	0.00	
Non-port service craft	2.68	2.49		0.12	0.28		0.00	0.03	0.01	0.04	1.08	0.04
Offshore service vessels	43.27	194.06	82.53	4.36	36.00	19.41	63.16	1.99		2.09	0.23	0.39
Passenger vessels	0.07	77.28	85.35	0.03	0.45	0.20	0.02	8.03	0.03	0.44	0.34	0.10
Port service craft	62.08	10.47		4.57	0.73	0.01	0.27	0.01	0.23	0.15	0.07	0.29
Recreation vessels							0.05	0.02	0.02	0.11	0.15	0.17
Research vessels		0.04	1.57		0.07	0.33						
Tankers							0.01			0.21	0.23	0.01
Unassigned	34.10	115.42	25.13	18.20	14.22	18.13	0.07	4.72	0.18	7.62	23.10	1.79
Total	**210.07**	**420.05**	**209.41**	**32.13**	**55.01**	**43.03**	**63.75**	**25.60**	**4.23**	**12.65**	**30.04**	**16.23**

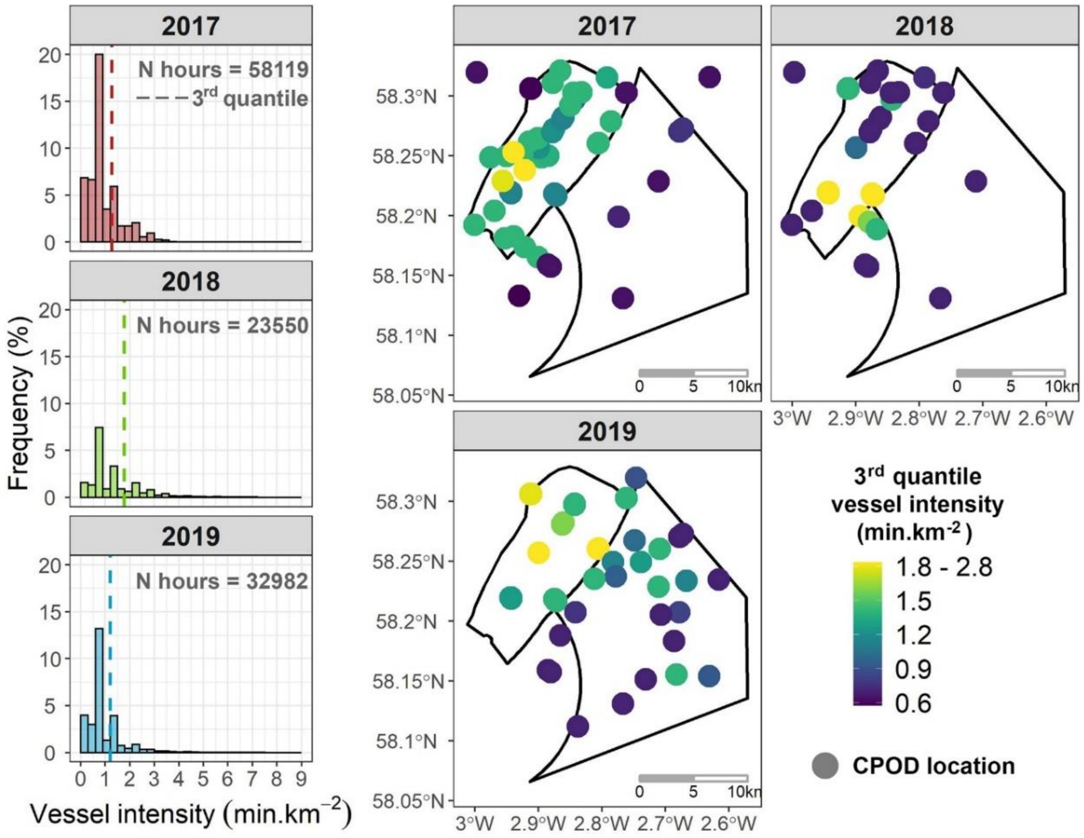


Figure 2.7 A) Histograms of the vessel intensity (min.km^{-2}) per hour in 2017 (pink), 2018 (green), 2019 (blue); dashed lines represent the third quantile of vessel intensity per hour; the sample size is showed by the number of hours monitored per year (N hours); (B) spatio-temporal distribution of the vessel intensity per hour within and around the two offshore windfarms between 2017 and 2019; the colour gradient indicates the annual third quantile of vessel intensity within a 5 km buffer of each CPOD site.

Based on the best fit *vessel effect* model, finer-scale variation in harbour porpoise occurrence within the windfarm sites was explained by the interaction between the vessel intensity and the mean vessel distance from each CPOD site within a 5 km buffer area (Wald test: $\chi^2 = 73.3$, $p < 0.001$) (Figure 2.8A). At a mean vessel distance of 2 km, porpoise occurrence decreased by up to 35.2% as vessel intensity increased, decreasing from 0.37 (95% CI: 0.36-0.39) when vessel intensity was zero (0 min.km^{-2}) to 0.02 (95% CI: 0.01-0.05) for a vessel intensity of 9.8 min.km^{-2}. Porpoise responses decreased as the mean vessel distance increased (-24% at 3 km) until no apparent response was observed at 4 km (+7.2%).

Vessel intensity also had a significant effect on the probability of buzzing (Wald test: $\chi^2 = 110$, $p < 0.001$). Throughout the three years, the probability of detecting buzzes in each hour that porpoises were present decreased by up to 24.5%, from 0.32 (95% CI:

0.32-0.33) when vessels were absent to 0.08 (95% CI: 0.06-0.11) for hours with a vessel intensity of 9.17 min.km^{-2} (Figure 2.8B). Mean distance to the vessel had no significant effect on the probability of detecting buzzes.

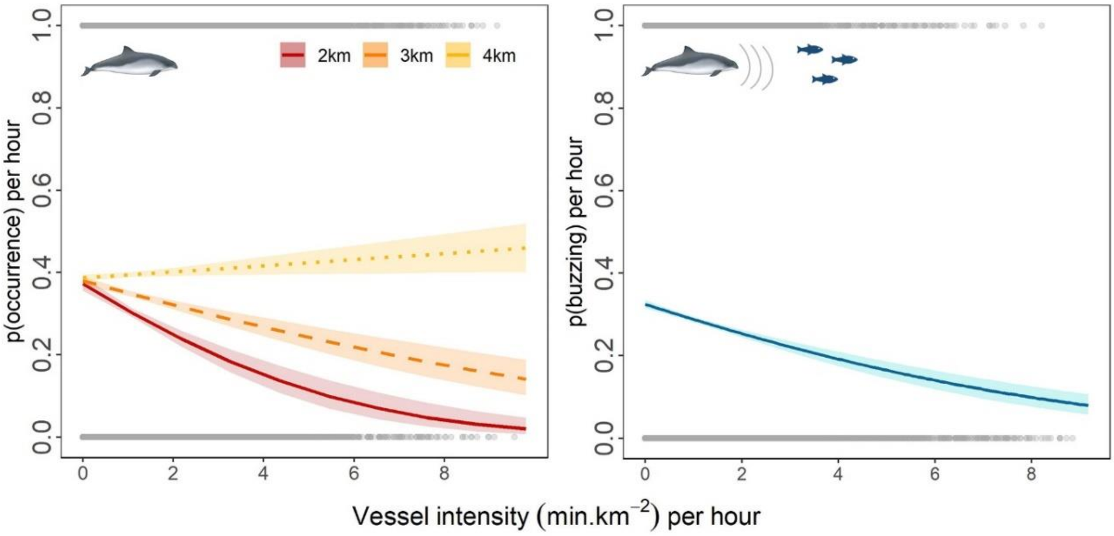


Figure 2.8 A) The probability of harbour porpoise occurrence per hour in relation to vessel intensity per hour at a mean vessel distance of 2 km (red line), 3 km (dashed orange line), 4 km (dotted yellow line); B) the probability of buzzing activity per hour in relation to vessel intensity (blue line); confidence intervals (shaded areas) estimated for the uncertainty in fixed effects only; grey points represent the raw data distribution along the vessel intensity gradient; see the raw data frequency distribution in Figure S 2.7 and Figure S 2.8.

Variation in occurrence and foraging activity of porpoises in relation to noise

Both averaged broadband sound pressure levels in each hour (Wald test: $\chi^2 = 32.9$, $p < 0.001$) and piling occurrence (Wald test: $\chi^2 = 47.2$, $p < 0.001$) had significant effects on the probability of detecting harbour porpoises. Outside piling hours, porpoise detections decreased by 17% as SPL increased, decreasing from 0.43 (95% CI: 0.39-0.48) at 102 dB re 1 µPa to 0.27 (95%CI: 0.27-0.35) at 159 dB re 1 µPa. During piling activities, porpoise occurrence was initially lower (0.17 95% CI: 0.11-0.25 at 102 dB re 1 µPa) and decreased by 9% as SPL increased by 59 dB (0.08 95% CI: 0.05-0.12 at 159 dB re. 1 µPa) (Figure 2.9A).

Similarly, variation in the probability of detecting buzzes was also explained by both SPL (Wald test: $\chi^2 = 23.1$, $p < 0.001$) and piling occurrence (Wald test: $\chi^2 = 14.4$, $p < 0.001$). However, while the probability of detecting buzzes decreased with increasing noise levels in either the presence or absence of piling, porpoises detected during piling exhibited much higher levels of buzzing activity (Figure 2.9B). Outside piling hours, the probability of detecting buzzes decreased by up to 45.9 % as the SPL increased, ranging from 0.47 (95% CI: 0.39-0.56) at 104 dB re. 1 µPa to 0.01 (95% CI: 0-0.04) at 155 dB re. 1 µPa. During piling hours, buzzing occurrence decreased by up to 70.5% as the SPL increased, ranging from 0.89 (95% CI: 0.67-0.98) at 104 dB re. 1 µPa to 0.18 (95% CI: 0.07-0.38) at 155 dB re. 1 µPa (Figure 2.9B).

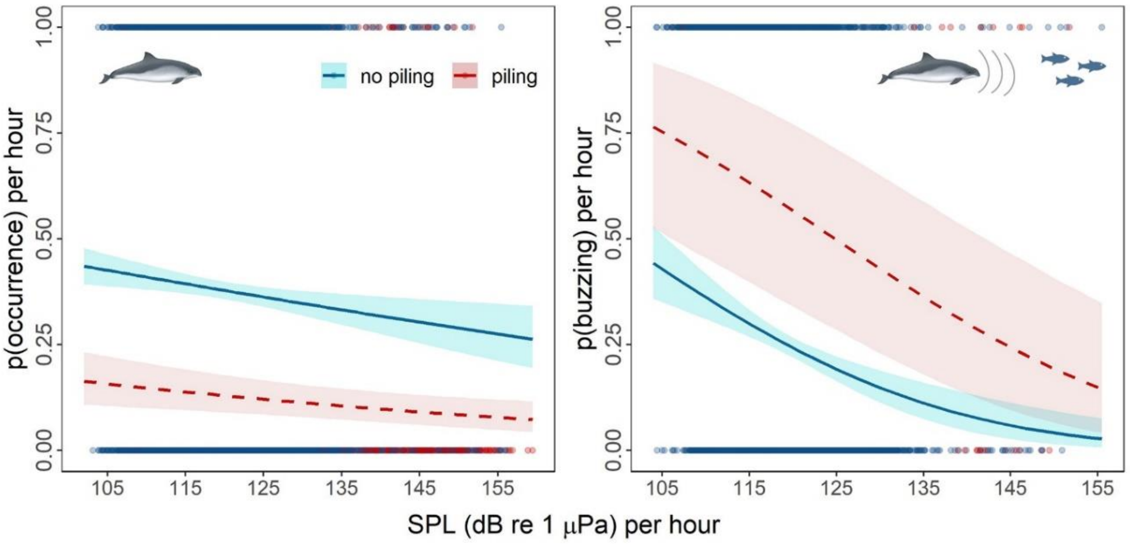


Figure 2.9 The probability of A) harbour porpoise occurrence and B) buzzing activity per hour in relation to the broadband sound pressure level (SPL) per hour during (dashed red line) and outside (blue line) pile-driving hours; confidence intervals (shaded areas) estimated for the uncertainty in fixed effects only; points represent the raw data distribution along the SPL gradient during piling (red) and no piling (blue) activities; see the raw data frequency distribution in Figure S 2.9.

DISCUSSION

Uncertainties exist over the extent to which marine mammal occurrence and foraging activity varies through different phases of offshore windfarm construction. This, in turn, currently constrains efforts to balance the development of renewable energy to meet carbon reduction targets with the need to minimise disturbance to protected wildlife populations. The BACI analyses provide evidence of broad-scale behavioural responses of harbour porpoises both to pile-driving and other construction-related activities (Figure 2.4). In addition, impact gradient analyses show that the magnitude of response varied depending on the activity type and distance from the disturbance source (Figure 2.5), and the cumulative pressure associated with vessels (Figure 2.8) and anthropogenic noise (Figure 2.9). Together, these analyses allowed us to quantify response levels during different construction contexts, while also highlighting that harbour porpoises continued to regularly use these sites throughout the three-year construction period (Figure S 2.4). These findings now provide new data to parameterise energetics and population simulation models (e.g. DEPONS Nabe-Nielsen et al. (2018) and iPCoD Booth et al. (2017)) that can explore potential population-level consequences of these cumulative disturbances.

Changes in porpoise occurrence during the two piling campaigns

As expected from previous studies of harbour porpoise responses to impulsive noise both at this (Thompson et al. 2013; Graham et al 2019) and other (Tougaard et al. 2009a; Brandt et al. 2011; Dähne et al. 2013; Haelters et al. 2015; Nabe-Nielsen et al. 2018) North Sea sites, the BACI analyses demonstrated a significant decrease in porpoise detections when pile-driving occurred at both the Beatrice and Moray East windfarm sites (Figure 2.4). In the BACI analyses, the observed changes in porpoise occurrence and buzzing activity between blocks and monitoring periods may be confounded by the varying sampling effort although similar relationships were found using the impact gradient analyses. In PAM based studies such as this, it is recognised that short-term decreases in acoustic detections could result from animals ceasing to vocalise rather than being displaced. However, given the high energetic requirements and foraging rates of harbour porpoises (Booth 2020), the broader-scale changes observed here are most likely to result from avoidance behaviour leading to lower densities over the impact block. In the German North Sea, the decrease in relative porpoise acoustic detection rates within 10 km of the pile-driving noise source, was associated with an increase in detection rates at 25 and 50 km distance, and matched the lower porpoise density, observed through visual aerial surveys, within 20 km of the noise source (Dähne et al. 2013). Similar findings were observed in response to seismic surveys over a range of 5-10 km in Thompson et al. (2013) and in response to a seal scarer in Brandt et al. (2013b). In these studies, the aerial surveys supported the assumption that porpoises exposed to anthropogenic noise sources such as pile-driving, airgun and acoustic deterrent devices left the ensonified area rather than ceasing vocalising. Consequently, even though this study relied solely on PAM data, decreases in acoustic detections in response to impulsive noise disturbance is likely to result from displacement. Approaches used for assessing the spatial scale of responses to piling have varied across studies. Nevertheless, in this study, the observed responses at distances of 10-15 km at both sites (Figure 2.5) are of a similar order of magnitude to results from the subset of Beatrice sites analysed in Graham et al (2019) and those from other North Sea windfarms (Tougaard et al. 2009a; Brandt et al. 2011; Dähne et al. 2013; Haelters et al. 2015; Nabe-Nielsen et al. 2018). One limitation of using CPODs in these situations is that elevated background noise close to piling locations may affect detection probability (Clausen et al. 2019). When analysing the magnitude of near-field responses to pile-driving (Figure 2.5), I accounted for this by excluding data from all CPOD locations that were within 2 km of the piling vessel.

Although patterns in porpoise detections were similar during the piling campaigns at the two Moray Firth sites, the magnitude of change in porpoise occurrence during the Moray East piling phase was lower than at Beatrice (Figure 2.4). Graham et al. (2019) showed that responses to pile driving noise at Beatrice diminished through 2017. The scale of response at Moray East in 2019 may therefore be smaller due to the increased tolerance of individuals remaining in the area (Bejder et al. 2009). However, little is known of broader-scale movement patterns of North Sea porpoises (see Sveegaard et al. (2011)) and it is not currently possible to follow individual porpoises over multiple years. Thus, it remains unclear whether or not displaced individuals returned to impacted areas or whether porpoises exposed in 2017 were still present in subsequent years (Graham et al. 2019). Alternatively, variation in the magnitude of response in the two piling phases could result from local changes in habitat quality or other differences in the nature of the disturbance during pile-driving. For example, the impact block was large, and included both windfarms. Thus, during the 2019 Moray East piling phase, the northern part of the impact block also contained 86 operational structures within the Beatrice windfarm. Harbour porpoise occurrence in this part of the impact block could therefore have increased, as seen in the Egmond aan Zee windfarm in Dutch waters (Scheidat et al. 2011). Scheidat et al. (2011) suggest that such changes could result from increases in prey due to artificial reef effects within established windfarms, or because most shipping is excluded from Dutch windfarm sites; potentially allowing porpoises to shelter from vessel disturbance. However, the analysis of AIS data suggests that a sheltering effect is unlikely in the Moray Firth as there continued to be high levels of windfarm and third-party traffic over the Beatrice site in 2019 (Figure 2.6). Finally, the two piling campaigns used different installation infrastructure that may explain observed differences between the responses in 2017 and 2019. The piling vessel at Beatrice used eight anchors, requiring the presence of additional pilot and anchor-handling vessels, while an independent jack-up piling vessel was used at Moray East. This study design did not enable me to discriminate between the fine and meso-scale spatio-temporal impact of this diverse range of construction-related activities, but these findings highlight the need for further work to explore how different pile installation techniques may affect the scale of response.

Changes in porpoise activity during the two piling campaigns

Building on previous work that has focussed on displacement during pile-driving, I also used information from the echolocation click characteristics to explore broad-scale changes in the activity of those porpoises that continued to use the windfarm sites. During an earlier seismic survey in this area, harbour porpoise occurrence decreased close to the noise source (Thompson et al. 2013), and animals remaining in exposed areas also exhibited a decrease in buzzing activity (Pirotta et al. 2014a). In the present BACI study, porpoises that remained in the impact block during pile-driving at Beatrice in 2017 also reduced their buzzing activity by 4.2% compared to baseline but, in contrast, buzzing activity during Moray East piling in 2019 was higher than baseline (Figure 2.4). As discussed in relation to differences in the magnitude of displacement during the two piling campaigns, differences in buzzing activity of porpoises remaining in the impact area may also result from local changes in habitat quality. The introduction of hard substrates (e.g. jacket foundations and scour protection) are likely to have enhanced the fine-scale habitat and changed fish assemblages, potentially increasing the prey availability for opportunistic and generalist feeders such as porpoises (Santos and Pierce 2003) and explaining higher buzzing activity during this period. Better understanding of any reef effects following construction is now urgently required so that potential ecosystem benefits can be integrated into an evaluation of the lifetime cumulative impacts of windfarm construction and operation on these populations (King et al. 2015; Booth et al. 2017; Nabe-Nielsen et al. 2018).

Within the impact area, gradient analyses of data collected through both piling phases also suggest that the probability of detecting buzzes decreased by 54% with decreasing distance from the piling vessel (Figure 2.5), and by 74% as hourly RMS SPL increased from 105 to 155 dB re. 1 µPa (Figure 2.9B). Thus, individuals remaining nearest exposed areas did spend less time buzzing, while porpoises displaced from exposed areas increased their buzzing activity, potentially compensating for lost foraging opportunities or increased energy expenditure. During extended periods of disturbance, porpoises must make trade-offs between fleeing, either permanently or temporarily, or remaining in areas that have a higher risk of disturbance or predation. Baseline distribution patterns suggest that the vicinity of both impact and reference areas represent high-quality feeding habitat (Brookes et al. 2013), and fleeing the area may incur high energetic costs and the risk of spending time in lower-quality habitat. Individual responsiveness to anthropogenic disturbances is therefore likely to be context-dependent and related to animal fitness (van Beest et al. 2018). Theoretically, individuals in poorer condition are less likely to leave high-quality habitats after a disturbance, as the energetic cost and risk of missing foraging opportunities may be too high (Gill et al. 2001; Beale and Monaghan 2004; van Beest et al. 2018). In this seascape of fear, marine mammals can alter activity budgets according to perceived levels of predation risk (Wirsing et al. 2008), and are expected to perceive anthropogenic disturbance, such as pile-driving and vessel activity, as a form of predation (Frid and Dill 2002). Consequently, porpoises in the vicinity of construction activities may reduce their buzzing activity as they adjust activity budgets to spend more time avoiding noise sources and less time engaged in foraging and/or social activities (Pirotta et al. 2014a; Wisniewska et al. 2018). Decreases in buzzing activity could also be explained by reduced prey availability or foraging performance as a result of the displacement or changed behaviour of prey species in response to anthropogenic noise (Hassel et al. 2004; Mueller-Blenkle et al. 2010; Herbert-Read et al. 2017). Further field studies on the behavioural responses of different prey to pile-driving activities are required to understand the extent to which any spatio-temporal variation in the local prey availability and abundance may have indirect consequences on individual porpoise fitness (Hassel et al. 2004; Mueller-Blenkle et al. 2010).

Changes in porpoise occurrence and activity in relation to vessels and other construction activity

During the turbine installation phase, the broad-scale BACI analysis showed that porpoise occurrence was also significantly lower at the impact block than during the baseline, even though no piling activities occurred (Figure 2.4). During this period (July-October 2018), various construction activities such as jacket foundation, turbine and cable installation occurred simultaneously at different locations within the windfarm, leading to high levels of vessel traffic (Figure 2.7A and Figure 2.7B). Additionally, gradient analyses showed that the probability of detecting porpoises within the site decreased by up to 35.2% as vessel intensity increased, and as distance to the nearest vessel decreased (Figure 2.8A). Previous experimental studies demonstrated that captive harbour porpoises displayed strong behavioural responses when exposed to low levels of medium to high frequency vessel noise (Dyndo et al. 2015). Many vessels emit high frequency noises that overlap with frequency bands biologically relevant for porpoises, which may lead to acoustic masking and/or elicit adverse behavioural responses (Hermannsen et al. 2014). In my study, increased vessel activity led to a significant decrease in porpoise acoustic detections and activity (Figure 2.8A and Figure 2.8B) at distances of up to 4 km (Figure 2.8A). However, studies using sound and movement recording tags that can detect finer-scale responses highlight that porpoise foraging may be disrupted at greater distances of up to 7 km (Wisniewska et al. 2018).

Using only data collected in the absence of pile-driving, I found that the probability of detecting porpoises decreased by up to 17% as the broadband sound pressure levels increased by 57 dB (Figure 2.9A). Increased levels of broadband noise emitted during other construction-related vessel activities may reduce the porpoise detection probability of CPODs (Clausen et al. 2019). However, to reduce the risk of vessel noise masking ultrasonic click detections and so the probability of false-negative detections, I discarded any hours with less than 60 minutes logged and during which vessels were within 1 km of CPODs (as Pirotta et al. (2014a)), and additionally chose a coarse binary metric (i.e. presence/absence per hour) (Williamson et al. 2016). Thus, in this case, the decline in porpoise detections, in the absence of piling activities, is unlikely to be caused by a reduction in the effective detection area of the devices. In contrast, the decline in porpoise detections suggests that porpoises have exhibited a behavioural response to high levels of background noise associated with vessel and construction activities. In Wisniewska et al.’s (2018) study, tagged harbour porpoises responded to fast ferry passages by making deeper dives, increasing swimming effort, and ceasing echolocation and foraging for several minutes. Although these individuals lived in highly trafficked coastal waters, they did not seem to have habituated to vessel noise (Wisniewska et al. 2018). Similarly, throughout the three-year monitoring, buzzing activity decreased by up to 24.5% as the vessel intensity increased in the study area (Figure 2.8B) and by up to 45.9% as the hourly RMS sound pressure levels increased from 104 to 155 dB re 1µPa (Figure 2.9B). However, for the same levels of broadband noise, buzzing activity appeared to be higher during rather than in the absence of piling activities (Figure 2.9B). This increase in porpoise buzzing activity during piling may be indicative of behavioural changes in echolocation activity in response to noise (the Lombard effect). Harbour porpoises may increase the signal level of their clicks or the signal repetition rate (Branstetter et al. 2018) to compensate for the increased noise levels during social interactions (Sorensen et al. 2018) or foraging activity. Alternatively, adverse effects of piling noise on prey (e.g. Herbert-Read et al. (2017)) may benefit predators by locally increasing prey availability and/or enhancing their foraging performance. Either way, these results highlight how chronic exposure to regular vessel activity and associated levels of anthropogenic noise could influence the foraging and/or social activity of those individuals which continue to use offshore construction sites during the pile-driving phase.

Vessel-tracking data provide a robust measure of the spatial distribution of windfarm construction vessels which are legally required to carry AIS. However, there are several reasons why these data may not fully capture variation in the soundscapes affecting species such as harbour porpoises and their prey. In coastal areas, many recreational vessels without AIS dominate the anthropogenic soundscape (Hermannsen et al. 2019). This is less likely to be an issue offshore but reports from guard vessels indicate that local fishing boats without active AIS commonly used areas over and around the construction sites throughout the study period. Furthermore, although information on construction vessel locations was available, detailed information on variation in the activity of those vessels, which could affect their acoustic signature, was not. This could be particularly important for construction vessels that jacked up to install jacket foundations and turbines, and vessels which periodically used dynamic positioning. Greater understanding of how acoustic signatures vary between vessels, and in relation to speed or activity, could in future help identify ways in which vessel management plans could reduce broader scale disturbance during windfarm construction and operation.

In the absence of more detailed information on the acoustic signatures or activities of vessels detected using AIS, the recordings of broadband noise at three sample locations provide a valuable measure of broad-scale variation in noise exposure to animals through construction phases, and opportunities for comparison with other study systems (Hermannsen et al. 2014). Even here, though, these measures may be biased by proximity to particular vessels. To characterise this variation, I used an unweighted RMS SPL based on a sampling rate of 48 kHz. These frequencies are appropriate for characterising long-term variation in shipping and pile-driving noise (Merchant et al. 2012; Merchant et al. 2014; Thompson et al. 2020), and analysis of a higher sample rate recording at the site (Figure S 2.10) indicated that almost all of the acoustic energy from both pile driving and shipping noise was contained below 24 kHz (the highest acoustic frequency that can be measured at a sampling rate of 48 kHz). However, these 48 kHz recordings do not capture those higher frequencies that may be particularly important to porpoises (Tougaard et al. 2015). More focussed investigation of porpoise behavioural responses to vessel noise would require an increase in sampling rate and focus on biologically significant spectral bands by using audiogram weighted SPL (Dyndo et al. 2015; Tougaard et al. 2015).

Management Implications

The planned expansion of offshore windfarms to meet decarbonisation targets must proceed within frameworks for safeguarding protected wildlife populations and minimising cumulative environmental impacts (Le Lièvre 2019; Thompson et al. 2020). Efforts to understand and mitigate impacts on marine mammals have focussed on the effects of impulsive noise produced during pile-driving. Whilst pile-driving does produce the highest amplitude noise, active piling occurred for < 10% of the time in the 9 to 10-month piling phases at Beatrice and Moray East. Whilst responses to these short but intense periods of impulsive noise sources are of greater magnitude, I showed that harbour porpoise occurrence and buzzing activity also decreased in response to more chronic exposure to vessel traffic throughout construction. Further disturbance may also be expected from routine operation and maintenance vessels, although this may be offset by benefits resulting from the creation of new reef habitat. Further understanding of the relative importance of these different disturbance sources is now required to assess the broader scale cumulative impact of construction, operation and decommissioning over the life cycle of an offshore windfarm. These data can now be integrated into existing tools (e.g. Nabe-Nielsen et al. (2018)) to explore the scale of these different disturbance impacts at both individual and population levels. This could then provide a framework that could be used by policy makers to explore how cumulative impacts can be minimised by combining existing mitigation measures to reduce piling impacts with other regulatory measures to manage vessel traffic and other maritime activities occurring in or around construction sites.

APPENDIX

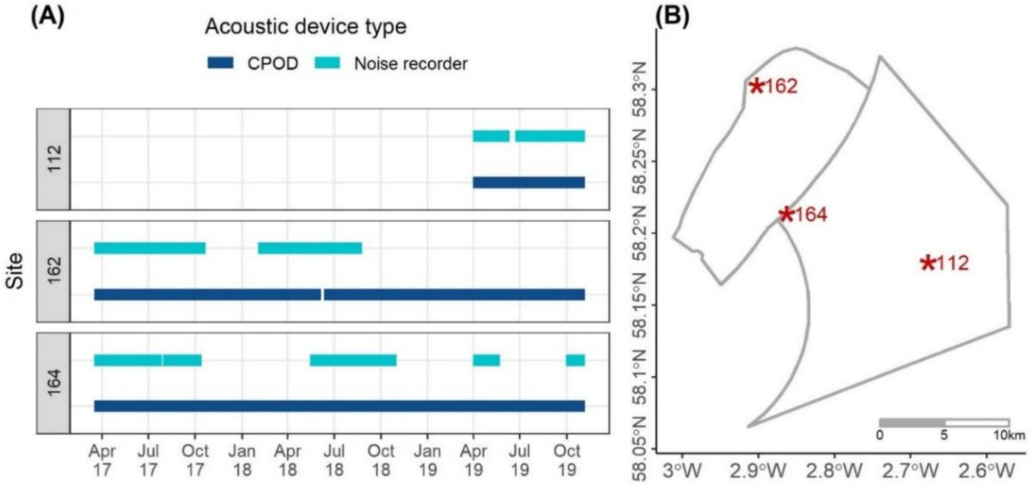


Figure S 2.1 A) Passive Acoustic Monitoring timeline at three long-term monitoring sites to detect echolocation clicks with CPODs (in dark blue) and record underwater broadband noise levels with noise recorders (in light blue); B) Noise recorders locations (in red) within the two offshore windfarms (grey line)

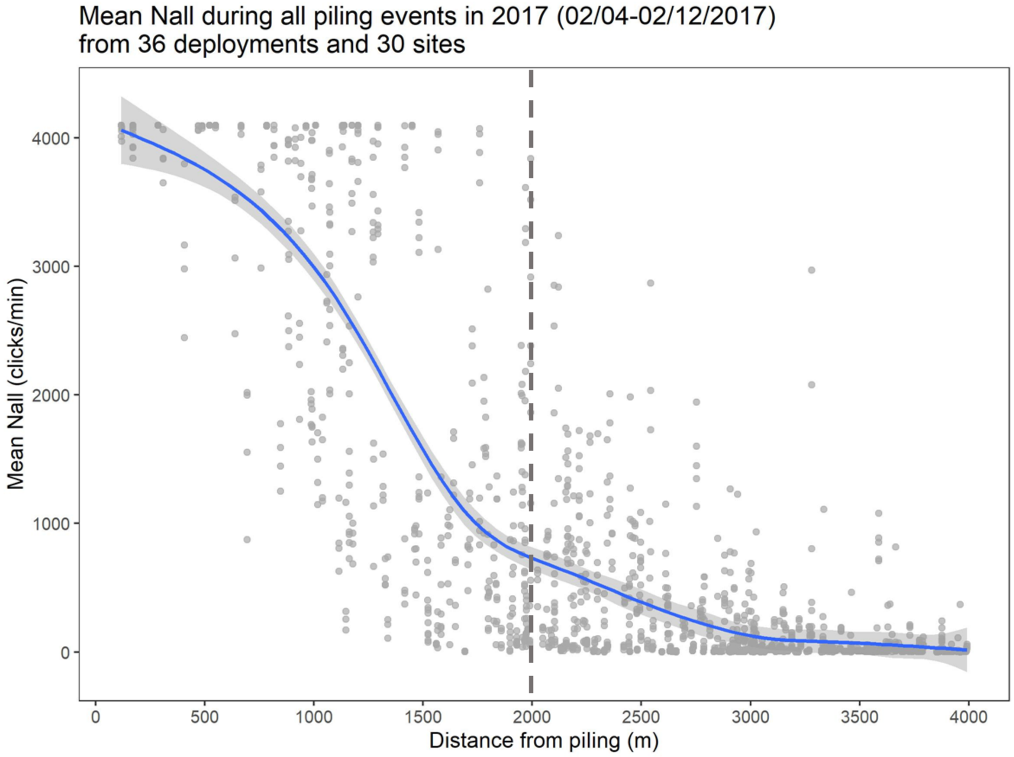


Figure S 2.2 Mean number of clicks per minute (Nall) (grey dots) at four sites in relation to distance from pile-driving locations for all pile-driving events at Beatrice offshore windfarm in 2017; the blue curve has been automatically fitted to the raw data using a loess smoother and shaded areas represent uncertainty; the grey dotted line represents the distance (2 km) from pile-driving locations at which I estimate CPOD saturation to be unlikely.

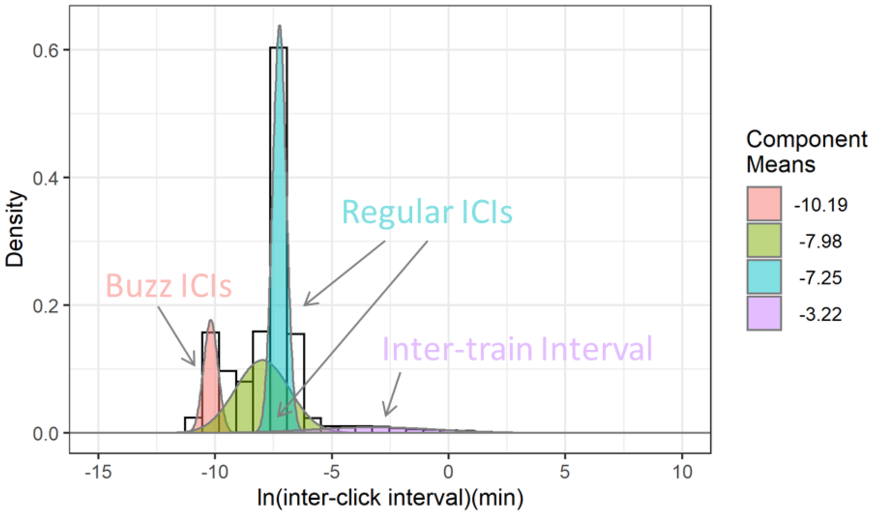
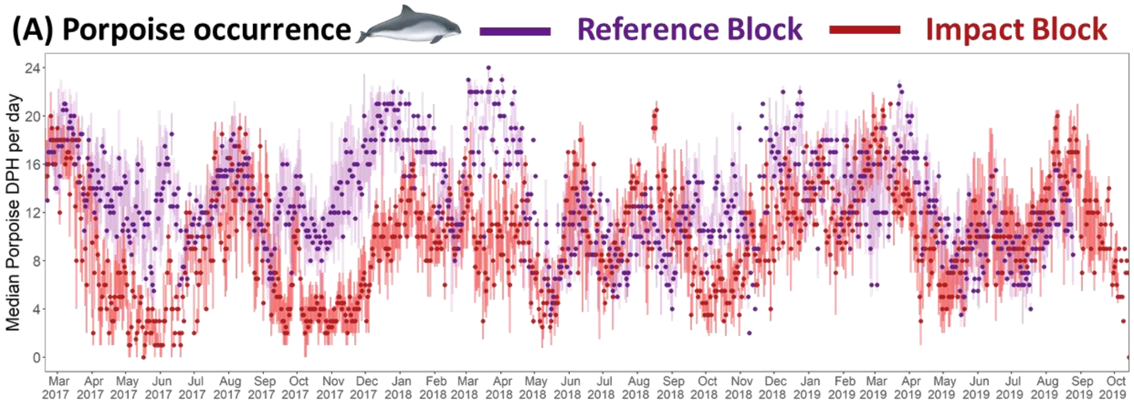


Figure S 2.3 Example of the results of a Gaussian mixture model using four components for one CPOD Inter-Click Interval (ICI) time series, following Pirotta et al. (2014). The buzz ICIs are represented in pink, the regular ICIs in blue and green, the inter-train intervals in purple. The x-axis represents the natural logarithm of the ICIs in minutes.

(A) Porpoise occurrence



(B) Buzz occurrence

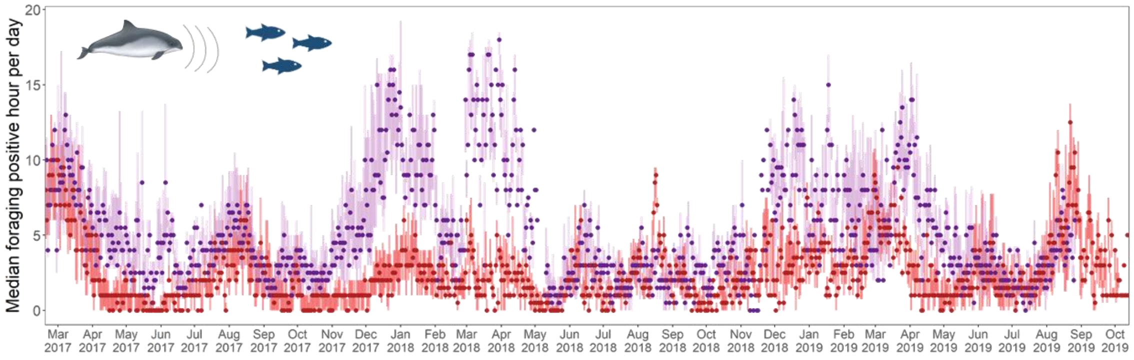


Figure S 2.4 Median harbour porpoise A) detections per hour (DPH) and B) foraging positive hours per day within the reference (in purple) and impact (in red) blocks between March 2017 and October 2019; the dots represent the median and the lines represent the first and third quantile

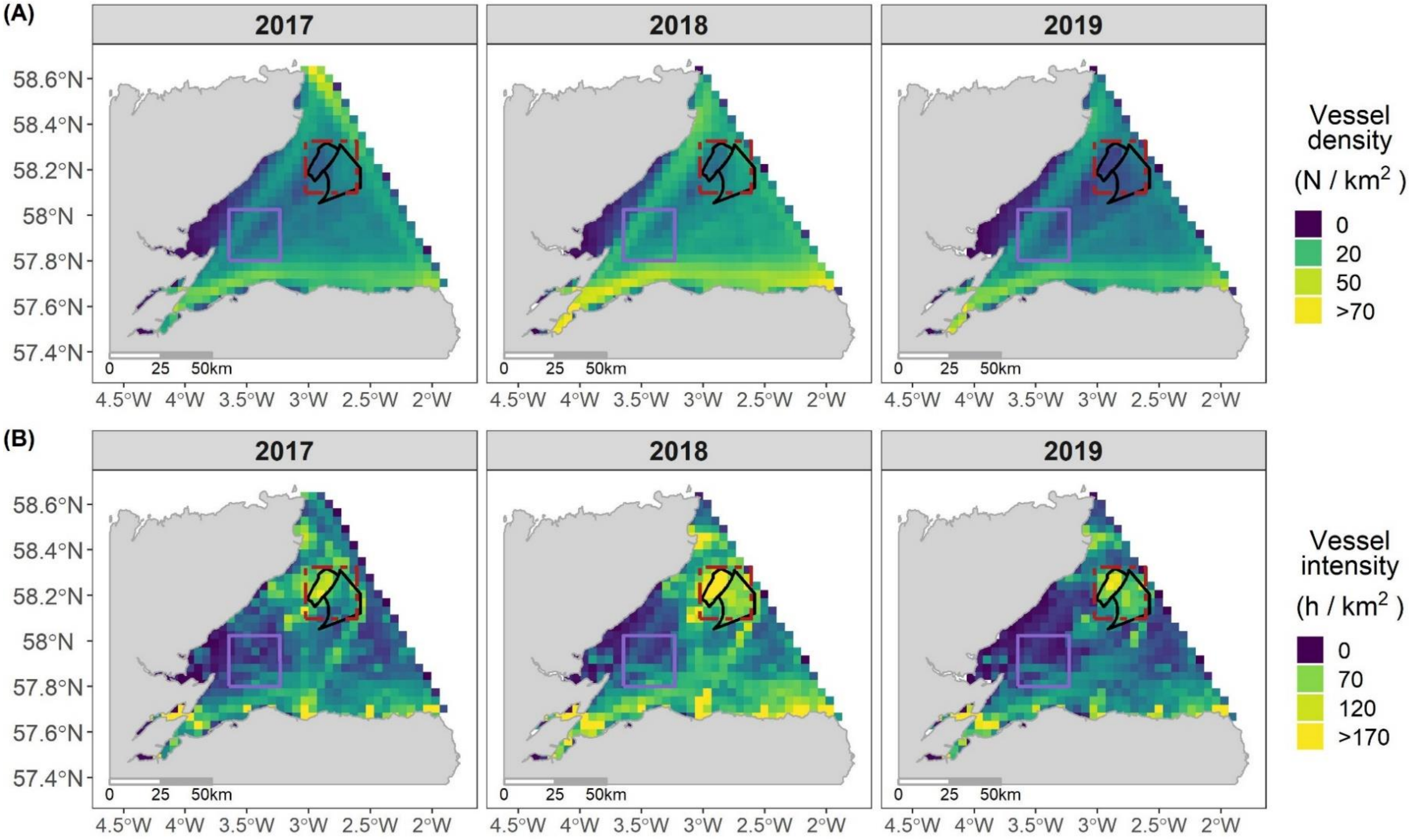


Figure S 2.5 Vessel A) density and B) intensity in the Moray Firth (4 x 4 km grid) between 2017 and 2019; the Beatrice and Moray East offshore windfarms are represented in black in the impact block (dashed red line) and the reference block is in purple.

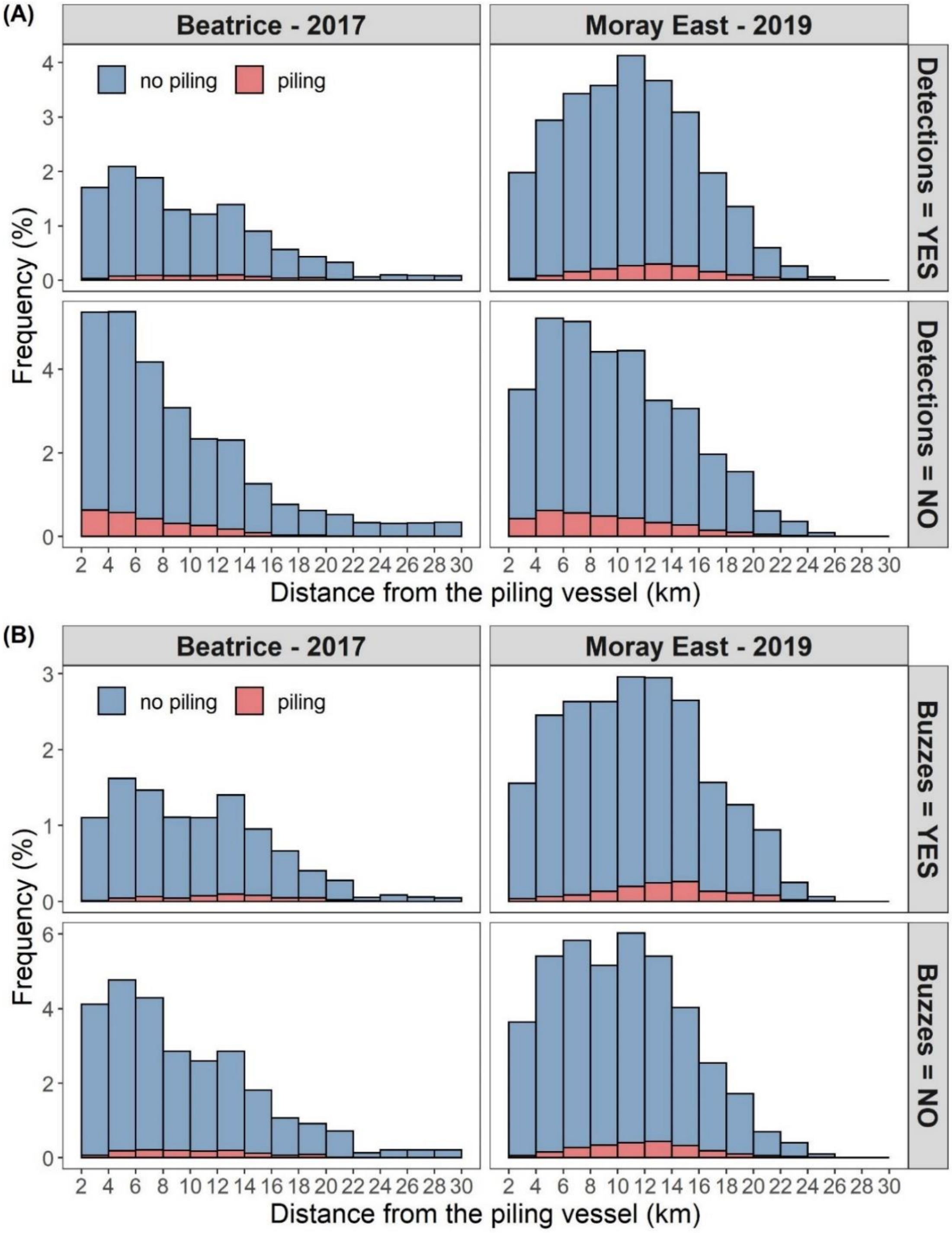


Figure S 2.6 Frequency distribution of the distance from the piling vessel (km) during (in red) and outside (in blue) piling hours, in presence-absence of harbour porpoise A) echolocation click detections and B) buzzing activity, between 2017 and 2019 at the Beatrice and Moray East offshore windfarms.

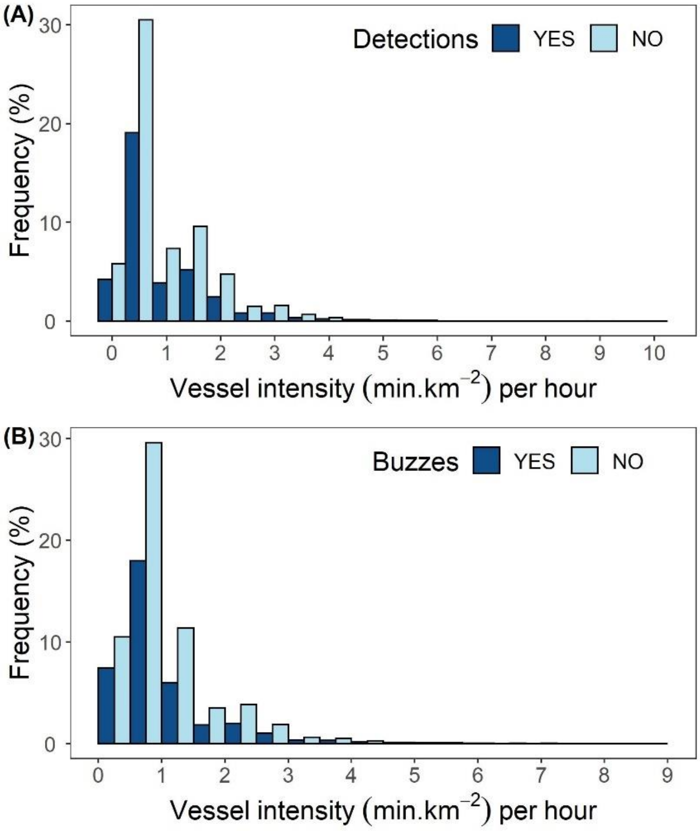


Figure S 2.7 Frequency distribution of the vessel intensity (min/km^2) per hour, in presence (dark blue) or absence (light blue) of harbour porpoise A) echolocation click detections and B) in presence (dark blue) or absence (light blue) of buzzes, between 2017 and 2019 at the Beatrice and Moray East offshore windfarms.

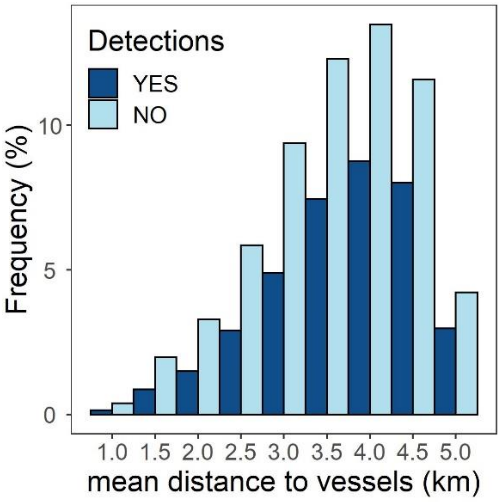


Figure S 2.8 Frequency distribution of the mean distance between any vessels and each CPOD per hour, in presence (dark blue) or absence (light blue) of harbour porpoise echolocation click detections between 2017 and 2019 at the Beatrice and Moray East offshore windfarms.

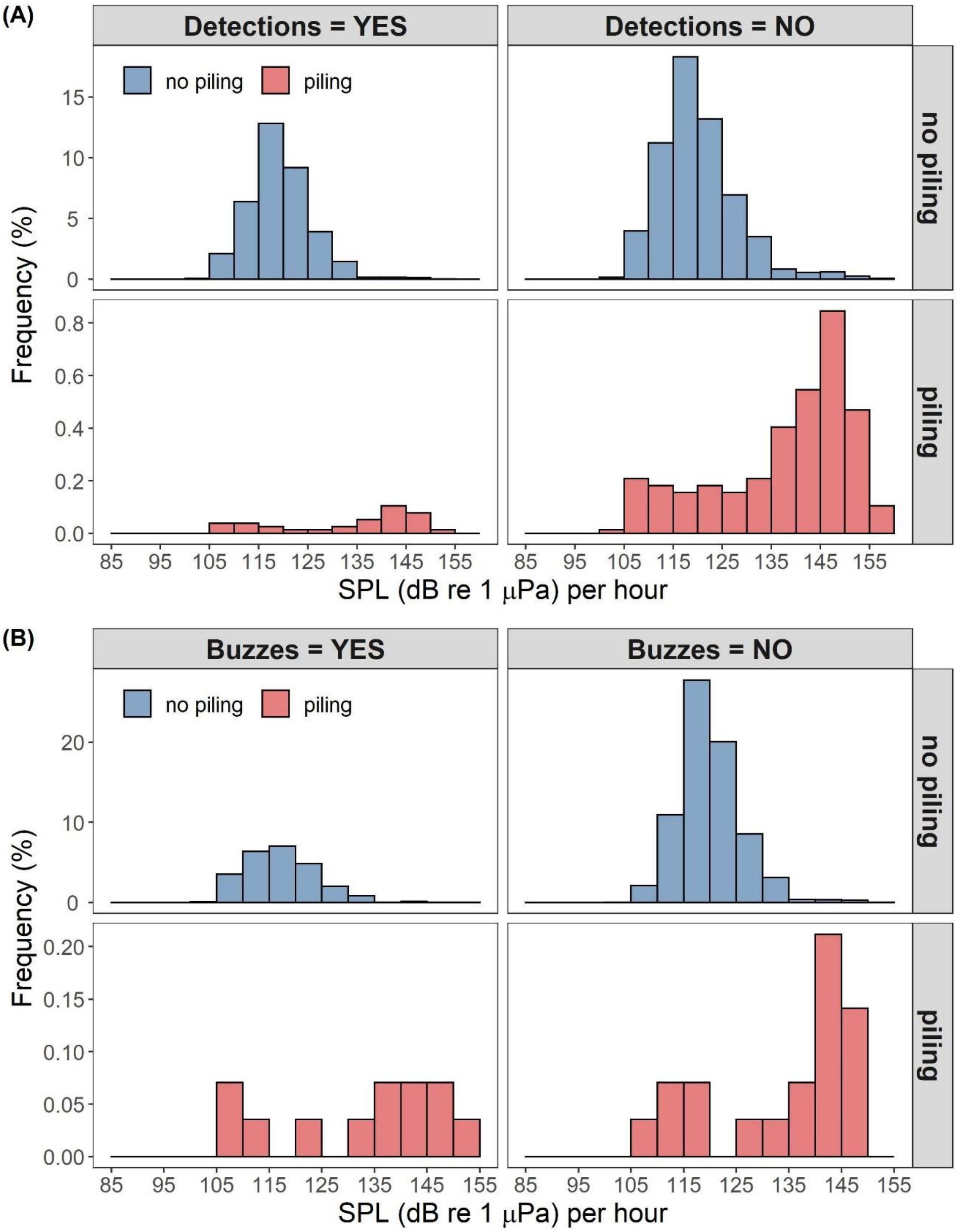


Figure S 2.9 Frequency distribution of the broadband sound pressure level (SPL) per hour during (in red) and outside (in blue) piling hours, A) in presence-absence of harbour porpoise echolocation click detections and B) buzzing activity, between 2017 and 2019 at the Beatrice and Moray East offshore windfarms.

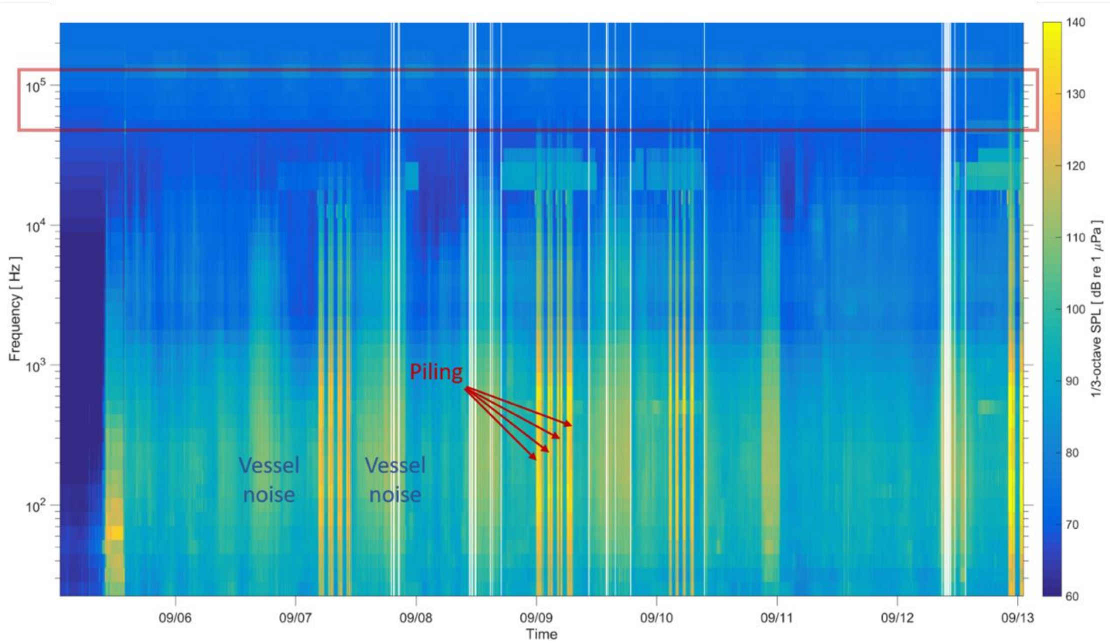


Figure S 2.10 1/3 octave band spectrogram to highlight variability in noise levels between 5 and 13 September 2017 at Beatrice offshore windfarm, 60-s resolution; at high frequencies (red box) both piling and vessel noise are below ambient noise levels.

Table S 2.1 Date of the key construction activities at Beatrice and Moray East offshore windfarms between April 2017 and February 2020

Construction Activity	***Windfarm***	***Time-period (dd/mm/yyyy)***
Piling	Beatrice	02/04/2017 - 02/12/2017
Jacket Foundation Installation	Beatrice	13/08/2017 - 11/07/2018
Turbine Installation	Beatrice	14/07/2018 - 14/05/2019
Piling	Moray East	20/05/2019 - 27/02/2020

Table S 2.2 Metadata of the noise recorder deployments used in the analyses

Deployment	***Location***	***Device Type***	***Recording (ON/TOTAL) period (mins)***	***Pre-amp gain***	***Sample rate (kHz)***	***Lat***	***Long***	***Date data started***	***Date data ended***	***Depth (m)***
760	162	SoundTrap	10/30	HIGH	96	58.3033	-2.9019	17/03/2017	21/05/2017	62
761	164	SoundTrap	10/30	HIGH	96	58.2139	-2.8615	17/03/2017	21/05/2017	40.9
790	164	SoundTrap	10/30	HIGH	96	58.2137	-2.8620	21/05/2017	28/07/2017	41
791	162	SoundTrap	10/30	HIGH	96	58.3030	-2.9019	21/05/2017	31/07/2017	63
887	162	SoundTrap	10/30	HIGH	96	58.3029	-2.9021	31/07/2017	22/10/2017	63
888	164	SoundTrap	10/30	HIGH	96	58.2138	-2.8612	30/07/2017	13/10/2017	41.1
917	162	SM2M	10/60		96	58.3030	-2.9013	02/02/2018	23/06/2018	62
958	162	SoundTrap	10/30	HIGH	96	58.3032	-2.9010	11/06/2018	25/08/2018	62
959	164	SM2M	10/60		96	58.2136	-2.8616	15/05/2018	31/10/2018	40
987	112	SoundTrap	No duty cycle	HIGH	48	58.1798	-2.6754	31/03/2019	11/06/2019	51
1032	164	SoundTrap	No duty cycle	HIGH	48	58.2136	-2.8613	31/03/2019	23/05/2019	41
1041	112	SM2M	10/60		96	58.1798	-2.6756	22/06/2019	06/11/2019	51

Table S 2.3 Construction-related vessel information based on AIS data, Marine Traffic and Metcalfe et al. (2018)

NAME	MMSI	IMO	CALL SIGN	VESSEL TYPE NAME	VESSEL CATEGORY BASED ON METCALFE ET AL. (2018)	VESSEL TYPE DETAILED	FUNCTION DURING WINDFARM CONSTRUCTION	LENGTH OVERALL X BREADTH EXTREME	GROSS TONNAGE	WEB-LINK
APOLLO	253586000	9769764	LXBP	Other Type	Unassigned	Offshore Supply Ship	Offshore Heavy Lift DP2 jack up Vessel	89.28 x 42.02 m	10510	link
ARAMIS	257167000	0	LDEH	Diving ops	Offshore service vessels	Dive vessel	Dive Support Vessel	16 x 5 m	-	link
ARTEMIS	235006582	8644802	2SNV	-	Unassigned	Trawler	Guard Vessel	23.13 x 6 m	131	link
ATLANTIC TONJER	374032000	7008025	MLPD5	Fishing	Fishing vessels	Multipurpose Offshore Vessel	Multipurpose Offshore Vessel; Boulder clearance	80.77 x 18 m	3349	link
ATLANTIS	205231000	9139139	ORKN	-	Unassigned	Motor Hopper	Motor Hopper Vessel - rock placement operations	80.06 x 22.06 m	1998	link
ATLAS WY 101	235001670	8792415	MPUD 3	Fishing	Fishing vessels	Fishing Vessel	Guard Vessel	20 x 6 m	-	link
BKM 103	212492000	0	5BFL2	Other Type	Unassigned	Tug	Multicat Tug	26.5 x 11 m	185	link
BLUE CASTOR	220294000	6403424	OUKR2	Other Type	Unassigned	Pollution Control Vessel	Pollution Control Vessel - Pre-Lay Grapnel Run	46.02 x 7.95 m	720	link
BOA SUB C	257342000	7814656	MJLF4	Fishing	Fishing vessels	Multipurpose Offshore Vessel	Multipurpose Offshore Vessel; Soil plug removal /Grouting	138.5 x 30.64 m	16562	link
BOUNTIFUL	232004846	9656644	LAFR8	Other Type	Unassigned	Fishing Vessel	Guard Vessel	22 x 7 m	-	link
BREMEN FIGHTER	304742000	9321287	V20Y1	Tug	Port service craft	Tug	Heavy Lift Vessel	48.81 x 14.06 m	1262	link

NAME	MMSI	IMO	CALL SIGN	VESSEL TYPE NAME	VESSEL CATEGORY BASED ON METCALFE ET AL. (2018)	VESSEL TYPE DETAILED	FUNCTION DURING WINDFARM CONSTRUCTION	LENGTH OVERALL X BREADTH EXTREME	GROSS TONNAGE	WEB-LINK
BUGSIER 10	218321000	9429560	DFWB2	Tug	Port service craft	Tug	Barge Tow Vessel	32 x 12.6 m	485	link
CENTAURUS	305820000	9433755	V2ED3	Tug	Port service craft	Anchor Handling Vessel	Barge Tow Vessel	48.81 x 14.06 m	1262	link
C-FENNA	232008023	9675963	MBAH3	Tug	Port service craft	Utility vessel	Buoy maintenance - cabling activities	26.48 x 11 m	255	link
C-ODYSSEY	235088132	9636307	2ETW7	Other Type	Unassigned	Utility vessel	trenching/boulder moving - Utility Vessel	25.55 x 11 m	150	link
CORAL WIND	235086491	0	2EMX8	-	Unassigned	Reserved	Workboat	14 x 5 m	-	link
EDT KENNEDY	210779000	8205620	HOBO	Other Type	Unassigned	Offshore Supply Ship	DP 2 ROV Support Vessel	75 x 17.25 m	2948	link
EGESUND	219012959	9059248	OWEQ2	Tug	Port service craft	Tug	Towing Vessel	22.5 x 7.7 m	136	link
ERACLEA	247278500	9499656	IITX2	Tug	Port service craft	Tug	Barge Tow Vessel	50 x 15 m	1397	link
FLINTSTONE	245861000	9275153	LAGE8	Other Type	Unassigned	Pipe Burying Vessel	Pipelayer/ Boulder Clearance and Scour Protection installation	154.6 x 32.23 m	21710	link
FORTH CONSTRUCTOR	235004217	0	GXAD	Tug	Port service craft	Work Vessel	Tunnelling and seabed preparation	28.5 x 9.5 m	265	link
FORTH WARRIOR	235116011	9803742	2JHR8	Tug	Port service craft	Tug	boulder moving - Utility Vessel	27 x 12 m	-	link
GV MORNING DAWN	232253000	8701416	MHEU3	Wing in ground	Unassigned	Standby Safety Vessel	Guard Vessel - Standby Safety Vessel	45 x 9 m	499	link
GENESIS	235008110	0	MGGT9	Other Type	Unassigned	Fishing Vessel	Guard Vessel	30.23 x 6 m	298	link

NAME	MMSI	IMO	CALL SIGN	VESSEL TYPE NAME	VESSEL CATEGORY BASED ON METCALFE ET AL. (2018)	VESSEL TYPE DETAILED	FUNCTION DURING WINDFARM CONSTRUCTION	LENGTH OVERALL X BREADTH EXTREME	GROSS TONNAGE	WEB-LINK
GV SEAGULL	233714000	9112545	MVBO2	Other Type	Unassigned	Fishing Vessel	Guard Vessel	30 x 8 m	349	link
GV HEATHERBELLE	235004610	0	MAEK9	-	Unassigned	Trawler	Guard Vessel	20.87 x 6 m	127	link
GV MOREMMA	235007170	0	MKHC6	Fishing	Fishing vessels	Other	Guard Vessel	15 x 5 m	-	link
GV REPLENISH	235055180		5BAJ4	Other Type	Unassigned	Fishing Vessel	Guard Vessel	24.45 x 48 m	158	link
GV RESPLENDENT	235018597	8825341	MJUJ2	Other Type	Unassigned	Fishing Vessel	Guard Vessel	25.41 x 6.43 m	155	link
GV SHEMARAH II	232007650	9120889	MVRR7	Anti-pollution equipment	Port service craft	Trawler	Guard Vessel	27 x 8 m	301	link
GV FISHER BOYS	234750000	0	MXHY7	Fishing	Fishing vessels	Fishing Vessel	Guard Vessel	20 x 6 m	-	link
GV PLEIADES BF155	235069008	0	2BSR7	Diving ops	Offshore service vessels	Fishing Vessel	Guard Vessel	18 x 6 m	-	link
HAVEN SEAJACK 3	235089873	0	2UQQ	Dredging	Offshore service vessels	Dredger	-	34 x 11 m	-	link
HAVEN SEARISER	235075576	0	2HFO4	Dredging	Offshore service vessels	Dredger	Jack Up Barge - Dredger	18 x 18 m	-	link
HAVILA VENUS	259305000		LAZC	Other Type	Unassigned	Offshore Supply Ship	Offshore supply ship	92 x 22 m	6455	link
ISLE OF JURA	232021966	0	PBZD	Dredging	Offshore service vessels	Utility vessel	Tug - Multicat Anchoring and Dive Vessel	33 x 15 m	499	link
IVERO	232013165	8108377	MCJO3	Other Type	Unassigned	Research/Survey Vessel	Research/Survey Vessel; Multi-purpose Vessel	34.4 x 9.78 m	307	link

NAME	MMSI	IMO	CALL SIGN	VESSEL TYPE NAME	VESSEL CATEGORY BASED ON METCALFE ET AL. (2018)	VESSEL TYPE DETAILED	FUNCTION DURING WINDFARM CONSTRUCTION	LENGTH OVERALL X BREADTH EXTREME	GROSS TONNAGE	WEB-LINK
KAMARINA	247278400	9499644	IITW2	Tug	Port service craft	Tug	Barge Tow Vessel	50 x 15 m	1397	link
KMS ABILITY N294	235014666	0	MDFW3	Fishing	Fishing vessels	Trawler	Guard Vessel	23.06 x 6.4 m	111	link
KMS RESOLUTE	235069036	9698939	3FSA3	Cargo	Bulk carrier and cargo vessels	Fishing Vessel	Guard Vessel	24 x 8 m	-	link
MAERSK CONNECTOR	219275000	9743813	OWEB2	Other Type	Unassigned	Cable Layer	Survey / ROV Vessel	139 X 28 m	10678	link
MAERSK TRADER	220584000	9388596	OXRO2	Tug	Port service craft	Offshore Supply Ship	Anchor Handler / ROV	73.2 x 20 m	4678	link
MANU PEKKA	212701000	0	5BJH2	Dredging	Offshore service vessels	Dredger	Dredger	48 x 15 m	-	link
MERLIN DIVER	576451000	8968856	YJQG4	Other Type	Unassigned	Utility vessel	Guard Vessel	24.99 x 6.18 m	110	link
NATALIE C	232004523	0	GHNE	Fishing	Fishing vessels	Other	Guard Vessel	23 x 6 m	-	link
NEXANS SKAGERRAK	257253000	7619458	LCEK	Other Type	Unassigned	Cable Layer	Cable Laying Vessel - cable laying and trenching	99.75 x 35.41 m	10147	link
NKT VICTORIA	257016000	9791016	LAWV7	Other Type	Unassigned	Cable Layer	Cable Laying Vessel	140 x 30 m	16171	link
NORMAND SERVICE	257062690	9484845	LAZT7	Cargo	Bulk carrier and cargo vessels	Offshore Supply Ship	Offshore Supply Ship - DP 2 Platform Supply Vessel	88.41 x 18.98 m	4007	link
NORSEMAID	235007967	9099107	VQEB2	Other Type	Unassigned	Research/Survey Vessel	Survey and Crew Transfer	23.3 x 5.1 m	200	link
OCEAN ENTERPRISE	232018791	0	MFFY6	Tug	Tug	Port Tender	Workboat	14 x 7 m	-	link

NAME	MMSI	IMO	CALL SIGN	VESSEL TYPE NAME	VESSEL CATEGORY BASED ON METCALFE ET AL. (2018)	VESSEL TYPE DETAILED	FUNCTION DURING WINDFARM CONSTRUCTION	LENGTH OVERALL X BREADTH EXTREME	GROSS TONNAGE	WEB-LINK
OLEG STRASHNOV	212905000	9452701	5BNL2	Other Type	Unassigned	Crane Ship	Heavy Lift Vessel	183 x 47 m	47426	link
OPPORTUNE	233608000	7816056	2WKN	Local vessel	Unassigned	Trawler	Guard Vessel	25.9 x 7.2 m	201	link
PD 174 CONSORTIUM	235000620	0	MNYQ5	Fishing	Fishing vessels	Other	Guard Vessel	21 x 7 m	-	link
PEGASUS	305389000	9433743	V2ED2	Tug	Port service craft	Anchor Handling Vessel	Barge Tow Vessel	48.81 x 14.06 m	1262	link
POLE STAR	235000500	9211987	ZQQC5	Other Type	Unassigned	Buoy-Laying Vessel	Buoy Laying Vessel	51.52 x 12.1 m	1174	link
PRESIDENT HUBERT	205067000	8117471	ORLD	Tug	Port service craft	Tug	Anchor Handling Tug	60 x 15 m	1738	link
RESILIENT GUARD VESS	232711000	7305382	GSEE	Tug	Port service craft	Fishing Vessel	Guard Vessel	26.21 x 6.58 m	154	link
RIX LYNX	235115745	9802358	2JGQ6	Cargo	Bulk carrier and cargo vessels	Offshore Supply Ship	CTV	27.4 x 8 m	106	link
ROCKPIPER	209449000	9583861	5BML3	Other Type	Unassigned	Pipe Burying Vessel	Pipe Burying Vessel - Rock Layer	158.6 x 36 m	30601	link
RONA	235084204	0	MACN3	Passenger	Passenger vessels	Dive vessel	Tail end Vessel; Crew Transfers; Medical Emergency	10 x 4 m	-	link
SARDONYX 2	234311000	8957182	MQUY6	Fishing	Fishing vessels	Trawler	Guard Vessel	18.38 x 6.5 m	119	link
SEACAT INTREPID	235107284	0	2HWU6	High speed craft	Passenger vessels	High speed craft	CTV	27 x 10 m	-	link
SEAHORSE	244137000	8213744	PCAP	Dredging	Offshore service vessels	Pipe Burying Vessel	Pipe Burying Vessel	162.01 x 38.05 m	19516	link

NAME	MMSI	IMO	CALL SIGN	VESSEL TYPE NAME	VESSEL CATEGORY BASED ON METCALFE ET AL. (2018)	VESSEL TYPE DETAILED	FUNCTION DURING WINDFARM CONSTRUCTION	LENGTH OVERALL X BREADTH EXTREME	GROSS TONNAGE	WEB-LINK
SEAJACKS SCYLLA	356068000	9429974	MSBB3	Tug	Tug	Offshore Construction Vessel	Offshore Construction Vessel	139 x 50 m	23641	link
SFF GV ARTEMIS	233975000	0	MVIX5	Tug	Port service craft	Trawler	Guard Vessel	27.69 x 8.7 m	399	link
SHAULORA BF794	234014000	0	2KDQ	Fishing	Fishing vessels	Fishing Vessel	Guard Vessel	17 x 5 m	-	link
SHUNA	235007528	0	VQBP3	Tug	Port service craft	Tug	Towing Vessel	17 x 6 m	-	link
SIEM AIMERY	258182000	9694737	LATX7	Other Type	Unassigned	Cable Layer	Cable Laying Vessel	95 x 21 m	8530	link
SIEM N-SEA	311031800	9424508	C6YG5	Diving ops	Offshore service vessels	Offshore Supply Ship	Offshore Supply Ship - Boulder Removal Vessel	93.6 x 19.74 m	4869	link
SIEM RUBY	257733000	9413444	LEQY	Dredging	Offshore service vessels	Offshore Supply Ship	Offshore Supply Ship - trenching	91 x 22.04 m	7558	link
SIEM STINGRAY	311000620	9676292	C6DA3	Other Type	Unassigned	Offshore Supply Ship	Cable Laying Vessel - Offshore Supply Ship	120.8 x 27 m	8878	link
SKUA	232006107	0	MANX9	Other Type	Unassigned	Other	-	16 x 4 m	-	link
SMIT SENTOSA	205696000	9662356	ORRX	Tug	Port service craft	Tug	Anchor Handling Tug	52 x 15 m	1463	link
SMIT SERAYA	566804000	9662368	9V9845	Tug	Port service craft	Anchor Handling Vessel	Barge Tow Vessel	51.8 x 15 m	1463	link
STANISLAV YUDIN	210334000	8219463	5BYM2	Other Type	Unassigned	Crane Ship	Heavy Lift Vessel	183.3 x 40 m	25527	link
STRIL EXPLORER	259006000	0	MASY7	Other Type	Unassigned	Offshore Supply Ship	DP2 Construction support and ROV vessel	76.4 x 16.2 m	3631	link

NAME	MMSI	IMO	CALL SIGN	VESSEL TYPE NAME	VESSEL CATEGORY BASED ON METCALFE ET AL. (2018)	VESSEL TYPE DETAILED	FUNCTION DURING WINDFARM CONSTRUCTION	LENGTH OVERALL X BREADTH EXTREME	GROSS TONNAGE	WEB-LINK
SWIFT SHORE	232005290	0	2JAH	Fishing	Fishing vessels	Other	Guard Vessel	21 x 7 m	-	link
TEAL OF WICK	235000773	0	ZQXM9	Diving ops	Offshore service vessels	Dive vessel	Tail end Vessel; Multipurpose vessel	20 x 8 m	-	link
UNION BOXER	205575000	9537537	ORPS	Tug	Port service craft	Anchor Handling Vessel	Anchor Handling Tug	41 x 12 m	810	link
UNION FIGHTER	205566000	9537525	ORPP	Tug	Port service craft	Anchor Handling Vessel	Barge Tow Vessel	41 x 13 m	810	link
UNION PRINCESS	205642000	9242766	ORQU	Tug	Port service craft	Anchor Handling Vessel	Barge Tow Vessel	67.4 x 16 m	2258	link
UNION SOVEREIGN	205644000	9262742	ORQW	Tug	Port service craft	Offshore Supply Ship	Barge Tow Vessel	67.4 x 15.5 m	2263	link
VOE JARL	235055168	0	MECU3	Tug	Tug	Tug	HDD Utility Vessel	26 x 11.5 m	161	link
VOS PRELUDE	245969000	9444340	PBZK	Cargo	Bulk carrier and cargo vessels	Offshore Supply Ship	Supply Vessel	73.6 x 16 m	2177	link
WALVIS	576448000	7208649	YJQG3	Other Type	Unassigned	Utility vessel	Guard Vessel	23.85 x 6 m	108	link
WATERFALL	235071392	0	2CBT4	High speed craft	Passenger vessels	High speed craft	Workboat	15 x 6 m	-	link
WB 1	232011832	0		Dredging	Offshore service vessels	Dredger	Dredger - stuck removal	24 x 24 m	-	link
WHALSA LASS	235089425	9633812	2EZQ4	Tug	Port service craft	Tug	Anchor Handling Vessel - Utility Vessel - HDD duct removal	26 x 11.5 m	167	link
WOODSTOCK	235069559	0	2BUY7	Tug	Port service craft	Crane Ship	Support Vessel	34.14 x 10 m	256	link

CHAPTER 3

COMPARISON OF UNDERWATER SOUNDSCAPES AND PORPOISE OCCURRENCE DURING PREPARATIONS FOR PILE-DRIVING AT TWO OFFSHORE WINDFARM SITES

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ABSTRACT

Mitigating cumulative effects of offshore windfarm development on protected marine mammals requires understanding and quantification of sub-lethal effects of multiple stressors related to construction activities. Extensive research has focused on understanding and mitigating the risk of intensive piling noise. However, uncertainty remains over the extent to which preparation work prior to piling activities may displace animals, and thus reduce the risks of injury once pile-driving begins. I investigated the effects of pre-piling activities on local soundscapes and harbour porpoise occurrence at two deep-water offshore windfarms in the Moray Firth, NE Scotland. Arrays of echolocation click detectors (CPODs) deployed at the windfarm sites throughout the 2017 and 2019-20 piling campaigns were used to assess porpoise occurrence within a 5 km buffer around a subset of turbine locations within a 48-h period prior to the initiation of piling. In parallel, I characterised local vessel activity using AIS data and underwater broadband noise levels using calibrated noise recorders (SoundTraps). I then used daily engineering records to quantify how broadband noise levels and porpoise occurrence varied in relation to vessel intensity and engineering activities during the 48 hours prior to piling. Based on engineering logs, the piling vessel arrived onsite on average 18 h before the start of pile-driving activities at both windfarms. Throughout the two piling campaigns, harbour porpoise occurrence gradually declined by up to 20% during the 12 h prior to piling. This decrease in detections was associated with increased levels of vessel activity around each site and specific underwater activities which increased local broadband noise levels. These results provide strong evidence of porpoise displacement several hours prior to piling activities and highlight the value of assessing cumulative

impacts of preparation when optimising mitigation measures to reduce the impact of construction activities at offshore windfarm sites.

INTRODUCTION

The offshore windfarm industry is expanding rapidly to support many nations' net zero ambitions, and adapting equipment, construction vessels, installation techniques and mitigation measures to minimize the environmental impact of these developments (Le Lièvre 2019). Concern over potential impacts upon marine mammals has typically focussed on assessing and mitigating the effects of intense impulsive underwater noise during pile-driving (Tougaard et al. 2003; Carstensen et al. 2006; Brandt et al. 2011; Teilmann and Carstensen 2012; Dähne et al. 2013; Russell et al. 2016; Dähne et al. 2017; Brandt et al. 2018; Graham et al. 2019; Thompson et al. 2020; Whyte et al. 2020). Near field, these impulsive noise sources have potential to injure marine mammals (Southall et al. 2008; Southall et al. 2019). Whilst far-field disturbance (e.g. Graham et al. (2019)) may reduce foraging opportunities (Wisniewska et al. 2018; Benhemma-Le Gall et al. 2021) and have population level impacts that must be assessed by regulators to meet environmental legislation (Booth et al. 2017; Nabe-Nielsen et al. 2018).

Where marine mammals may occur within construction sites, mitigation measures must therefore be integrated into the engineering procedures to reduce the risk of near-field injury from pile-driving noise. Typically, these measures aim either to: 1) ensure animals are absent from a potential injury zone before piling is initiated by conducting visual or acoustic observations prior to piling (JNCC 2010); 2) deter animals from the potential injury zone by using Acoustic Deterrent Devices (ADDs) and soft start protocols (Thompson et al. 2020); or 3) attenuate pile-driving noise using noise abatement techniques (e.g. bubble curtains) (Dähne et al. 2017; Brandt et al. 2018). However, there are difficult trade-offs to balance when deciding how best to implement these different measures. Both visual and passive acoustic monitoring may fail to detect animals that enter the injury zone (JNCC 2010). While the efficacy of ADDs for many species remains uncertain and, where effective, may have far-field disturbance effects (Brandt et al. 2013b; Gordon et al. 2019; Thompson et al. 2020; Findlay et al. 2021). The efficacy of noise abatement techniques also remains uncertain, particularly in deeper waters (Verfuss et al. 2019b; Wagenknecht 2021). These approaches also often require additional vessels onsite (Brandt et al. 2018) which could impact marine mammals through altered underwater soundscapes or have broader environmental impacts.

Further, decisions over best mitigation should be made using a risk-based approach. In particular, the level of mitigation required should be based on the likelihood of individuals of different species being present within the injury zone at the start of piling activity. Currently, such decisions are made using baseline data that are collected at least one year before construction. However, pre-piling activities have the potential to disturb marine mammals through increased levels of background noise and vessel activity. For example, harbour porpoise detections declined in the three hours prior to piling activities at eleven offshore windfarms in the German Bight (Rose et al. 2019). This was assumed to be related to construction traffic, but empirical data on vessels were not available to test this assumption (Brandt et al. 2018; Rose et al. 2019). These findings suggest that construction-related vessel activity may deter animals from the disturbed area before piling. Better understanding of variation in marine mammal occurrence prior to the start of impulsive noise activities would reduce the uncertainties in the number of individuals likely to be in the injury zone at the start of impulsive noise activities. In return, mitigation measures may then be optimised to ensure near-field animal displacement while minimizing unnecessary far-field disturbance (Thompson et al. 2020).

Here, I address this data gap by investigating the levels of vessel activity and underwater broadband noise during the 48-hour period prior to pile-driving at the UK’s first two large-scale deep-water offshore windfarms. I assessed harbour porpoise responses to specific activities and characterized the soundscape of various activities conducted by the piling and ancillary vessels. Finally, I discussed the wider management implications of the results and provide some recommendations for optimizing construction work while integrating context-dependent, adaptive mitigation measures.

MATERIAL AND METHODS

Characterisation of construction activities

The study was conducted around two offshore windfarms in the outer Moray Firth, NE Scotland (for details see Benhemma-Le Gall et al. (2021)). Piling at the Beatrice windfarm occurred on 103 days, between 2nd April and 2nd December 2017, and required impulsive pile driving techniques to install a set of four piles at each of the 84-wind turbine and two-Offshore Transformer Module locations (Graham et al. 2019).

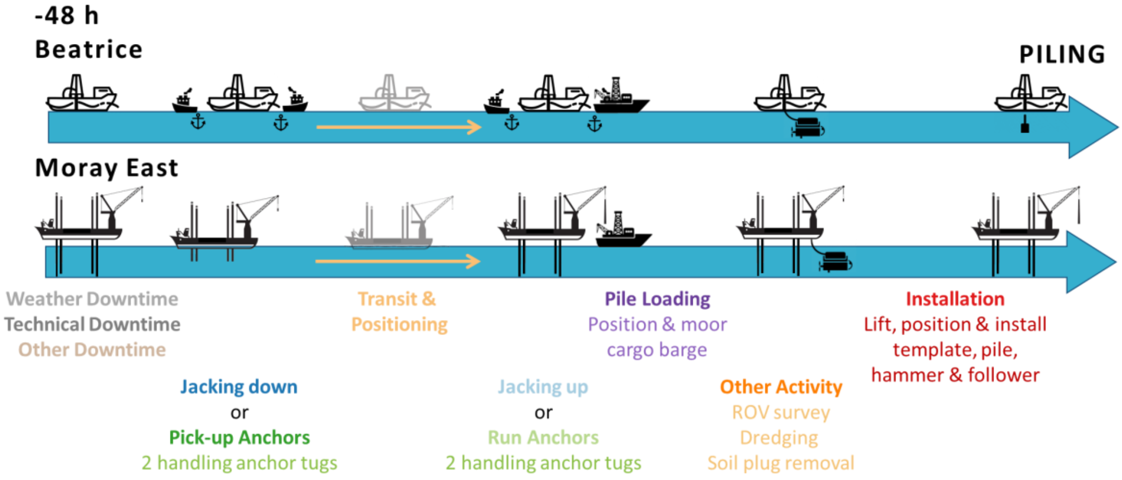

Figure 3.1 Schematic of activities conducted by the piling and ancillary vessels prior to piling at Beatrice and Moray East offshore windfarms (NE Scotland)

Prior to the pile installation works, a heavy lift vessel was positioned by two dedicated anchor handling tugs using an eight-point anchor spread and a pile installation frame (PIF) was lowered onto the seabed. A cargo barge delivered the four piles to the piling vessel, which then placed each pile in the PIF sleeves. Piling at the Moray East offshore windfarm occurred on 132 days, between 19th May 2019 to the 27th February 2020. Similar impulsive pile driving techniques were used to install a set of three piles at each of the 100-wind turbine and three-Offshore Substation Platform locations. Prior to piling operations, the heavy lift vessel undertaking the pile installation works was positioned in readiness and jacked up to operational draft. At every two locations, a supply vessel delivered six pin piles to the piling platform, which loaded them onto the deck. As at Beatrice, the PIF was lowered onto the seabed and the three piles were positioned into the PIF sleeves. At both windfarms, an Acoustic Deterrent Device (ADD) was deployed and a soft start piling procedure implemented prior to the start of piling to deter marine mammals from the predicted near-field injury zone (for more details see Thompson et al. (2020)).

Information on the type and duration of different piling vessel activities were extracted from engineering logs and grouped into 8-9 categories (see Table S 3.1 and Table S 3.2 for details). Briefly, *Transit and Positioning* corresponds to the movement of the heavy lift vessel to each piling location before either mooring at Beatrice or jacking-up at Moray East. Once positioned, the piling vessel at Beatrice was moored with the support of two ancillary vessels. This activity, called *Run Anchors*, was usually followed by *Pile Loading*. At Moray East, this activity occurred every two turbine locations after jacking-up (*Jacking-up*). Once the piling vessel was jacked-up at operational draught, it remained out of the water until the end of pile installation. A diverse range of activities such as ROV seabed survey and dredging were grouped into the *Other Activity* category. *Installation* included lowering the PIF to the seabed and placing piles and the hammer in position. However, the deployment of ADD mitigation and pile-driving that have been the focus of previous studies (Graham et al. 2019; Thompson et al. 2020; Benhemma-Le Gall et al. 2021) were not included in the main analyses. Once pile-driving and PIF retrieval were completed, the piling vessel jacked-down at Moray East (*Jacking-down*), whereas two anchor handling vessels picked-up the eight anchors and brought them to the piling vessel at Beatrice (*Pick-up Anchors*). Remaining categories included *Weather Downtime* (due to poor weather conditions), *Technical Downtime* (during breakdown of equipment) and *Other Downtime* (which encompassed crew changes, waiting for piles etc.). The overall proportion of each activity was calculated for the whole piling campaign for each windfarm. Similarly, the duration of each activity per piling location was summarised for the whole piling campaign and compared between windfarms using a non-parametric Kruskal-Wallis test. Additionally, using a subset of locations, proportions of activity per hour were summarised from 48 h before the start of piling. For these analyses, the *Installation* phase also encompassed the deployment of ADD mitigation and pile-driving activities.

Furthermore, to investigate whether activity influenced harbour porpoise occurrence, each hour prior to piling was classified based on the dominant activity, conditional on that activity occurring for at least 70% (42 min) of that hour. Hours not meeting this condition were excluded from this analysis.

Piling timeline

Timelines for ADD deployments and piling activity were extracted from the mitigation records and engineering logs provided by each developer. As outlined in Thompson et al. (2020), ADDs were deployed for 15 mins prior to the start of piling, followed by a soft start. On some occasions, piling events at individual turbine locations were spread across several days due to weather or technical downtime. For this study, I focussed on piling bouts that occurred at the same turbine location with no breaks in piling of > 12 hours. Additionally, I considered only those piling bouts with at least a two-day gap between the end of piling at the previous turbine location and the start of piling at the focal turbine location. Preliminary analyses of engineering logs showed that the piling vessel arrived onsite on average 18.2 h (range: 10.7-32.7 h) before the start of pile-driving activities at Beatrice, and around 17.6 h (range: 7.5-29.1 h) at Moray East. Based on these findings, the main analyses focussed on the period 48 hours prior to the start of piling at each location. Four locations from both the Beatrice and Moray East datasets were excluded from these analyses because the piling vessel arrived onsite more than 48 hours prior to the start of piling (Figure 3.2). At the 31 sites remaining in the analyses, vessel data were considered in relation to the Hour relative to piling (HRP) ranging from -48 to 0 h and where hour 0 represents the hour in which the ADD was deployed and piling was initiated.

Spatial analysis of vessel-tracking data

Vessel activity in the vicinity of turbine locations was extracted for all months in which piling took place within 2017, 2019 and 2020, based upon from 1 and 5 min-resolution Automatic Identification System (AIS) vessel-tracking datasets (Astra Paging Ltd. and Anatec Ltd.). Following the procedure used by Benhemma-Le Gall et al. (2021), vessel-tracking data were processed to produce an hourly index of vessel intensity within a 5 km buffer around each piling location. Georeferenced AIS data, and piling locations were first projected into WGS84 UTM 30N using the *sf* package in R (Pebesma 2018; R Core Team 2019). AIS data were then interpolated either every 1 or 5 min, and spatially filtered to retain data within a 5 km buffer around each location. The interpolated vessel locations were then temporally filtered to extract vessel locations that were recorded in each HRP. I excluded the piling vessel at both windfarms and focused this analysis on additional vessels that were within the 5 km buffer around piling locations.

Passive Acoustic Monitoring

During both piling campaigns, an array of echolocation click detectors (V.0 and V.1 CPODs; www.chelonia.co.uk) was deployed within the two windfarm sites, following the sampling design used in Graham et al. (2019). A subset of these 2017 and 2019 arrays was used in the present study to investigate variation in harbour porpoise occurrence in relation to pre-piling activities at both windfarm sites. I selected CPOD sites that recorded data from within 5 km of active piling locations (Figure 3.2). Data were processed and extracted with the manufacturer's software CPOD.exe (v2.044). High and moderate quality Narrow Band High Frequency echolocation click trains of porpoise origin were identified and filtered using the standard "KERNO" classifier. For each CPOD, the number of porpoise echolocation clicks was exported, summarised and converted into presence-absence of detections per HRP for each piling location.

To ensure that high levels of background noise did not saturate memory and prevent the CPODs from logging clicks (Wilson et al. 2013), I only kept data from hours which recorded clicks in all 60-min samples within the hour. Additionally, to prevent any masking effect of vessel noise on porpoise echolocation click detections, I only included hours during which the mean vessel distance from CPOD locations was greater than 1 km. This threshold was based upon previous analysis of data from NE Scotland which suggested that vessel noise was unlikely to saturate the CPOD click threshold beyond 1 km (Pirotta et al. (2014a)'s Supplementary material).

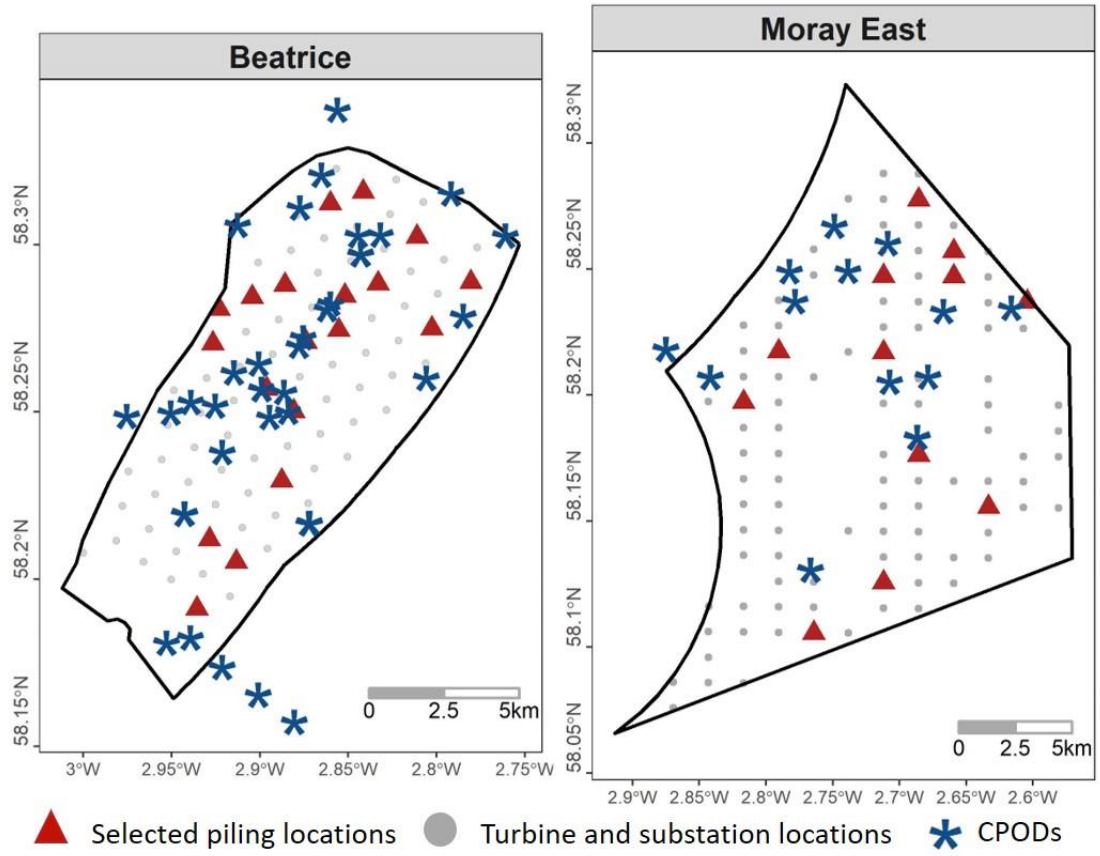

Figure 3.2 Spatial distribution of the echolocation clicks detectors (CPODs; blue stars) within 5 km of the subset of piling locations (red triangles) selected for the study at Beatrice and Moray East offshore windfarms; grey circles represent the turbine and substation layouts at both windfarms.

Variation in broadband noise levels

Calibrated measurements of broadband noise levels were made for 7-19 days in September 2017 at Beatrice and for 12 days in July 2019 at Moray East. In each case, three bottom-mounted noise recorders (Ocean Instruments SoundTrap) were moored at distances of 0.5 to 5 km from piling locations (Figure 3.3). At Beatrice, recorders were duty cycled (1/10 mins) and sampled at 576 kHz. At Moray East, continuous recordings were made with a sampling rate of 48 kHz (see Table S 3.4 for details). During these short-term deployments, eight turbine locations were piled at Beatrice and seven at Moray East (Figure 3.4). Data were processed through MATLAB following Merchant et al. (2015) and broadband Sound Pressure Levels (SPL) were extracted at 1-min resolution.

To investigate the acoustic signatures of the piling vessel activities prior to the start of piling, I linked the activity timeline of the two piling vessels with the broadband noise level timeline. Given the duration of activities varied, I randomly extracted 30 one-minute broadband noise samples per event, activity type, turbine location and noise recorder device. I then summarised the mean and 90th percentile broadband SPL to compare acoustic signatures of different activities along a distance gradient. The mean and 90th percentile were collinear and mean broadband SPL was therefore used in subsequent analyses.

To compare the overall pre-piling broadband noise levels between the two windfarm construction sites, irrespective of the activity type, I selected a subset of 6 piling locations (three for each site) for which there was a gap of at least 24 h between piling events. As previously, I summarised the mean broadband noise levels per hour relative to piling, from 24 h prior to the start of piling and deterrence activities. For each one-minute sample, Euclidean distances between the noise recorder deployment sites and 1) the piling vessels and 2) any other vessels within a 5 km buffer around the noise recorder were calculated. The mean distance to the piling vessel and the minimum distance to any other vessels were summarised per hour relative to piling and then log-transformed. These metrics were included in models as the background noise levels were likely influenced by the presence and distance from other noise sources such as other vessels.

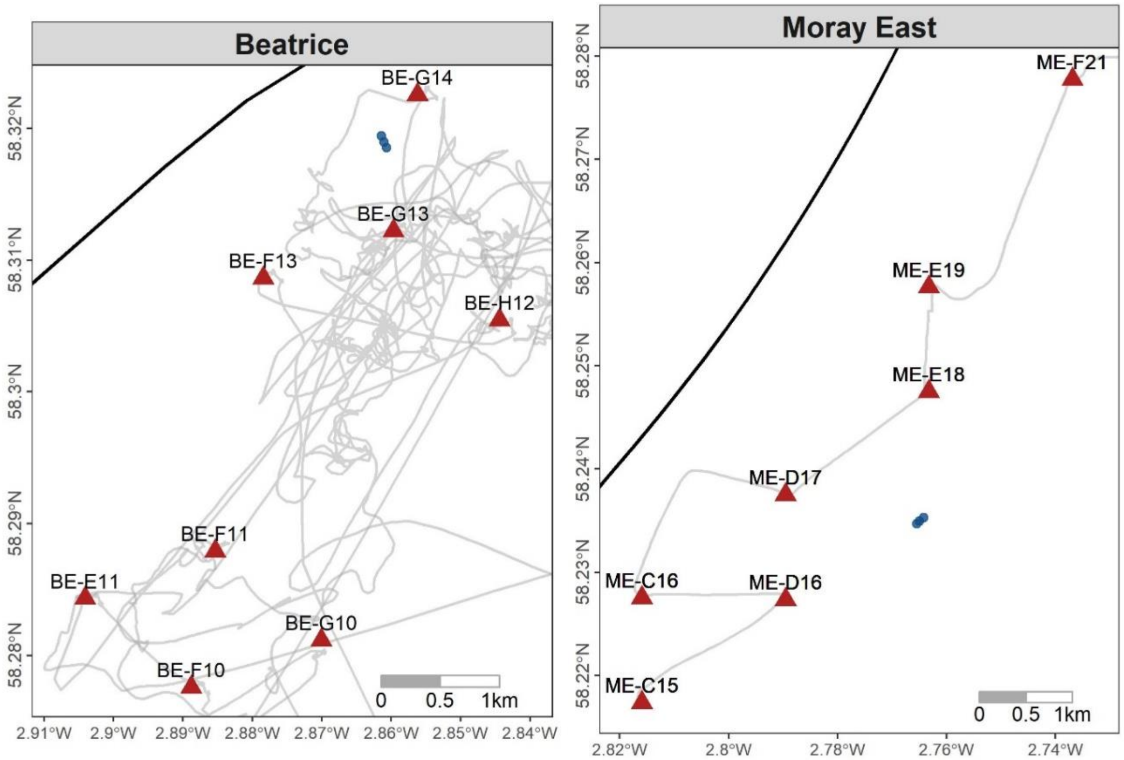

Figure 3.3 Maps of the turbine sites piled at Beatrice offshore windfarm between 5 and 23 September 2017 and at Moray East offshore windfarm between 10 and 23 July 2019 (red triangles), used for the noise analyses. Noise recorder deployment sites are represented as a blue circle; the grey lines represent the piling vessel track line during the indicated time period.

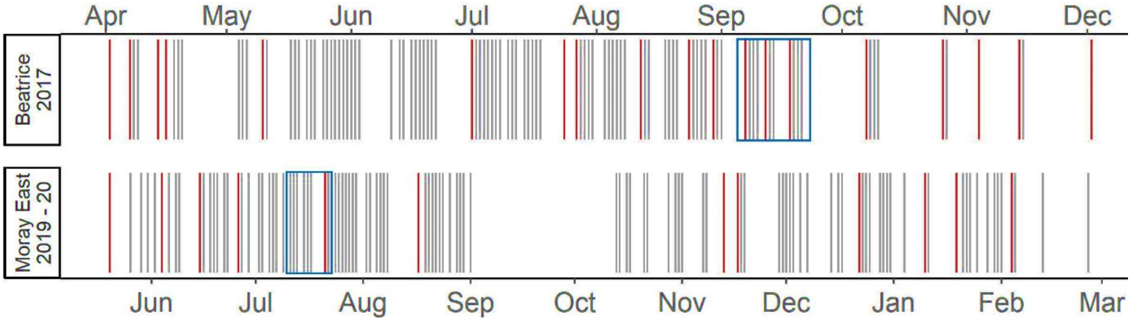

Figure 3.4 Pile-driving timeline at Beatrice and Moray East offshore windfarms, with the lines representing piling bouts and the red lines those bouts selected for the study; the blue rectangle indicates the time period during which underwater noise recordings were collected.

Modelling

Overall differences in harbour porpoise acoustic detections and levels of vessel intensity in the 48-hour period prior to piling were compared between the two windfarms using a Mann-Whitney U test. Generalised Additive Models (GAMs) were used to characterise variation in vessel intensity and porpoise detections during the 48 h before to the first hour of piling activities at each windfarm. A GAM was also used to characterise variations in mean broadband noise levels during the 24 h to the first hour before piling activities at each windfarm. In the *porpoise model*, the binary presence/absence of porpoise detections per HRP was fitted with a binomial distribution and a *probit* link function using the *gam* function of the *mgcv* R package (Wood 2011). Similarly, in the *vessel model*, the vessel intensity, ranging from 0 to 4.3 min.km^{-2}, was used as the response variable and was fitted to a Tweedie distribution. The factor *windfarm ID* and the continuous variable *HRP*, ranging from -48 to 0 and defined by cubic regression splines, were included as explanatory variables. In the *noise model*, the mean broadband SPLs, ranging from 101.1 to 142.3 dB re 1 µPa, was used as the response variable and fitted to a Gaussian distribution, with the *identity* link function. To avoid under- or overfitting the models, the basis dimension *k* was arbitrarily set up large and then decreased based on the model diagnosis tool of the *gam.check* function. A double penalty approach and the Restricted Maximum Likelihood (REML) method were used for automatic term and smoothness selection. To account for potential temporal autocorrelation in porpoise detections between hours relative to piling for each piling bout, a temporal autocorrelation *corA1* was used.

To estimate the levels of background noise generated during different activities at both windfarms, SPLs averaged over 30 minutes were fitted to a Linear Model in separate *noise models* for each windfarm. The activity type, the log-transformed mean distance to the piling vessel and the log-transformed minimum distance to any other vessels within a 5 km-radius of the noise recorders were included as explanatory variables. The significance of the fixed effect was then tested using a sequential analysis of deviance table (Type II Wald chi-square tests) with the R package *car* (Fox and Weisberg 2019). Pairwise comparisons were conducted, using the *emmeans* package (Lenth 2020), to statistically compare the acoustic signature of the *Weather Downtime* phase with the other activities. The response variable was predicted, and 95% confidence intervals (CI) calculated fixing the distance to the noise source to 2 km and to any other vessels to 4 km for each windfarm. Boxplots of the observed measurements were displayed to illustrate model fit. To compare variation in broadband noise levels between windfarm construction sites from 24 h to the first hour of piling activities, a linear model was used with the mean SPLs as response variable. The factor windfarm, and the interaction between the *log (mean distance)* to the piling vessel and the *log (min distance)* to any vessels were the explanatory variables.

To investigate the variation in porpoise occurrence in relation to different construction activities occurring in the 48 h prior to piling at each windfarm, the binary presence/absence of porpoise detections per HRP (with 70% of the same activity) was fitted to a Generalised Linear Mixed Effect Model (GLMM) with a Binomial distribution. Based on the Akaike Information Criterion (AIC), a *cloglog* link function fitted the Beatrice dataset best, while a *probit* link function was used for the Moray East dataset. The construction activity type (i.e. *Weather Downtime, Run Anchors, Jacking-up, Other Activity, Pile Loading* and *Installation*) was used as explanatory covariate and the HRP associated with the turbine ID was used as nested random effect to account for potential temporal autocorrelation. I excluded the first hour of piling (i.e. 0 HRP) to focus only on the pre-piling variation in porpoise occurrence. Wald's tests were conducted to assess significance of the fixed effect and pairwise comparisons were carried out between the *Weather Downtime* and the other activities. The response variable was predicted, and uncertainty (95% CI) calculated by bootstrapping (100 simulations) with the bootMer function of the *lme4* package (Bates et al. 2015). Finally, model validation was undertaken using the *DHARMa* package to verify the uniformity, dispersion, spatial and temporal autocorrelation of residuals (Hartig 2020).

RESULTS

Piling vessels arrived onsite and started anchoring or jacking up at the sub-set of turbine locations, used for this study, 18 hours prior to piling at both windfarms (Table 1). At Beatrice, 31 PAM sites could be used to estimate variation in porpoise detections within 5 km of the 19 turbine locations selected for these analyses. Sample sizes at Moray East were slightly smaller, with 13 PAM sites within 5 km of 12 turbine locations (Table S 3.5). Overall, harbour porpoise occurrence was higher at Moray East, with a mean porpoise detection probability per hour of 0.4 against 0.28 at Beatrice ($W = 7{,}720{,}876$, $p < 0.001$). Additionally, levels of vessel intensity were significantly higher at

Beatrice with an averaged vessel intensity of 1.0 min.km^{-2} against 0.7 min.km^{-2} at Moray East (W = 4,987,337, $p < 0.001$).

Table 3.1 Sample size, sampling effort, overall vessel intensity, porpoise detection probability at selected piling locations at the Beatrice and Moray East offshore windfarms; HRP corresponds to the hours relative to piling and PAM stands for Passive Acoustic Monitoring; for further information on the sample size per HRP see Table S 3.5.

Windfarm	Turbine Locations (N)	PAM Sites (N)	Vessel intensity (min.km^{-2}) Mean [min; max]	Piling vessel time of arrival (hour) Mean	Porpoise detection probability per hour
Beatrice	19	31	1.03 [0; 4.30]	-18.2	0.28
Moray East	12	13	0.74 [0; 2.74]	-17.6	0.4

Pre-piling timeline description of activities, vessel pressure and porpoise occurrence at two offshore windfarms

Throughout the entire piling campaign, the proportion of time spent on each activity type varied between the two windfarms. For instance, the proportion of *Weather Downtime* was 9.1% higher at Beatrice than at Moray East (Figure 3.5A). Overall, the piling vessel was on *Weather Downtime* 47.2% of the piling campaign at Beatrice and 38.1% at Moray East. At Beatrice, the piling vessel anchored at sea outside the construction site or returned to port for most (>75%) of the weather downtime. At Moray East, however, the piling vessel always stayed on site during weather downtime (Figure 3.5). For both windfarms, the *Installation* phase (incl. piling activities) was the second main activity, followed by *Technical Downtime*. No significant differences in the duration spent on *Weather* (H (1) = 1, p = 0.32) and *Technical Downtime* (H (1) = 3.244, p = 0.07) at piling sites were observed between windfarms. However, the time spent conducting other activities differed significantly between windfarms. For instance, *Pile Loading* took nearly three hours longer at Moray than at Beatrice (H (1) = 63.77, p < 0.0001), whereas the *Installation* phase was on average 3.7 h faster at Moray East than at Beatrice (H (1) = 6.87, p < 0.01). Jacking-up was, on average, 1.7 h longer than Running Anchors (H (1) = 59.12, p < 0.0001), but Jacking-down was 2.5 h faster than Picking-up Anchors (H (1) = 110.91, p < 0.0001). Overall, this securing phase was around 55 min faster at Moray East than at Beatrice (H (1) = 15.11, p = 0.0001). The phases *Transit and Positioning* (H (1) = 82.78, p < 0.0001), and *Other Activities* (H (1) = 30.38, p < 0.0001), were on average longer at Moray East than at Beatrice by 1.5 h or 2 h respectively. (Figure 3.5C).

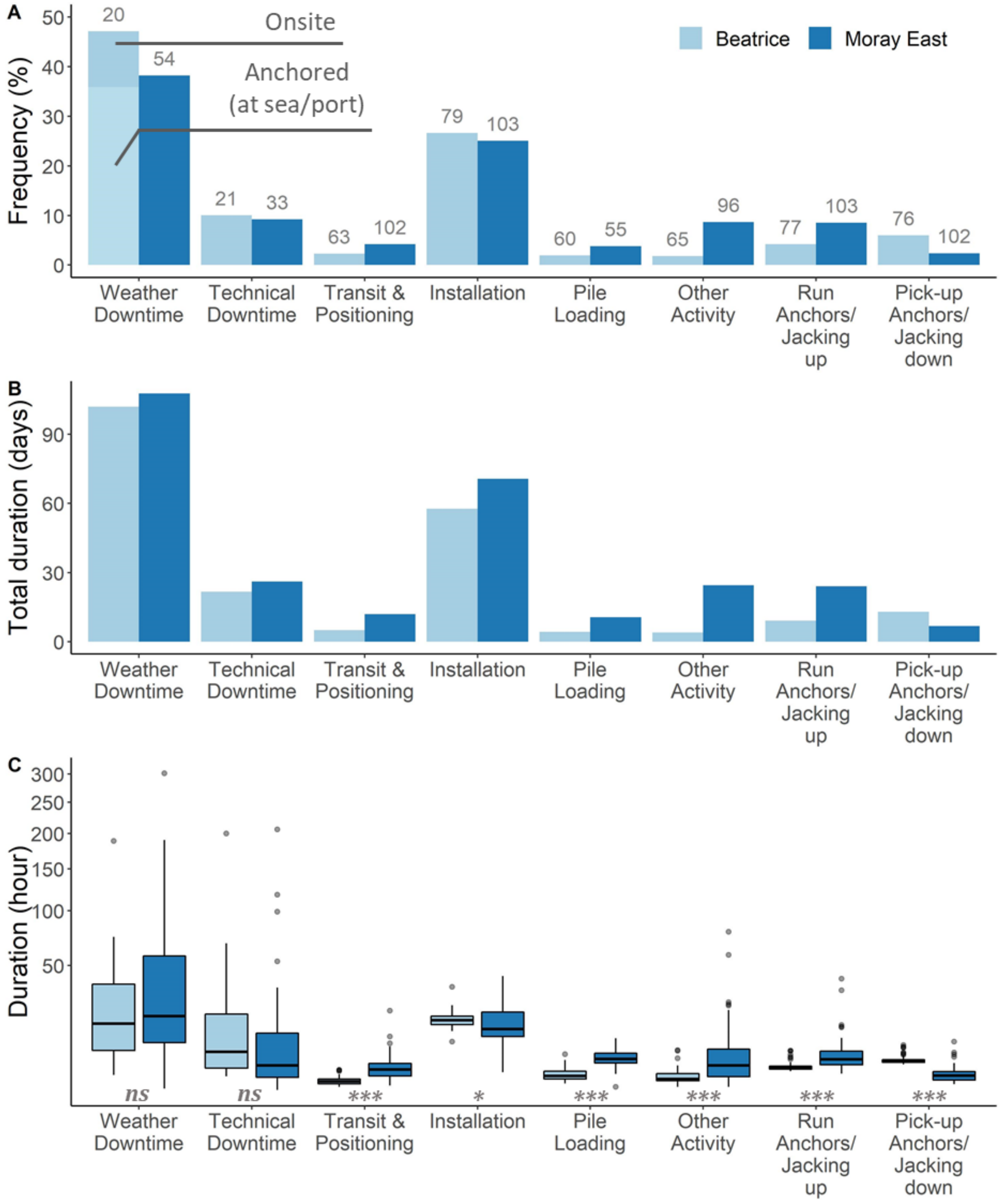

Figure 3.5 A) The percentage of time spent in different construction activities during piling campaigns at Beatrice (22/04 - 02/12/2017) and Moray East (19/05/2019 - 27/02/2020) offshore windfarms; B) Total duration of construction activities (in days) during the piling campaigns; C) Activity durations (in hours); grey dots are the outliers; numbers are piling site sample size per windfarm; The piling vessel at Beatrice spent ~36% of its time anchored at sea or at the port in weather downtime (light-blue), and ~11% of its time at the construction site in weather downtime. Run and Pick-up anchors were activities conducted at Beatrice only, while Jacking-up and down were only

*conducted at Moray East; ns Kruskal-Wallis test non-significant; *** p-value < 0.0001, * p-value < 0.01.*

For the focal subset of turbine locations, pre-piling activities were spread throughout the 48-hour pre-piling phase. As for the entire campaigns, the main activity recorded at both windfarms was *Weather Downtime*. Levels of downtime decreased to < 50% about 12 h before piling. In parallel, levels of *Pile Loading* and *Installation* activities increased at this time, concurrent with a 20% decrease in porpoise occurrence at both construction sites (Figure 3.6A and Figure 3.6C). At Beatrice, an increase in vessel intensity was recorded from -17 HRP onwards, coincident with higher proportions of *Running Anchors* and *Pile Loading* activities (see Figure 3.6A and Figure 3.6B; Figure S 3.2). At Moray East, levels of vessel intensity were generally lower than at Beatrice but did increase slightly around 24 h before piling (Figure 3.6A and Figure 3.6B).

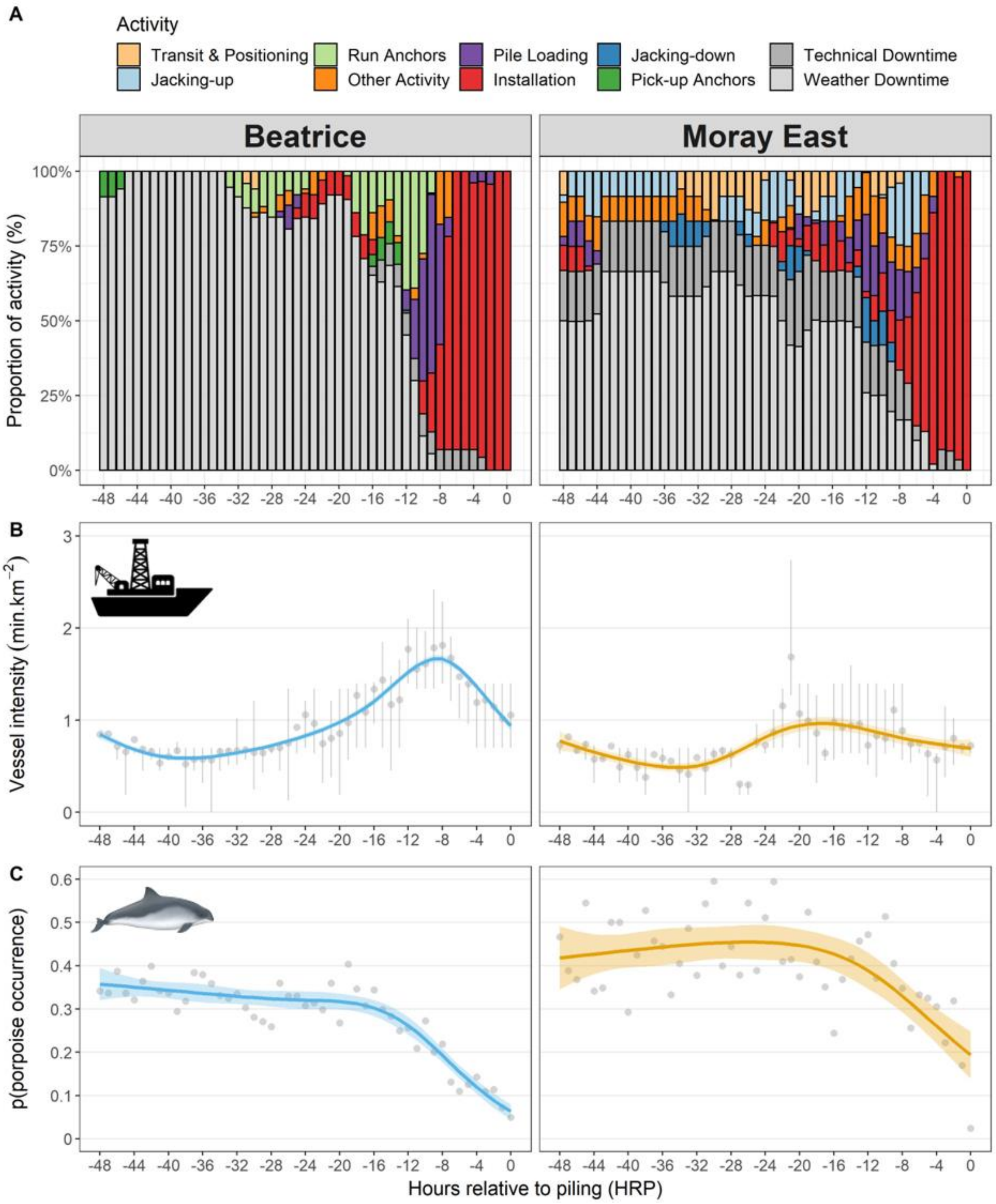

Figure 3.6 Variation in A) Percentage of construction activities B) Vessel intensity (min.km^{-2}) and C) probability of porpoise occurrence throughout the 48 hours prior to the start of pile-driving activities at a subset of piling locations at Beatrice and Moray East offshore windfarms. Grey dots in figures B) and C) represent the mean of observed data, the line range in figure B) represents the 1st and 3rd quartile; the blue and yellow lines are the GAM fitted lines and the shaded areas represent the 95% confidence intervals to estimate the uncertainty. For further information on the sample sizes in each hour relative to piling see Table S 3.5.

Soundscape in the vicinity of a subset of piling locations

At both windfarms, levels of anthropogenic background noise varied with activity type (Beatrice $F_{6,116} = 8.11$, $p < 0.001$; Moray East $F_{7,297} = 44.35$, $p < 0.001$). Additionally, broadband noise levels significantly varied with the *log (mean distance)* between the noise recorders and the construction site at Moray East ($F_{1,297} = 126.18$, $p < 0.001$), and with the *log (min distance)* to any vessels at Beatrice ($F_{1,116} = 8.59$, $p < 0.01$).

At Beatrice, noise levels during *Weather Downtime* (115.82 ± 2.1 dB re 1 µPa) were not significantly different from those measured during *Technical Downtime*, *Pile Loading, Installation* and *Other Activity*, with estimated mean SPLs ranging between 109.85 and 120.57 dB re 1 µPa (Figure 3.7A). However, in comparison with *Weather Downtime,* higher levels of background noise were generated both when running the eight anchors (126.65 ± 3.85 dB re 1 µPa; $p < 0.01$) and picking them up (127.68 ± 3.65 dB re 1 µPa; $p < 0.01$) (Figure 3.7A).

At Moray East, noise levels generated during *Weather Downtime* (114.57 ± 3.89 dB re 1 µPa) were not significantly different from those measured during any other activity, with estimated mean SPLs ranging between 113.18 and 119.93 dB re 1 µPa (Figure 3.7B). Nevertheless, whilst not statistically significant, the noise levels produced during the *Pile Loading*, *Jacking-up* and *Technical Downtime* phases were typically higher than those recorded during *Weather Downtime* (Figure 3.7B).

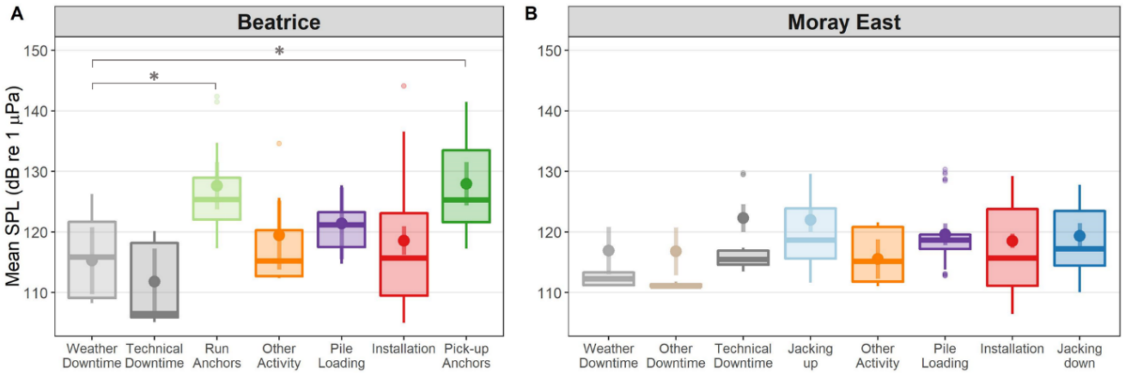

Figure 3.7 Observed (boxplot and small dots) and estimated (big dots) Sound Pressure Levels (dB re 1 µPa) during different construction activities at three turbine locations with both the Beatrice (A) and Moray East (B) offshore windfarms. Estimated levels were based on an average distance between the piling vessel and the noise recorders of 4 km and a minimum distance between any vessels and the noise recorders of 2 km. 95%

*confidence intervals (line range) are estimated for the uncertainty in fixed effects only; * for p-value < 0.05.*

For the two weeks of noise measurements, broadband noise levels varied significantly between the 24 h to the first hour before piling activities at Beatrice ($F_{4.77,5} = 52.88$, $p < 0.001$) but not at Moray East ($F_{0.48,5} = 0.19$, $p = 0.17$). A peak in mean SPLs, ranging between 120.7 and 128.3 dB re 1 µPa, was detected between -14 and -8 HRP at Beatrice (Figure 3.8A).

Furthermore, variation in the overall broadband noise levels generated from 24 h to the first hour before piling was explained by the factor windfarm ($F_{1,415} = 19.63$, $p < 0.001$) and the interaction between the *log(mean distance)* to the piling vessel and the *log(min distance)* to any vessels within a 5 km-radius around noise recorder locations ($F_{1,415} = 12.25$, $p < 0.001$). Noise levels were estimated to be 3.8 dB louder at Moray East than at Beatrice, when the piling vessel was 4 km and other vessels were at 2 km from the noise recorders (Figure 3.8B). Noise levels were negatively related to the distance from the piling vessel and from any other vessels (Figure S 3.3).

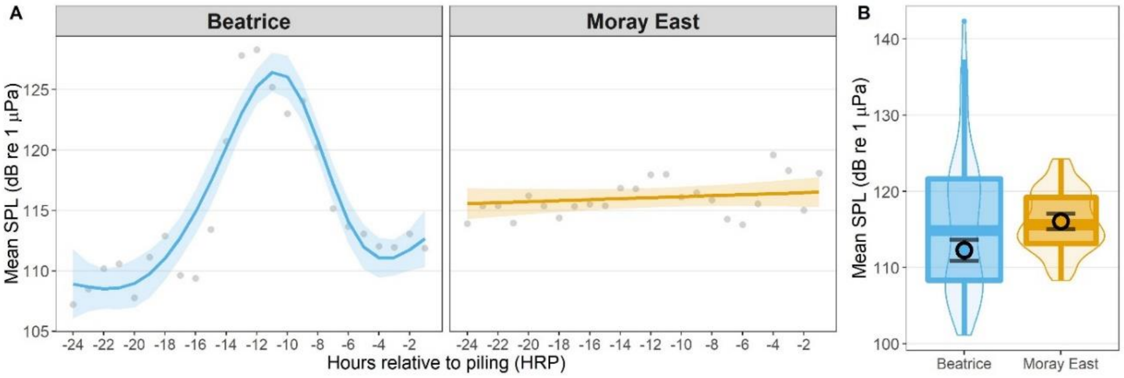

Figure 3.8 A) Mean broadband sound pressure levels (SPLs) per hours relative to piling (i.e. from 24 h to the first hour before the start to of pile-driving activities) at three piling locations at both the Beatrice and Moray East offshore windfarms (n = 6); the grey dots represent the mean values of observed data; the blue and yellow lines are the GAM fitted lines and the shaded areas represent the 95% confidence intervals to estimate the uncertainty; B) Observed (boxplot and violin plot) and estimated (black circle and error bar) mean SPLs per windfarm, when the mean distance from the piling vessel is fixed to 4 km and the minimum distance to any vessels is fixed to 2 km.

Variation in porpoise occurrence during different construction activities

The type of construction activity influenced porpoise occurrence at both windfarms (Wald test - Beatrice χ^2 (5) = 64.436, p < 0.001; Moray East χ^2 (5) = 19.894, p = 0.001). At Beatrice, the probability of detecting porpoises during *Weather Downtime* was around 0.35 (95%CI: 0.22-0.54). Although porpoise detections were lower during all other activities, this difference was only significant for *Installation* (0.13, 95%CI: 0.07-0.23; p = 0.001) and *Other Activity* (0.09, 95%CI: 0.02-0.19; p < 0.01) (Figure 3.9A). At Moray East, the probability of detecting porpoises during *Weather Downtime* was around 0.4 (95%CI: 0.32-0.47). No significant differences in porpoise detections were observed between this downtime phase and other activities (Figure 3.9B).

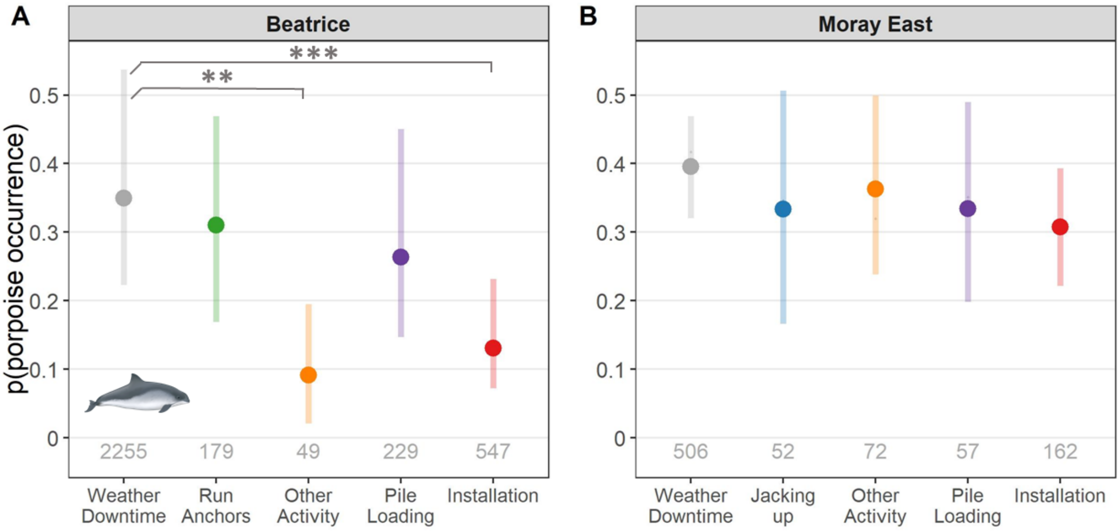

*Figure 3.9 Probability of harbour porpoise occurrence per hour during different construction activities at A) the Beatrice and B) Moray East offshore windfarms; the 95% confidence intervals (line range) estimated for the uncertainty in fixed effects only; sample size for each activity (numbers in grey) are the numbers of hours relative to piling during which at least 70% of the same activity; *** for p-value < 0.001 and ** for p-value < 0.01.*

DISCUSSION

Efforts to assess and mitigate the environmental impacts of proposed offshore windfarms require good understanding of different construction procedures and responses of key receptors such as marine mammals. As a new industry with rapidly evolving infrastructure, this has led to significant uncertainties during the early phases of the development of offshore renewables. Previous strategic monitoring in this (Graham et al. 2019; Thompson et al. 2020) and other regions (Tougaard et al. 2003; Teilmann and Carstensen 2012; Dähne et al. 2013; Brandt et al. 2018) has focused primarily on marine mammal responses to pile-driving and deterrence activities. Here, I specifically focused on assessing cumulative effects of pre-piling activities to optimise mitigation of piling impacts. These data highlight how differences in construction vessels and operational procedures may, to some extent, influence variation in the local soundscape and in harbour porpoise occurrence at those construction sites.

Construction activities throughout the piling campaign

Whilst both windfarms in the study used pin-piled jacket structures for their foundations, the piling vessels and engineering processes used were markedly different; with Beatrice using an anchored piling vessel and Moray East a jack-up vessel. These different procedures led to variations in the time spent undertaking key construction activities (e.g. *Installation, Pile Loading, Anchoring/ Jacking-up* and *down*) between the two piling campaigns (Figure 3.5). During *Pile Loading*, six piles were delivered to the piling vessel at Moray East against four at Beatrice, which would explain the longer time undertaking this activity at Moray East. On the other hand, the jacket structures required only three pin-piles at Moray East against four at Beatrice, which may explain why *Installation* activities took less time at Moray East. The securing phase encompassing jacking-up and down at Moray East was slightly shorter than running and picking up anchors at Beatrice. In contrast, the time spent on downtime by the piling vessels was similar for both piling campaigns. The piling vessel was on weather downtime for much of time throughout both piling campaigns; 47% of the time at Beatrice and 38% at Moray East, highlighting the logistical and financial challenges of construction work offshore. In comparison, observed differences in time spent in other activities at the two windfarms are small. Nevertheless, the choice of vessel type or procedures may, in some cases, reduce the overall cost and time of the piling campaign.

Variation in vessel activity at the two windfarms

Overall, higher levels of vessel intensity were observed at Beatrice compared to Moray East. However, a peak in vessel intensity was observed in the period before piling was initiated at both sites (Figure 3.6B). Higher levels of vessel intensity at Beatrice were primarily due to anchoring activities that required the support of two anchor handling tugs. For both developments, *Pile Loading* and *Other Activity* were also associated with higher levels of vessel intensity (Figure S 3.2). However, the frequency of pile loading activities varied between the two campaigns because cargo barges supplied the piling vessel at each location at Beatrice but only at every second location at Moray East. Furthermore, at Beatrice crew changes required the support of ancillary vessels, whereas Moray East crew changes were made by helicopter. There were also spatial differences in vessel intensity at the two sites. For instance, the Beatrice piling vessel was on weather downtime within the windfarm site for 11 % of the time, but not anchored. In contrast, the Moray East vessel was jacked up during similar downtime, which reduced the spatial footprint of vessel activity (see Figure 3.3). Thus, the choice of piling vessel type and scheduling of trips to supply equipment, or transfer crew, provide opportunities to reduce vessel intensity at construction sites. However, this must be balanced against overall costs and carbon emissions associated with these options. In this study, all vessels involved in construction used AIS. However, it should also be recognised that decisions over management of other marine activities within construction sites may affect overall levels of vessel activity and underwater soundscapes. For example, some fisheries activity continued to varying degrees within both these construction sites, but this could not be quantified because these inshore fishing vessels were often operating without using their AIS.

Changes in harbour porpoise occurrence and background noise levels in relation to construction and vessel activity

A subset of piling locations (19 at Beatrice and 12 at Moray East) with a minimum of 48 h between piling events was chosen to examine the effect of pre-piling activities on harbour porpoises. Within the study area, the observed return time after pile-driving and deterrence activities was < 48 h (max: 40 h within 2.5 km from source, Graham pers.

comm.) and was < 24 h after seismic surveys (Thompson et al. 2013). In the Danish North Sea, the effect of piling activities on porpoise occurrence was observed up to 24-72 h at 2.6 km from the source (Brandt et al. 2011). Thus, I assumed, this ≥ 48-hour gap should have given enough time for harbour porpoises to return to the site after piling before subsequent piling events. However, this resulted in a smaller sample size for Moray East than Beatrice due to the overall faster piling rate and the lower sampling effort at Moray East. Nonetheless, this conservative approach enabled us to estimate the magnitude of harbour porpoise response induced by vessel arrival and preparation work. For locations with < 48-hour gap between piling events, the cumulative effects of pre- and post-piling activities are likely to further reduce harbour porpoise occurrence in the vicinity of the construction site prior the next piling event.

In German waters, during eleven windfarm developments, a 14-16% decrease in porpoise detection rates was observed in the three hours before piling within a 5 km buffer around construction sites (Rose et al. 2019). Rose et al (2019) suggested that this drop in porpoise occurrence was related to vessel traffic, but they were not able to quantify variation in vessel activity to explore this further. In this study area, harbour porpoise occurrence decreased by 20%, within a 5 km buffer around the piling location, in the 12 hours before the initiation of piling (Figure 3.6C). Using AIS vessel-tracking data, I was able to associate this decline with increased levels in vessel intensity, especially at Beatrice offshore windfarm (Figure 3.6B).

Furthermore, during the two weeks of noise measurements and for the subset of three piling locations, broadband noise levels increased between 14 h and 8 h before piling activities at Beatrice (Figure 3.8A). This peak in noise levels likely coincided with increased levels of vessel intensity and the start of louder activities e.g. running anchors (Figure 3.7A). However, no consistent patterns in noise levels from 24 h before piling were detected at Moray East. Based on the relatively small sample of three sites within each development, noise levels associated with Moray East were estimated to be 3.8 dB louder than at Beatrice (Figure 3.8B). Higher levels of broadband noise at Moray East may be due to differences in the noise profiles of construction activities at the two sites. *Technical Downtime,* at Moray East may, for example have included louder activities than those occurring at Beatrice. Alternatively, this difference may have been due to elevated ambient noise from natural or other anthropogenic noise sources, such as wind and wave action or undetected fisheries activity (Farcas et al. 2020). This limits direct comparison of the noise profiles of pre-piling activities in these two developments. Nevertheless, these indicative results highlight the need for additional work to improve characterisation of noise profiles both of different construction activities and of other natural and anthropogenic factors contributing to averaged ambient noise levels in these offshore environments.

When focussing on specific pre-piling activities, acoustic signatures were of the same order between the two windfarms, except for the positioning and securing phase. In this case, anchoring at Beatrice was estimated to be between 7 and 10.7 dB louder than jacking up and down at Moray East (Figure 3.8) but running anchors did not seem to be linked to lower porpoise occurrence (Figure 3.9A). On the other hand, even though the *Installation* phase and *Other Activity* did not generate significantly higher levels of noise in comparison with *Weather Downtime,* porpoise occurrence decreased by 22% and 26% respectively, at Beatrice (which was not the case at Moray East). Although these activities were associated with similar levels of broadband noise at both sites (Figure 3.8), vessel intensity was higher at Beatrice (Figure S 3.2) which may explain the contrast in porpoise response between the two windfarms. Alternatively, some differences may simply result from small sample sizes and unbalanced sampling effort, both between windfarms and between the datasets used to measure noise levels and porpoise responses. Finally, there were also differences in the way that activities were categorised in Daily Progress Reports from the two windfarms, particularly for the activities pooled within the *Other Activity* category (see Table S 3.1 and Table S 3.2). At Beatrice, *Other Activity* included pre- and post-installation ROV seabed surveys as well as debris and soil-plug removal and was associated with higher levels of vessel intensity (1.8 $\mathrm{min.km^{-2}}$) compared to *Weather Downtime* (0.7 $\mathrm{min.km^{-2}}$) (Figure S 3.2).

Porpoise deterrence may be triggered by increased levels of disturbance from a combination of factors. For instance, certain construction activities such as debris and soil-plug removal, analogous to dredging activities, may alter local habitat quality through increased suspended sediment (Pirotta et al. 2013; Todd et al. 2015; Culloch et al. 2016). Other construction (e.g. *Installation* phase) and vessel activities may generate high levels of noise in the frequency bands biologically relevant for porpoises which, consequently, may elicit adverse behavioural responses in porpoises (Hermannsen et al. 2014). However, the noise metric used initially to assess the acoustic signatures of different construction activities was unweighted (i.e. not audiogram weighted for harbour porpoises). This unweighted metric can inform developers, regulators and stakeholders of the overall broadband noise levels generated during specific activities. Alternatively, this metric can be converted into an audiogram weighted metric for the species of interest, which may provide further insight into the activity soundscapes likely to disturb this species. For further studies, I suggest deploying simultaneously noise recorders along a distance gradient, so the range of distances to the noise source will be greater, allowing the models to fit better the noise measurements (*cf.* Weather Downtime, Figure 3.7B). On the other hand, high levels of background noise at construction sites have the potential to reduce the porpoise detection probability of devices. However, the method used in this study aimed to reduce this masking limitation, and thus elevated ambient noise levels were unlikely to have affected porpoise echolocation click detection. Finally, lower detections of harbour porpoises within construction sites may be due to animals ceasing vocalising (Wisniewska et al. 2018) rather than being displaced. Although it is unlikely that a species with high vocalisation rates, such as porpoises (Sorensen et al. 2018), would stop vocalising for several hours, this behavioural response may result in underestimating the levels of porpoise occurrence at construction sites before initiating piling activities. That is why the use of deterrent devices, prior to piling, might still be required, but could be optimised (Thompson et al. 2020).

Despite differences in vessel intensity, noise levels and baseline detection rate, the proportion of decline in porpoise detections was similar for both windfarms. Harbour porpoises may therefore respond to higher levels of disturbance from increased vessel traffic and/or underwater noise levels associated with specific activities that were common to both developments. Alternatively, the gradual decrease in porpoise occurrence at both windfarms may represent a response to the cumulative impact of several construction activities (Graham et al. 2019). An individual's decision to leave and/or return to an exposed area likely depends on its fitness, energetic status and perception of predation risk (Frid and Dill 2002; Beale and Monaghan 2004). Some activities may trigger a stereotypical response, displacing some animals away from the noise source, while other individuals remain in the exposed area. Although the magnitude of response is likely to be site specific and context dependent (Gill et al. 2001; van Beest et al. 2018), these results can be used to modify baseline estimates of animal densities within the predicted impact zones at the start of piling. This, in turn, can support efforts to optimise mitigation measures that seek to minimise the risk of near-field injury from pile-driving.

Implications for developers and regulators

Efforts to reduce potential impacts of pile-driving on marine mammals have typically focussed on reducing noise levels at source (Dähne et al. 2017) or ensuring that animals are either not present (JNCC 2010) or deterred (Brandt et al. 2013a) from the site. However, these measures to mitigate any risk of near-field injury may result in other environmental pressures; for example, where use of ADD may increase far-field disturbance (Brandt et al. 2013b; Thompson et al. 2020) or noise abatement results in additional vessel traffic. Similarly, area/time disturbance thresholds used in the English waters (JNCC 2020) may reduce the spatial footprint of piling at any one time, but extend the temporal spread of construction activity. My results help inform the trade-off between these different impacts, supporting efforts to optimise mitigation measures to reduce overall cumulative impacts.

Further, the use of jack-up, autonomous, vessel may help reduce the overall levels of vessel intensity and vessel noise in the vicinity of the construction, as the piling vessel would not require the support of guard vessels for running and picking up anchors. In future, ancillary and support vessels' behaviour and spatial distribution could be monitored to estimate the spatial footprint of disturbance from these vessels in the vicinity of construction sites, and vessel management plans developed to reduce potential impacts. Similarly, assessing levels of noise and vessel intensity from other anthropogenic activities such as fisheries would help disentangle and quantify noise levels from activities unrelated to windfarm construction. This could also help ensure that buffer areas around construction sites provide lower levels of disturbances where harbour porpoises displaced from ensonified areas can forage.

To mitigate the risk of instantaneous death and injury to marine mammals during pile-driving activities, the use of ADD was integrated into the engineering processes during the Beatrice and Moray East piling campaigns (Thompson et al 2020). Although my study indicates that harbour porpoises were still acoustically detected in the vicinity of construction sites prior to the start of piling, the overall detection rate dropped by up to 20% from 12 hours before to the start of piling and ADD activities. These results can be used to modify estimates of the number of individuals likely to be in the vicinity of construction sites prior to piling activities to optimise mitigation measures. Ultimately, given the magnitude of harbour porpoise responses to different construction activities and varying levels of vessel intensity may be site- and context-specific, I emphasize the importance of baseline studies to estimate the “reference” likelihood of porpoise presence and predict the risk associated with varying levels of disturbance.

APPENDIX

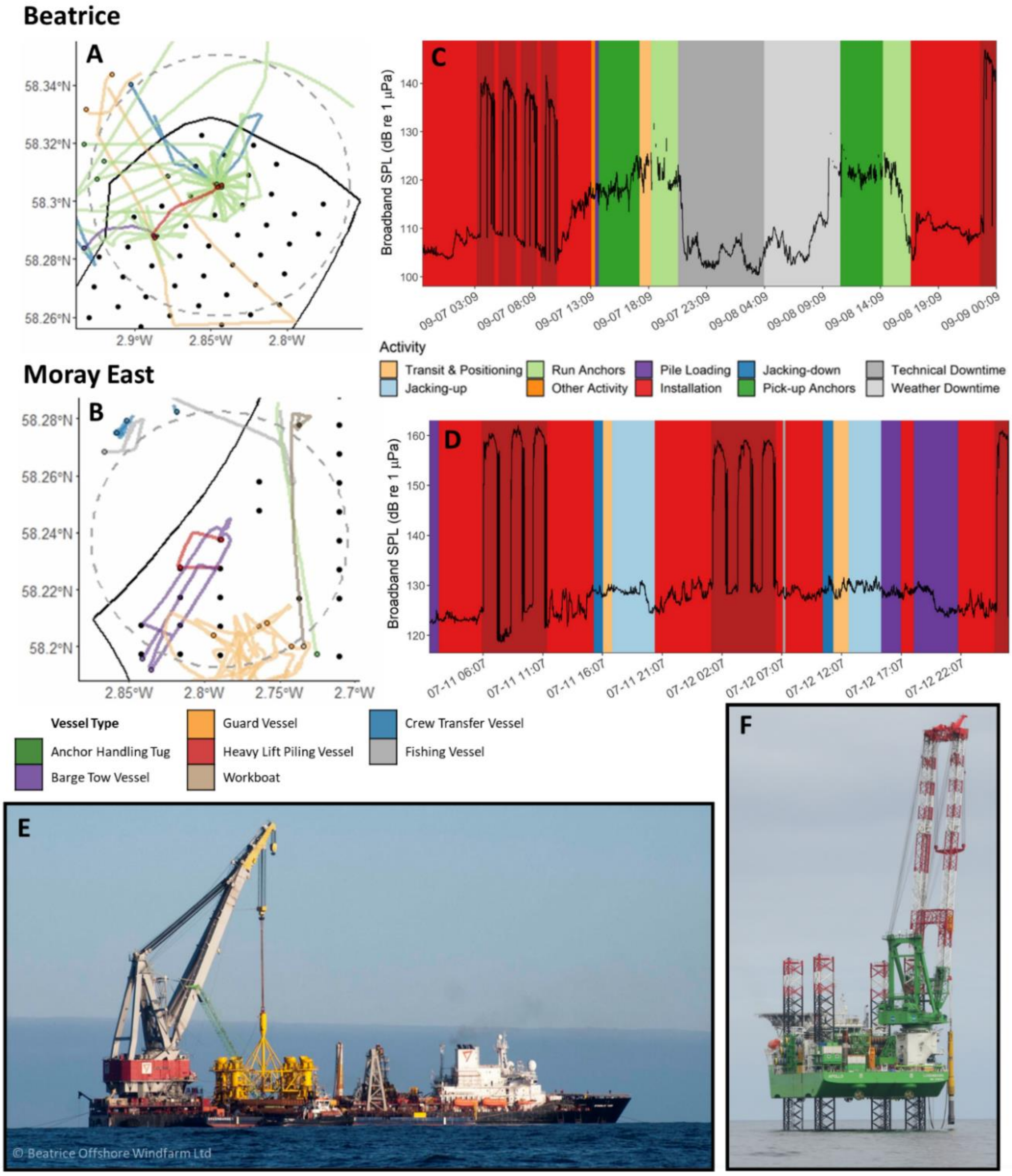

Figure S 3.1 Example of the spatio-temporal distribution of vessels in the vicinity of a turbine location from 48 hours prior to the start of pile-driving at A) Beatrice, B) Moray East offshore windfarms, NE Scotland; the lines are individual vessel tracks and each colour represents a vessel type; the coloured dots represent vessel position based on AIS data, while the black dots are the turbine locations; the dotted grey circles are the 5 km buffer zones around a turbine location, while the black line is the windfarm boundary. C) Example of broadband noise level timeline at C) Beatrice and D) Moray East offshore windfarms; each coloured rectangle represents an activity E) Picture of the piling vessel deploying the pile installation frame (PIF) at Beatrice offshore windfarm; F) Picture of the piling vessel deploying the hammer at Moray East offshore windfarm

Table S 3.1 Description of the various activities recorded on the piling vessel Daily Progress Reports, from 22nd April to 2nd December 2017, during the piling campaign, at Beatrice offshore windfarm

Activity	Beatrice DPR Code	Definition
Weather Downtime	WOW WOW*	Weather downtime
Technical Downtime	BRD	Breakdown of equipment
Transit & Positioning	REPO	Reposition HLV to next location
	SALC Transit	Sail to field/ location
Run Anchors	RUAN	Run Anchors and Position HLV
Pile Loading	MRCB	Moor cargo barge a/s HLV
	UMCB	Unmoor cargo barge
Jacking-up	-	-
Installation	DRPI/ DRPIL	Install hammer onto pile, install follower, soft start etc. (Drive Piles, ADD → *not in this study*)
	INPI	Lift and install piles
	INPIF INPIF/PRE_SUB	Lift, install and position PIF
	RMPIF RPIF	Retrieve PIF to HLV deck
	POSV	Move-in / position HLV into installation position
Other Activity	SUBM	Perform pre-installation ROV seabed survey
	POST_SUBM PMMC	Perform post-installation ROV seabed survey
Pick-up Anchors	PUAN	Move-out HLV and pick-up anchors
Jacking-down	-	-

Table S 3.2 Description of the various activities recorded on the piling vessel Daily Progress Reports, from 19th May 2019 to 27th February 2020, during the piling campaign, at Moray East offshore windfarm

Code	Activity
Installation	1st, 2nd or 3rd pile: Drive pile to intermediate depth 1st, 2nd or 3rd pile: Drive pile to target depth 1st, 2nd or 3rd pile: Hook/reset lifting tool, mount on pile, upend pile 1st, 2nd or 3rd pile: Overboard pile, Lower, position in Template, stab pile 1st, 2nd or 3rd pile: Prepare and install hammer on pile 1st, 2nd or 3rd pile: Release lifting tool, recover to deck 1st, 2nd or 3rd pile: Remove hammer, lift to next pile 1st, 2nd or 3rd pile: Remove hammer, return to deck 1st, 2nd or 3rd pile: Self weight penetration with Hammer Drive to intermediate depth Installation of Hammer and Follower

Code	Activity
	Lift piling template, position on seabed Lower template to seabed MS: Installation Complete On deck works requiring main crane Pile driving Pile Insert ILT, Upending, Stabbing Pile survey Recover Subsea Pile Template Rigging, Upending, Connecting follower & Lifting Hammer Soil plug removal Survey checks Template deploy Template recover Toolbox talk
Jacking-down	Jacking-down
Jacking-up	Jacking up Jacking up to height & pre-loading
Other Activity	(Un)seafastening main crane Auxiliary Activities & Operational Stanby Drilling - Tooling in operation Intermediate Dredging - Connect tooling Intermediate Dredging - Deploy tooling Intermediate Dredging - Tooling in operation Maintenance Operational standby Other Activity Template Inspection
Other Downtime	Crew change Other Downtime (specify in comments) Waiting on Piles
Pile Loading	1 No. Pile Lifting and Set Down 2 No. Pile Lifting and Set Down 3 No. Pile Lifting and Set Down 4 No. Pile Lifting and Set Down 5 No. Pile Lifting and Set Down 6 No. Pile Lifting and Set Down 7 No. Pile Lifting and Set Down Load-out Others Load-out Prepile Load-out Prepile - Connect/Disconnect spreader bar and Cranemaster Load-out Prepiles-PSV approach PSV Positioning
Technical Downtime	Breakdown Technical breakdown of equipment Technical breakdown of vessel
Transit & Positioning	Positioning Prepare for transit Sailing and Positioning Sailing to location

Code	Activity
	Transit to site
Weather Downtime	Waiting on lightning
	Waiting on wave height
	Waiting on wave period
	Waiting on wind speed
	Weather standby - Other
	Weather standby - Swell
	Weather standby - Wave standby
	Weather standby - Wind standby

Table S 3.3 Total and proportion of construction activity duration (min; %) during the piling campaigns, at both the Beatrice and Moray East offshore windfarms; mean and median duration (h) per piling sites with sample size indicated in the column "N turbines"; rank represent the decreasing order of overall activity duration per windfarm.

	Beatrice (22/04 - 02/12/2017)				**Moray East (19/05/2019 -27/02/2020)**			
Activity	Total duration (min; %)	rank	Mean (median) duration (min)	N turbines	Total duration (min; %)	rank	Mean (median) duration (min)	N turbines
Weather Downtime	146,984 (47.16%)	1	1,754.05 (941.5)	20	155,189 (38.06%)	1	2,821.62 (1,145)	55
Technical Downtime	31,228 (10.02%)	3	1,307.71 (359)	21	37,469 (9.19%)	3	1,135.42 (180)	33
Transit & Positioning	7,190 (2.31%)	6	50.94 (45)	63	17,319 (4.25%)	6	168.47 (137.5)	102
Run Anchors	13,019 (4.18%)	5	169.86 (150)	77	-	-	-	-
Pile Loading	6,006 (1.93%)	7	99.73 (80)	60	15,335 (3.76%)	7	278.82 (257)	55
Jacking-up	-	-	-	-	34,534 (8.47%)	5	335.28 (252)	103
Installation	82,888 (26.59%)	2	1,050.37 (1,029)	79	101,768 (24.96%)	2	988.04 (811)	103
Other Activity	5,686 (1.82%)	8	87.42 (60)	65	35,236 (8.64%)	4	367.04 (179.5)	96
Pick-up Anchors	18,670 (5.99%)	4	254.54 (230)	76	-	-	-	-
Jacking-down	-	-	-	-	9,712 (2.38%)	8	95.22 (83)	102
Total	311,671 (100%)			79	407,748 (100%)			103

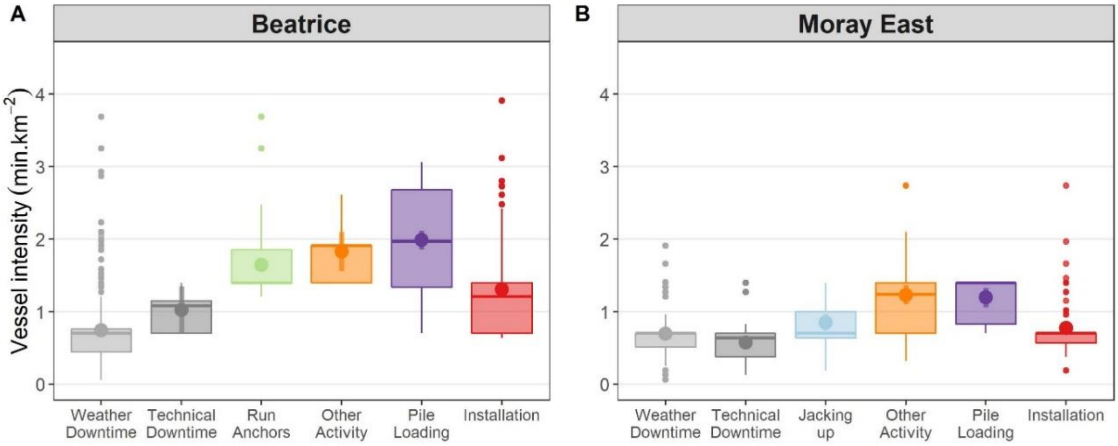

Figure S 3.2 Observed (boxplot and small dots) and estimated (big dots) vessel intensity (besides the piling vessel) per hours relative to piling during which at least 70% of the same activity, in relation to the activity type conducted by the piling vessel at Beatrice (A) and Moray East (B); the 95% confidence intervals (line range) estimated for the uncertainty in fixed effects; At Beatrice, in comparison with Weather Downtime, levels of vessel intensity were significantly higher during the other activities. At Moray East, levels of vessel activity were higher only during Pile Loading and Other Activity than during Weather Downtime, whereas no differences were observed between Jacking-up, Installation and Weather Downtime.

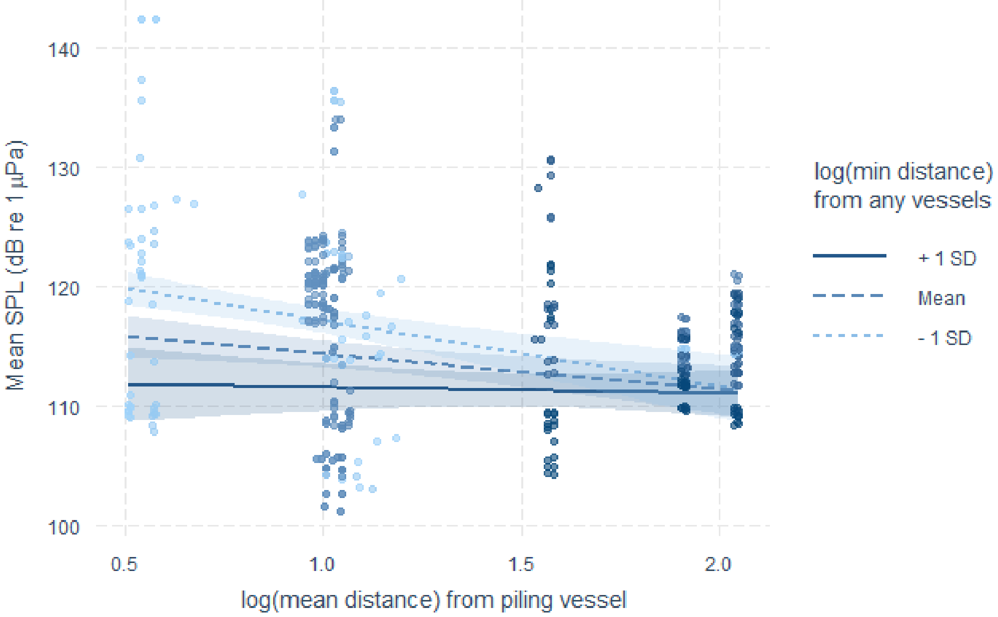

Figure S 3.3 Linear fitted mean broadband Sound Pressure Levels per hour in relation to the log-transformed mean distance to the piling vessel and the log-transformed minimum distance to any vessels at the Beatrice and Moray East offshore windfarms. This graph indicates the closer the piling vessel is from the noise recorders, the higher noise levels are. However, when other vessels are close to the noise recorders, higher levels of noise are recorded highlighting the potential masking effect of other vessels on detecting construction noise from the piling vessel.

Table S 3.4 General information about the short-term noise recorder deployments at both the Beatrice and Moray East offshore windfarms

Windfarm	Deployment	Location	Device Type	Recording (ON/TOTAL) period (mins)	Pre-amp gain	Sample rate (kHz)	Latitude	Longitude	Date data started	Date data ended	Depth (m)
Beatrice	903	201	SoundTrap	1/10	LOW	576	58.31848	-2.86053	05/09/2017	13/09/2017	55.8
Beatrice	905	202	SoundTrap	1/10	HIGH	576	58.31893	-2.86087	05/09/2017	24/09/2017	56.1
Beatrice	907	203	SoundTrap	1/10	LOW	576	58.31938	-2.86128	05/09/2017	13/09/2017	55.8
Moray East	1055	247	SoundTrap	No duty cycle	HIGH	48	58.23463	-2.76537	10/07/2019	23/07/2019	44.5
Moray East	1056	248	SoundTrap	No duty cycle	HIGH	48	58.23487	-2.76492	10/07/2019	23/07/2019	44.4
Moray East	1057	249	SoundTrap	No duty cycle	HIGH	48	58.23522	-2.76410	10/07/2019	23/07/2019	44.5

Table S 3.5 Sample size per hour relative to piling at the Beatrice and Moray East offshore windfarms; These tables refer to the sample size used in Figure 3.6; The variation in sample size between hours relative to piling for each windfarm is due to the selection of hours with 60-min PAM data only and with no vessels within 1 km buffer of PAM sites.

Windfarm *(N hours)*	**-48**	**-47**	**-46**	**-45**	**-44**	**-43**	**-42**	**-41**	**-40**	**-39**	**-38**	**-37**	**-36**	**-35**	**-34**	**-33**	**-32**	**-31**	**-30**	**-29**	**-28**	**-27**	**-26**	**-25**	**-24**
Beatrice	167	178	199	184	168	170	173	181	172	170	179	211	216	195	190	194	185	178	160	177	189	200	215	179	172
Moray East	45	36	38	44	41	43	44	38	41	40	36	35	36	36	37	37	45	46	42	40	36	37	44	54	45

Windfarm *(N hours)*	**-23**	**-22**	**-21**	**-20**	**-19**	**-18**	**-17**	**-16**	**-15**	**-14**	**-13**	**-12**	**-11**	**-10**	**-9**	**-8**	**-7**	**-6**	**-5**	**-4**	**-3**	**-2**	**-1**	**0**
Beatrice	191	201	192	172	166	176	169	157	167	166	204	210	182	176	190	169	153	155	159	161	183	194	166	163
Moray East	37	39	41	40	42	44	37	41	38	36	35	36	35	37	37	46	43	42	40	36	36	47	53	42

CHAPTER 4

CHANGES IN MESO-SCALE DISTRIBUTION OF SEABIRDS DURING CONSTRUCTION OF AN OFFSHORE WINDFARM

CHAPTER 4

CHANGES IN MESO-SCALE DISTRIBUTION OF SEABIRDS DURING CONSTRUCTION OF AN OFFSHORE WINDFARM

ABSTRACT

Increasing demand for sustainable energy has led to rapid offshore windfarm development worldwide, resulting in concern over impacts on protected seabird populations. Pre-construction surveys are required to assess baseline distributions and model collision risk and displacement of these species. However, construction may take several years and changes in baseline once substructures are installed may confound assessments. I used digital aerial survey data to investigate variation in seabird distribution and abundance across an offshore windfarm site throughout construction. Observed distances of auks, large gulls, terns and small gulls to the nearest turbine location were compared to pre-construction baselines and randomised datasets. Increased numbers of sightings were observed during and after construction. Auk, tern, and small gull species exhibited no avoidance or attraction to offshore structures. However, during construction, large gulls were strongly attracted to jacket structures in the year before turbines were installed, with 72% of birds observed perched on jackets. Roosting behaviour was not observed after final construction, but high numbers of large gulls were observed on the water < 250 m from turbines. This unforeseen shifting baseline distribution, part-way through construction, should be considered when assessing vulnerability of large gulls to collision risk once turbines are installed.

INTRODUCTION

Extensive research has been conducted to improve quantitative assessments of potential impacts of offshore windfarm developments on protected seabird populations (e.g. Furness et al. (2013); May et al. (2015); Dierschke et al. (2016); Cook et al. (2018)). In particular, risk of collision with wind turbine blades could lead to population level impacts (Green et al. 2016), but there are also concerns over displacement from windfarms sited in key foraging areas (Dierschke et al. 2016). Collision Risk Models used in Environmental Impact Assessments (EIAs) for regulatory assessment generally use baseline distributions derived from surveys conducted several years before construction (Desholm and Kahlert 2005; Chamberlain et al. 2006; Bailey et al. 2014; Thaxter et al. 2018). However, during the planning and assessment period, seabird distribution, abundance or behaviour could change due to natural or anthropogenic variation in environmental conditions (Harwood et al. 2017). Significant changes in baseline throughout the windfarm construction may therefore bias predicted collision risk probabilities and confound regulatory assessments.

In UK waters, developers are typically required to assess whether offshore windfarms influence distribution by comparing data from surveys made before construction starts with data from surveys made once the windfarm is operational. However, the construction process may take several years, and baseline distributions may change before the final installation of turbines results in a risk of collision. In this study, I was able to explore changes in seabird distribution at finer temporal scales by integrating survey data from two adjacent developers with overlapping, but asynchronous, survey programmes. I used this unique opportunity to assess variation in the horizontal meso-scale (< 1 km) distribution of seabirds around offshore wind turbine locations, before, during (i.e. following jacket foundation installation) and after construction of an offshore windfarm.

MATERIAL AND METHODS

Study area and context

This study focussed on the construction of the Beatrice offshore windfarm on the Smith Bank, NE Scotland; an important foraging area for seabirds breeding in the East Caithness Cliffs Special Protection Area (Mudge et al. 1984; Mudge and Crooke 1986). Between March 2017 and May 2019, 84 7-MW wind turbines, each with a tip height of 165 m above sea level, were installed on four-legged jacket structures at Beatrice. The mean distance between turbines was 1,157 m [946; 1,171 m]. The adjacent Moray East offshore windfarm started construction in 2019 and became fully operational in 2021 (see Benhemma-Le Gall et al. (2021) for more details on windfarm construction) (Figure 4.1).

Digital aerial surveys

I used digital aerial surveys conducted in 2015, 2018 and 2019 to quantify changes in the meso-scale spatial distribution of seabirds around individual turbines within the Beatrice windfarm site (Table 4.1). Ornithology monitoring at Beatrice involved six pre- and six post-construction digital aerial surveys during the 2015 and 2019 breeding seasons, each surveying an area from the Caithness coast to the Smith Bank, passing over the windfarm and a 4 km buffer area (Figure 4.1). Digital aerial surveys were carried out by HiDef Aerial Surveying Limited (HiDef), flying at an altitude of 500-588 m, with two or four forward-looking high-definition video cameras. Within a 2 km buffer of the Beatrice site boundary, four cameras were used giving a total transect width of 500 m (Buckland et al. 2012; Williamson et al. 2016).

During the intervening 2018 breeding season, three pre-construction digital aerial surveys were conducted over the Moray East windfarm site and a 10 km buffer zone which included the Beatrice site. These surveys were conducted by APEM Limited from an altitude of 350-442 m, capturing high-definition digital still images along 500 m wide transects. In all surveys, seven transects were flown over the Beatrice site, oriented along the wind turbine rows, perpendicular to the nearest coastline (Figure 4.1A). Four aerial transects passed over rows of turbines (n = 19 turbines), while three transects passed between rows of turbines (Figure 4.1B). Both digital still images and digital video provided geo-referenced images of 2 cm resolution. Birds were recorded on several frames (1 to 2 frames for still images and 7 to 10 frames for video), maximizing detection probability of birds on the surface or flying and allowing uniform detection of seabirds across the transect (Webb and Nehls 2019). Surveys were conducted at sufficiently high altitude to avoid disturbance while allowing detection and identification of seabirds (Buckland et al. 2012).

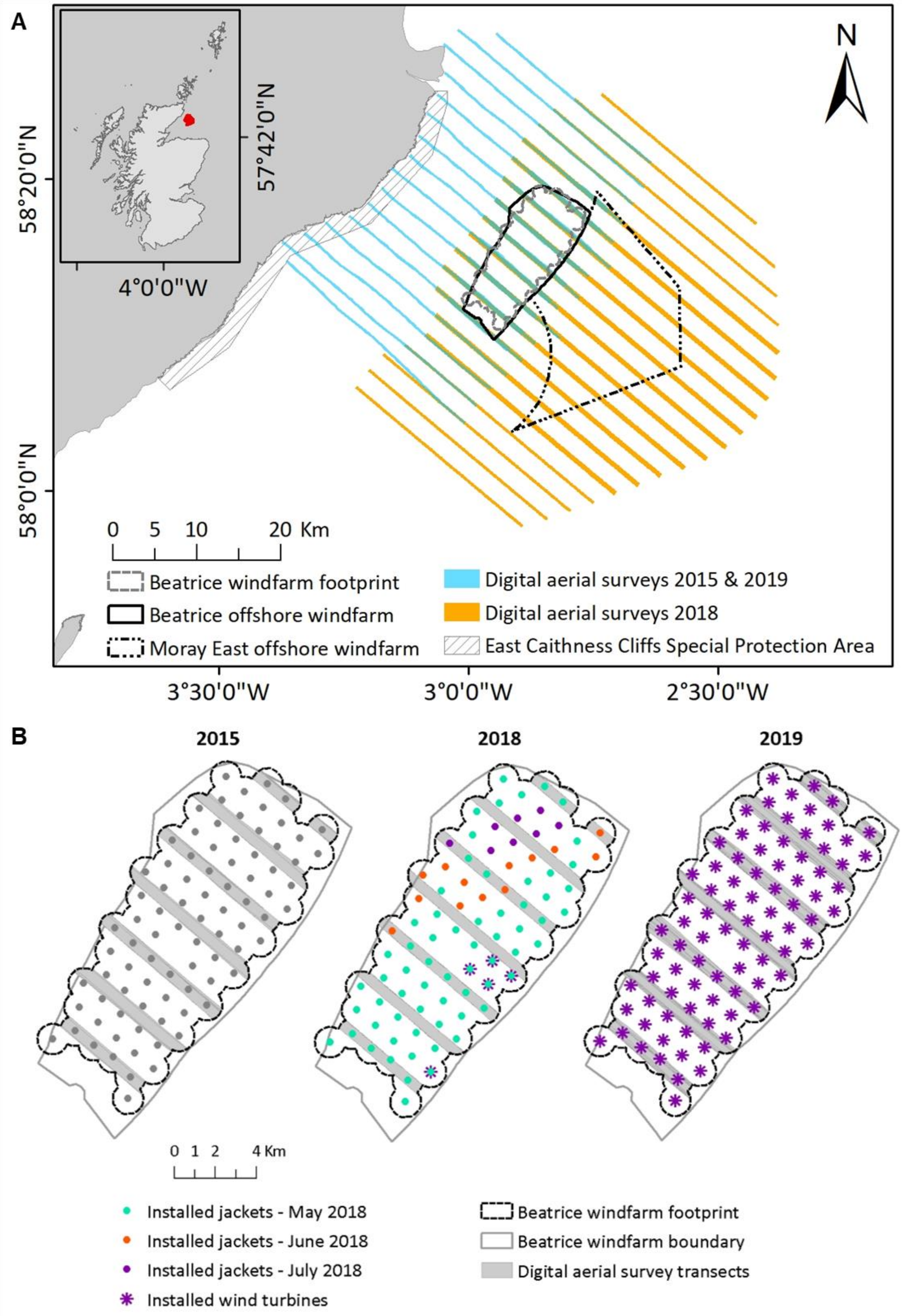

Figure 4.1 A) Map of Scotland with the two offshore windfarms in development in the outer Moray Firth; Strip transects of the Beatrice offshore windfarm digital aerial surveys in blue and of the Moray East offshore windfarm in orange; B) Beatrice offshore windfarm boundary and footprint area

Digital images were post-processed and analysed using industry standard approaches within HiDef and APEM (Buckland et al. 2012; Moray Offshore Renewables Limited 2012c, d). Each seabird sighting was georeferenced and identified to species or species group with an associated confidence level. Given the number of individuals not identified at the species level (HiDef 8%; APEM 21.5%), sightings were grouped based on taxonomic characteristics. The non-identified species sightings not fitting into these groups were removed from the datasets (< 1%). Here, I consider only the three species groups *Auk species*, *Tern and Small gull species* and *Large gull species*, with an overall sample size > 1,000 sightings (Table 4.1).

Table 4.1 Classification of seabirds sighted within the windfarm footprint, during digital aerial surveys in the outer Moray Firth, based on their taxonomic characteristics and feeding behaviour

Species groups used for the study (*Common feeding method*)	**Sightings identified at the species level**	**2015**		**2018**		**2019**	
		N	**Density** (N/km^2)	***N***	**Density** (N/km^2)	***N***	**Density** (N/km^2)
Auk species *(pursuit diving)*	Common guillemot	*1735*	13.10	*611*	9.21	*5297*	40.01
	Black guillemot						
	Razorbill	*43*	0.32	*100*	1.51	*372*	2.81
	Atlantic puffin	*156*	1.18	*23*	0.35	*37*	0.28
	Non-identified	*169*	1.28	*753*	11.35	*883*	6.67
	Total	***2103***	**15.88**	***1487***	**22.41**	***6589***	**49.77**
Tern & Small gull species *(dipping, shallow plunging, catalyst)*	Black-legged kittiwake	*602*	4.55	*771*	11.62	*1753*	13.24
	Common gull	-	-	*15*	0.23	*7*	0.05
	Arctic tern	*25*	0.19	-	-	-	-
	Non-identified	*8*	0.06	*3*	0.05	*63*	0.48
	Total	***635***	**4.80**	***789***	**11.89**	***1823***	**13.77**
Large gull species *(klepto/surface seizing, scrounger)*	Great black-backed gull	*6*	0.05	*48*	0.72	*14*	0.11
	Herring gull	*4*	0.03	*394*	5.94	*561*	4.24
	Lesser black-backed gull	-	-	*6*	0.09	*8*	0.06
	Non-identified	*1*	0.01	*15*	0.23	*13*	0.10
	Total	***11***	**0.08**	***463***	**6.98**	***596***	**4.50**

Data processing and analysis

In 2015, no man-made structures were present at the Beatrice windfarm site. During the 2018 digital aerial surveys, the number of jacket foundations installed increased. There were 65 jackets installed in May 2018, 78 in June 2018 and by July 2018 all 86 jackets were installed. In 2019, the 84 wind turbines were installed and fully operational.

To calculate the minimum distance between a sighting and the nearest offshore structure (either planned or installed), georeferenced sightings and wind turbine locations were projected into WGS84 UTM 30N, using the *st_transform* function of the *sf* package in R (Pebesma 2018; R Core Team 2019). The Beatrice footprint was defined as the area limited by a 750 m buffer around the turbines located at the edge of the windfarm site, resulting in a total area of 111 km^2 (Figure 4.2). The minimum distance between each sighting and the nearest turbine location was calculated using the *st_distance* function of the *sf* package. For the 2018 dataset, the jacket installation timeline was considered when calculating minimum distance to structures for each survey during that year. As the number of sightings between the three monitoring years was low and/or unbalanced for certain species groups, I created a simulated dataset consisting of 1,000 seabirds randomly distributed within those strip-transects that were located within the windfarm footprint. As with the observed data, I then calculated the minimum distance between each simulated point and the nearest turbine location. Only the sightings within 750 m of structures (> 96%) in the windfarm footprint were kept for further analyses.

Randomization tests were then conducted to assess whether construction stage influenced seabird distribution around man-made structures. These tests were used to compare observed distances of seabirds to the nearest planned or installed turbine location between construction stages (*i.e.* before, during and after jacket and turbine installation) and with a random distribution. The advantage of these randomization tests is that they are distribution-free methods that are robust to unbalanced and small samples, and do not require assumptions of normality, heteroscedasticity and independence required in linear models (Peres-Neto and Olden 2001; Osiecka et al. 2020). First, I randomly assigned each level of the explanatory variable (*i.e.* construction phase) to the original dataset, keeping the same sample size for each category. A test statistic was then calculated between every two levels of the randomised dataset, and the output was stored after each of the 100,000 replicates. If the construction phase has little power to explain variation in seabird distribution, then randomising it would have little effect on the test statistic (Osiecka et al. 2020). The observed test statistic (*e.g.* the difference between the mean distance of two construction phases) and the randomised test statistic were compared and a p-value was calculated as the proportion of replicates that have an absolute randomised test statistic value greater than or equal to the absolute observed test statistic value (Osiecka et al. 2020).

Behaviour of seabirds

Seabird behaviours identified from both HiDef and APEM imagery were homogenised and divided into the following behaviour groups: *Surfacing* (incl. sitting, diving and taking off), *Flying* or *Perched* on man-made platforms. For each monitoring period and species group, the proportion of birds exhibiting each behaviour was calculated. To assess whether the behaviour of seabirds significantly changed between monitoring periods, Chi-squared tests were performed, on a contingency table (*i.e.* Behaviour *vs* Construction phase) for each species group.

RESULTS

Over 2015, 2018 and 2019, a total of 15 surveys (2015 n = 6; 2018 n = 3, 2019 n = 6) were conducted. For each survey, seven transects passed within the windfarm footprint, surveying 132 km^2 in 2015 and 2019 and 66 km^2 in 2018. For the three focal species groups, the number of sightings within the windfarm footprint increased during and after jacket and turbine installation (Table 4.1), although the significance of this increase could not be assessed using this design. More importantly, the distribution of seabirds around structures differed between construction stages, with the magnitude of changes varying between focal species groups. Before the windfarm construction, distributions of each of the three focal species groups around planned turbine locations were similar to a randomised/null distribution, ranging between 410 m and 430 m (two-tailed randomisation test: n = 100,000, p-values ranging between 0.25 and 0.98). Furthermore, although auk distributions around turbines were significantly different between each construction stage (two-tailed randomisation test: n = 100,000; During *vs* Before, p = 0.01; After *vs* Before, p = 0.004; After *vs* During, p < 0.001), the magnitude of these changes remained small (< 20 m; Figure 4.2A). Most auks were observed sitting on the water and this behaviour did not change between construction phases ($\chi^2 = 2.51$, df = 2, p = 0.29). In comparison to the baseline distribution (*i.e.* Before), the mean distance of tern and small gull sightings from the jacket foundations/turbines decreased by 100 m during construction and increased by 30 m after construction (two-tailed randomisation test: n = 100,000; p-values < 0.001) (Figure 4.2A). Their behaviour mostly changed from *Flying* to *Surfacing* between the pre- and post-construction phases ($\chi^2 = 27.968$, df = 2, p < 0.001) (Figure 4.2B). In comparison to the baseline phase (*i.e.* Before), large gulls were significantly closer to offshore structures and were on average, 20 m away from the jacket foundations during construction (two-tailed randomisation test: n = 100,000; p < 0.001). Although, large gull sightings were, on average, 250 m (± 150 m) away from turbines after construction, there was no significant difference in their distribution before and after construction (two-tailed randomisation test: n = 100,000; p = 0.15) (Figure 4.2A). During construction, 72% of large gull sightings (66% Herring gull; 6% Great black-backed gull) were perched on jacket foundations. Perching behaviour was not observed after construction, when most large gulls were observed sitting on the water ($\chi^2 = 199.403$, df = 4, p < 0.001) (Figure 4.2B).

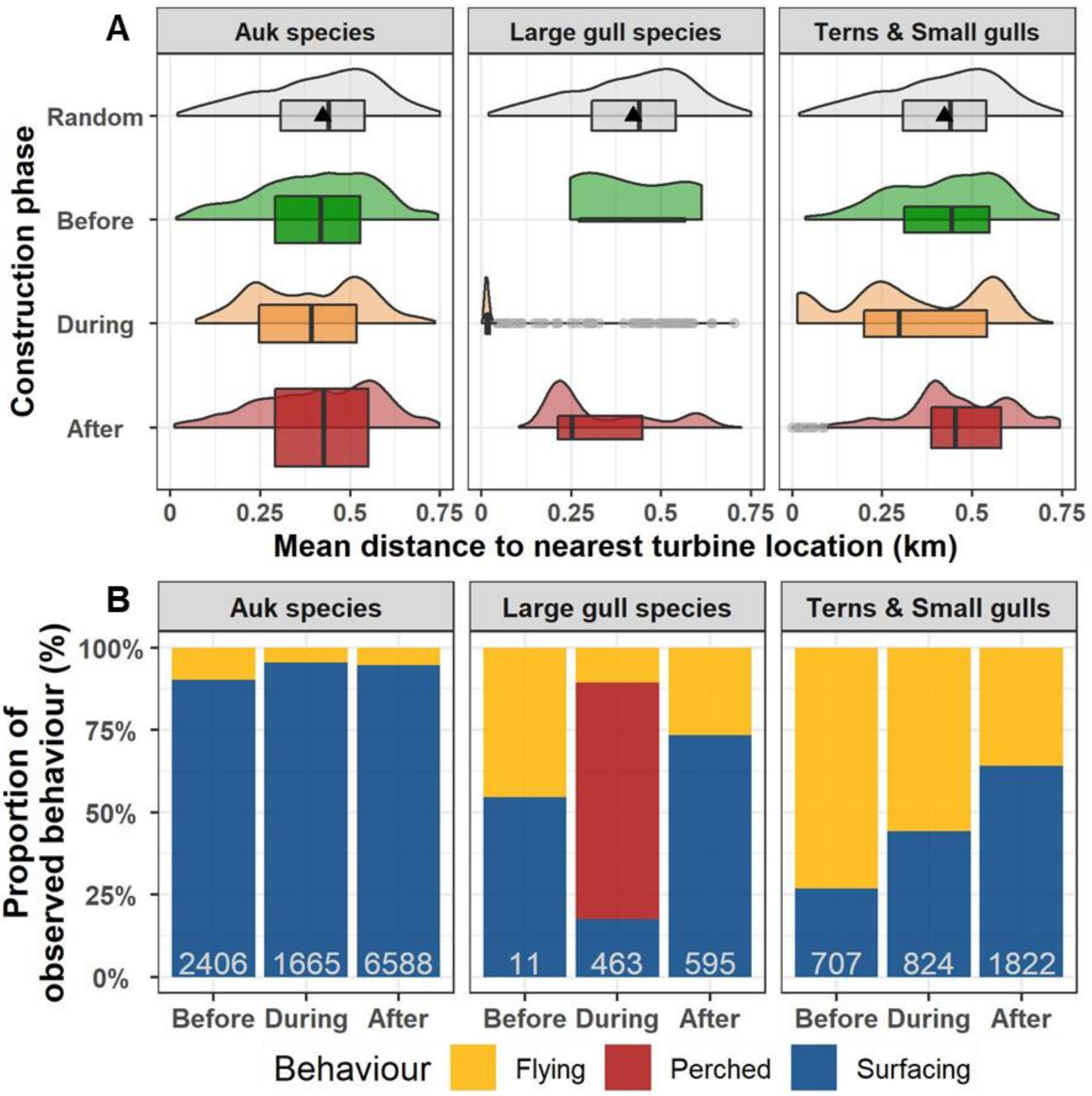

Figure 4.2 A) Boxplot and density distribution of the minimum distance between sighted seabirds from each focal species group and the nearest jacket foundation/planned turbine location before (green), during (orange), after (red) the Beatrice offshore windfarm construction and when seabirds are randomly distributed in the surveyed transects within the windfarm footprint (grey); the black triangle represents the mean minimum distance for a randomly distributed population of 1,000 sightings after 200 iterations. B) Proportion of the three main behaviours observed for each seabird species group before, during and after the Beatrice offshore windfarm construction. For each species group, the sample size is indicated at the bottom of each bar.

DISCUSSION

Changes in seabird distribution and behaviour around individual offshore structures during and after offshore windfarm construction were group-specific. Auk species showed little to no response to offshore structures during and after construction. Auks are common species with relatively homogeneously distributed numbers, implying that even for moderate avoidance levels, changes in distribution are often significant (Perrow 2019). Similarly, common guillemots did not change their distribution within the footprint of the UK Robin Rigg windfarm during the breeding season. As central-place foragers, auks may be constrained to use specific foraging areas that are relatively close to breeding colonies and, therefore, may be less ready to displace (Vallejo et al. 2017).

Large gull species abundance, mainly herring gulls, increased markedly during and after the windfarm construction. They showed a strong attraction to structures during jacket installation as they perched on the platforms. As no wind turbines had been erected by that stage, the jacket foundations offered increased roosting possibilities that may provide ecological incentive for breeding seabirds (Vanermen et al. 2015). The large gulls' attraction to structures has been reported at the Thornton Bank windfarm in Belgian waters, where 79% of the great black-backed gulls were observed roosting on turbine foundations, concentrating along the windfarm edge (Vanermen et al. 2020). Furthermore, even though no roosting behaviour was reported in the study area during the post-construction surveys, large gulls were still observed in large numbers sitting on the water within < 250 m of turbines (Figure 4.2). Tern and small gull species, mainly black-legged kittiwakes, showed a weak attraction to structures during but not in the first year after windfarm construction. Further strategic research investigating the variation in meso-scale responses of individual species or foraging guilds to offshore structures several years after windfarm commissioning is required, especially for species with high population vulnerability to collision mortality (*e.g.* black-legged kittiwakes and common gulls; Furness et al. (2013); Bradbury et al. (2014)). The introduction of hard structures (*i.e.* jacket foundations) into a soft-bottom marine ecosystem may result in a cascade of bio-physical changes with the colonisation and fouling of the hard lattice substrate, the development of hard-substratum communities and increased fine-scale vertical mixing in the wake of the foundation structures (Vanermen et al. 2015). Consequently, these structures may provide enhanced feeding opportunities for seabirds.

The increase in large gull sightings within the windfarm footprint and their strong attraction to offshore structures during construction may make large gull species more vulnerable to collision risk. The population vulnerability to collision mortality at offshore wind turbines of herring gulls, great and lesser black-backed gulls have been classified as “very high” in UK (Furness et al. 2013; Bradbury et al. 2014). Unforeseen by the EIAs, the shift in the baseline distribution, abundance and behaviour of large gull species during construction should be considered in the collision-risk framework to estimate more accurately the mortality rate of individual seabirds caused by collision with turbines and the potential population consequences of offshore windfarm developments.

CHAPTER 5

PREDATOR – PREY DISTRIBUTION IN THE VICINITY OF AN OPERATIONAL OFFSHORE WINDFARM

CHAPTER 5

PREDATOR – PREY DISTRIBUTION IN THE VICINITY OF AN OPERATIONAL OFFSHORE WINDFARM

ABSTRACT

Understanding and assessing the spatial association between predators, prey and their surrounding habitat is essential to assess how environmental changes associated with offshore developments influence marine top predator populations. Most impact assessment studies focus on monitoring single receptors without considering the potential effects of anthropogenic activities on the pelagic ecosystem. This study aimed to collect empirical data to characterise the underlying physical and biological processes driving variation in prey and seabird distribution and abundance in the vicinity of an operational offshore windfarm in the northern North Sea. During boat-based surveys conducted in June and August 2019, data on the water column structure, and the primary and secondary production were collected at sampling points, within- and outside the windfarm. In June, the low thermal variation of the water coupled with high concentrations of phytoplankton distributed homogeneously within the water column highlighted vertical mixing. In contrast, in August, waters were stratified, and the relative abundance of zooplankton greater, indicating that production had shifted towards higher trophic levels through the summer. Visual sightings of seabirds and fisheries acoustic data were collected simultaneously along five transects that passed between rows of turbines. I examined whether measurements of prey availability coincided with the abundance or occurrence of three seabird species groups. Prey density was higher in August but mainly distributed in the lower water layer. In June, patchy fish aggregations were near the surface, and thus likely to be more available for surface feeders. Variations in schooling prey patchiness and depth significantly influenced seabird distribution within the study site. Overall, the study provides empirical evidence that prey patchiness is an important factor influencing predator distribution in the pelagic environment in and around an operational windfarm. Further

work is now required to investigate the effect of offshore structures on fine-scale associations between predators and prey.

INTRODUCTION

Offshore renewable energy developments often overlap with habitats of high importance for marine top predators and their prey (Bailey et al. 2014). For instance, subtidal sandbanks are suitable sites for offshore windfarm developments and their contrasting topographical habitat have the potential to aggregate high levels of primary productivity and attract upper trophic level predators (Stevick et al. 2008; Embling et al. 2013; Scott et al. 2013). The introduction of man-made hard-substrate structures into soft-bottom marine environments is likely to change these offshore ecosystems locally and over time. To date, impact assessment studies mainly focus on quantifying the potential adverse impacts of offshore windfarm development on ecosystem components of conservation relevance (*e.g.* marine mammals, seabirds, diadromous fish), while only a few empirical studies have investigated the ecological effects of windfarms on the pelagic ecosystem (Floeter et al. 2017). For instance, the focus of marine top predator monitoring programmes has been to assess variation in the occurrence, spatial distribution or behaviour in relation to direct impacts of offshore windfarm developments (Harwood et al. 2017; Thaxter et al. 2018; Graham et al. 2019; Thompson et al. 2020; Vanermen et al. 2020; Whyte et al. 2020; Benhemma-Le Gall et al. 2021; Fernandez-Betelu et al. 2021), without accounting for potential changes in habitat quality and prey availability at these sites. However, changes in top predator distribution and abundance at windfarm sites may be driven by direct disturbance from windfarm construction and operation or indirectly via changes in prey availability.

Physical processes including currents, vertical mixing and stratification determine the spatial distribution of primary production. As a result, these hydrodynamic features can create complex, scale-dependent spatial partitioning in the resources available to secondary producers (zooplankton) and pelagic schooling fish, which are key prey for top predators such as seabirds (Fauchald 2009). While feeding in the water column during the day, forage fish can exhibit collective self-organizing behaviour and form schools to minimize the risk of predation (Partridge 1982). Although this adaptive anti-predatory behaviour makes pelagic schooling fish a highly elusive and patchy resource, once detected by foragers, functional aggregations of predators can exploit prey patches rapidly (*e.g.* Temming et al. (2007)). Thus, forage fish provide an important trophic link in marine ecosystems, as they efficiently transfer energy, converting secondary production into fish tissue which becomes available for higher order consumers (Van Der Kooij et al. 2008; Van Deurs et al. 2013; Mustamäki et al. 2016).

For predators, prey availability is conditioned by prey prevalence and density, prey depth and patch structure (Benoit-Bird et al. 2013) but also by the predators' abilities to detect and successfully catch prey (Chimienti et al. 2017). Moreover, prey profitability is determined by the energetic costs and benefits of foraging, which will be specific to a predator's foraging strategy, its flying and/or diving abilities (Goldstein 1990; Jodice et al. 2003; Elliott et al. 2013), prey consumption rates and prey energy content (Harding et al. 2009; Booth 2019). Further, individual energy requirements will vary seasonally and at specific life stages (Markones et al. 2010; Dunn et al. 2020). Consequently, predators, foraging in patchy environments and/or with varying energetic and life history constraints, should be able to exhibit flexible foraging strategies to optimise their foraging efficiency (Hazen et al. 2015). Thus, to understand predator distribution, it is essential to characterise the prey field in a way that is relevant to each predator guild or species (Benoit-Bird et al. 2013). Further, highly mobile air-breathing predators, such as seabirds, do not necessarily forage where prey are most abundant, but more likely where suitable prey is most accessible. Breeding seabirds, as central-place foragers, can move over relatively long distances around the breeding colony (see Thaxter et al. (2012)), but their prey are often elusive and may have a scale-dependent patchy distribution (Fauchald 2009). Spatial aggregations of resources, or patchiness, are likely to condition consumers' aggregative responses and, at small-scales, increased prey patchiness and abundance can enhance predators' foraging success (Fauchald and Erikstad 2002).

Ultimately, understanding predator-prey interactions at offshore development sites is going to be essential to assess the cumulative impacts of windfarms throughout their lifecycle. Studies at commissioned sites are therefore required to provide empirical information to support assessments of future developments at sites with similar habitat characteristics or adjacent to existing sites. Synchronous data collection, using boat-based active acoustic monitoring and visual surveys, helped identify key features of prey availability that concur with seabird distributions and unravel the processes driving predator-prey interactions (Embling et al. 2012; Cox et al. 2013; Waggitt et al. 2018; Lambert et al. 2019). The main aims of this study were to characterize the primary and secondary production in parallel with *in situ* hydrographic measurements, and to investigate the distribution of pelagic schooling fish and their seabird predators within and around an operational windfarm.

MATERIAL AND METHODS

Study area and boat-based surveys

In June (5th-11th) and August (27th-29th) 2019, two boat-based surveys were conducted, during daylight hours, onboard Marine Scotland's 27-m Research Vessel *Alba na Mara* at the Beatrice and Moray East offshore windfarms on the Smith Bank, NE Scotland. The Smith Bank is a large sand bank approximately 15 km offshore with water depths ranging between 35 and 68 m (Figure 5.1). The Beatrice offshore windfarm, which was operational during both surveys, was commissioned in May 2019 and comprises 84 (7 MW) wind turbines and two Offshore Transformer Modules (OTM) mounted on four-legged jacket structures. The Moray East offshore windfarm started piling activities on 19th May 2019 and four piling events occurred at the site during the surveys (two in June and two in August).

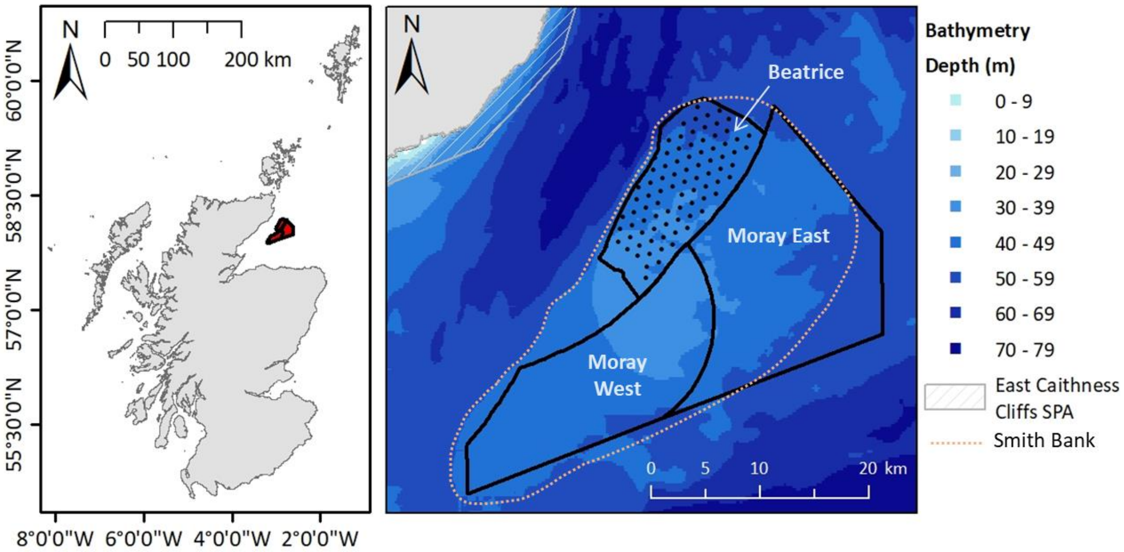

Figure 5.1 Study area located in the Smith Bank (dotted orange line) in the outer Moray Firth, NE Scotland. In June and August 2019, the Beatrice offshore windfarm was operational wile Moray East was under construction and Moray West not yet built. Along the coast, the East Caithness Cliffs Special Protection Area (dashed grey area) is home to several seabird breeding colonies.

Each day, the research vessel would leave its anchorage point at 05:00 (GMT), heading towards the start of a pre-determined fisheries acoustic transect. The morning was dedicated to the fisheries acoustic survey and pelagic fishing, as key prey species such as sandeels exhibit diel schooling behaviour (Freeman et al. 2004). During fisheries acoustic transects, visual boat-based surveys of seabirds were conducted to investigate whether the distribution of prey schools influenced seabird distribution. The afternoon was dedicated to hydrographic casts with bottom and upper mid-water seawater collection and zooplankton hauls. The averaged cruising speed was 8 knots.

Fisheries acoustic surveys

Fisheries acoustic transects were conducted between rows of turbines within the Beatrice offshore windfarm and a 4 km buffer area. Active acoustic data were collected during daylight hours (06:00 to 13:00 GMT) with a Simrad EK60 scientific multi-frequency echosounder and three split-beam transducers operating at 38, 120 and 200 kHz. The echosounder was set to generate one ping per second on all three transducers simultaneously, with a pulse duration of 1.024 msec. The transducers were mounted on the vessel's drop keel at a depth of 5.6 m. The 38 kHz transducer was calibrated with a 38.1 mm tungsten-carbide sphere before the start of the first survey, using standard procedures (Demer et al. 2015). The other two transducers were calibrated on 30th October and 1st November 2018. Sound pulses are transmitted downwards through the water column and the returning echoes are measured by the echosounder and arranged in echograms, which are depth-distance representations of any objects that scatter sound (Fernandes 2009). Each pixel on an echogram corresponds to a time-stamped, digitized-volume backscattering strength (VBS) (symbol: S_v; unit: dB re 1 m^2/m^3).

Pelagic trawling surveys could not be conducted between rows of turbines within the windfarm site. Instead, trawls were conducted at the south east boundary of the Beatrice offshore windfarm (Figure 5.5), in an area identified as important for sandeels during the developers' pre-consent surveys (Beatrice Offshore Windfarm Limited 2012b; Moray Offshore Renewables Limited 2012a). Pelagic fish were sampled using an International Young Gadoid Pelagic Trawl (PT 154) of 8 m vertical opening and 6 mm-mesh in the cod-end. For each haul, information on the catch composition (in numbers), time of start and end of the haul, latitude and longitude, and headline depth (*i.e.* depth of the upper frame of the net) were recorded. The trawling speed was approximately 2.7 knots (1.4 m. s^{-1}) and the mean trawl depth was 16.56 m (range: 11 - 19 m). Pelagic trawls were not directed at observed echotraces, but biological samples collected from hauls provided information on fish species present in the area during the surveys. Sub-samples of sandeels caught were measured to the nearest half cm, from the tip of the snout to the end of the tail, weighed, and the otoliths extracted to estimate age. Finally, subsamples were frozen at -20 °C for other studies on energetic content and toxin analyses.

Echotrace data analysis

Acoustic data were processed using Echoview® (Echoview Software Pty Ltd 2020), which facilitates the analyses of multifrequency echograms, synchronising the time and depth of each VBS from each echogram frequency (Mosteiro et al. 2004). First, standard post-processing steps were carried out, to remove VBS samples below the detected seabed (1 m offset bottom exclusion) and above the nearfield (8 m), to avoid surface transmit pulse. Any data collected outside the fisheries acoustic transects and hauls were classified as bad data region and were not included in the analyses.

Initially, I intended to use multi-frequency techniques to identify sandeel schools (Mosteiro et al. 2004) and discriminant analyses to distinguish between fish species based on their acoustic-frequency responses (Johnsen et al. 2009). However, due to the presence of mixed species schools and lack of ground truth data, I was unable to divide schools by species.

Instead, a two-stage algorithm was used to identify VBS of swimbladdered fish following the methodologies in Fernandes (2009) and Lawrence et al. (2016). First, the multifrequency thresholding algorithm summed VBS from all three frequencies (*Plus*) and applied a median (3x3) and a dilation (5x5) convolution filter to remove isolated scatterers and smooth the echograms, while preserving the shapes of the targets. A -180 dB threshold (*Data range bitmap*) was then applied, and the resulting virtual echogram was used to mask the original 120-kHz echogram. The remaining areas, representing the backscattered energy from strong scatterers, such as fish schools and dense aggregations of macro-zooplankton, were processed using Echoview's Schools Detection module. The detection parameters used were a minimum total school height of 1 m, a minimum candidate length of 5 m, a minimum candidate height of 1 m, a maximum vertical linking distance of 2 m, a maximum horizontal linking distance of 15 m and a total school length of 10 m (Fernandes 2009). The 120-kHz virtual echogram was resampled across 3 pings every metre, and a second school detection was conducted on this resampled echogram, as the resampling can affect the geometry of the school edges.

The second step of the algorithm compared the dB-difference between frequencies of the targets retained by the multifrequency thresholding algorithm, to keep only the fish-display geometric scatterers and remove the Rayleigh scatterers, which have a stronger frequency response on the 200-kHz echogram. The 38- and 200-kHz original echograms were resampled across 3 pings per metre and the resampled 200-kHz data were subtracted (*Minus*) from the resampled 38-kHz data to create a dB-difference echogram. The areas isolated by the school detection algorithm in the multifrequency thresholding stage (*Region bitmap*) were used to mask the dB-difference virtual echogram, and any areas with a mean dB-difference (*Region statistic*) below -2 dB were excluded. The remaining areas, representing fish-display geometric scattering, were again processed using the School Detection module algorithm with the same parameters as above. The regions detected as fish schools were subjected to a dilation (3x3) filter to ensure the edges of the schools were captured. The detected fish schools were then used to mask the 38-kHz echogram (from the multifrequency thresholding output) and produce an echogram of fish schools (Lawrence et al. 2016). This process was done separately for each cruise.

VBS data from the 38-kHz fish school echograms were then integrated per 100 m cell: (i) over the whole water column; (ii) over 5 m depth bins, to identify the minimum depth at which fish schools were detected and the depth bin with the highest fish density; and (iii) over 22 m depth bins, to compare fish schools' aggregation levels between three water column layers (8 – 30 m, 30 – 52 m, 52 – 74 m). Finally, measurements of prey availability were summarised and divided into 1,000 m long elementary sampling distance units (ESDU) and linked to each sampling point from the boat-based visual seabird surveys. The metrics used as proxies for prey availability were: (i) the presence/absence of prey; (ii) the prey density, represented by the log-transformed mean value of Nautical Area Scattering Coefficient (NASC) (*i.e. log (mean NASC + 1)*); (iii) the minimum depth of prey; (iv) the depth of maximum NASC (Depth $_{\text{max NASC}}$) and (v) the Index of Aggregation, which indicates the level of patchiness of schools, with higher values representing dense schools (Table 5.1).

Table 5.1 Variables used to describe the prey availability for seabirds in predator-prey models

Prey availability measurements	Hydro-acoustic metrics	Units	Resolution *(depth x length)*	Definition
Nautical Area Scattering Coefficient	NASC	$m^2.nmi^{-2}$	water column x 1 km EDSU	Proxy of fish density in the water column
School occurrence	Derived from NASC	# (1/0)	water column x 1 km EDSU	Presence/Absence of fish schools
Minimum depth of prey	Derived from NASC	m	5 m depth bin x 1 km EDSU	Depth at which the shallowest fish schools are encountered; proxy of fish accessibility
Depth of maximum NASC	Derived from NASC	m	5 m depth bin x 1 km EDSU	Depth with the highest density of fish
Index of Aggregation	IA	m^{-1}	water column x 1 km EDSU	Proxy of patchiness of fish school; IA is high when small areas are denser than the rest of the distribution (Urmy et al. 2012)

Hydrography

Hydrographic stations were conducted at each of the five fisheries acoustic transects, and the same sampling points were used in June and August to allow direct comparison between surveys (Figure 5.2). The physical properties of the water column (*i.e.* temperature and salinity) were measured with the Seabird SBE19plus CTD profiler (Seabird Electronics, Bellevue, Washington) in June (n = 9), and with the RBR concerto3 CTD profiler (RBR Ltd., Ottawa, Canada) in August (n = 11). For the June survey, the speed of sound for the whole water column was calculated following the equation of Coppens (1981) using the mean temperature and salinity of each upcast, and then averaged (mean) across all stations. Sound speed was provided by the RBR CTD used in August, thus the sound speed of the upcasts was directly averaged between all stations. The average sound velocity values per cruise were integrated into the Echoview calibration file for each survey. Furthermore, the standard deviation of temperatures ($\sigma(T)$) from each station cast was calculated to investigate variation in temperature throughout the water column between stations and surveys. Additionally, the pycnocline or density transition layer (defined as the depths at which the vertical density gradient is > 0.01 kg.m^{-4}) was identified. This was then used to distinguish between the mixed upper and lower water layers that should have a vertical density gradient ≤ 0.01 kg.m^{-4}. The mean temperatures of the upper and lower water layers were calculated and subtracted to estimate the thermal stratification (*ΔT*) index at each station (Scott et al. 2010; Embling et al. 2012).

Finally, at each sampling station, water samples were collected with two 1.5 L sampling bottles at specified depths (*e.g.* bottom, surface and/or upper mid-water layer). A Scanmar depth sensor (www.scanmar.no) was attached to the CTD frame to record approximate sampling depth, prevent the device from touching the bottom, and ensure that sample collections were made 5 m above the bottom. At each station, a subsample of water was transferred into a 200 mL glass bottle and stored in a temperature-controlled room until salinity was measured at Marine Scotland's Marine Laboratory to calibrate the CTD.

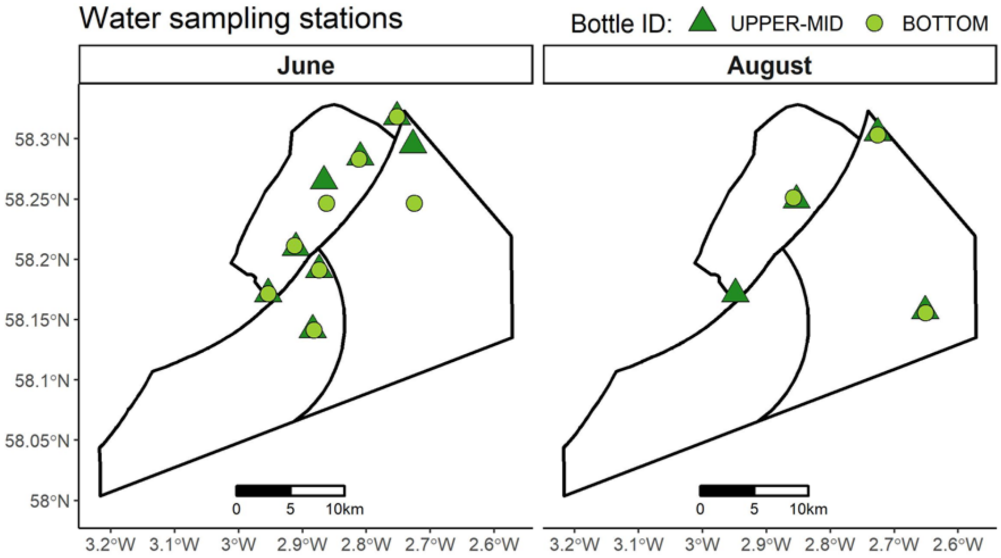

Figure 5.2 Upper-mid (light green circles) and bottom (green triangles) water sampling stations collected in June and August 2019 within and around the Beatrice offshore windfarm, NE Scotland.

The remainder of each water sample was then decanted in a 1 L polyethylene bottle and stored in a cool and dark environment prior to chlorophyll extraction. After gently agitating the sample bottle to resuspend large cells that may have settled, the volume of water was measured and then filtered (within two hours) using a vacuum pump and a Whatman GF/F filter paper (nominal porosity 0.7 µm, glass fibre). Using forceps, the filter paper with chlorophyll was carefully folded in half and placed in a labelled sample tube. As chlorophyll is light sensitive, the tube was covered with tin foil and frozen at -20°C. Chlorophyll-α analysis was conducted by Marine Scotland's Marine Laboratory, following the fluorometric methods of Arar and Collins (1997) and Smith et al. (2007), using a Turner AU Fluorometer (see section 2.5. in Bresnan et al. (2015)). Chlorophyll-α concentrations were then compared between depths, stations, and surveys to provide a proxy of phytoplankton primary production available to the lower trophic level of the food web (*e.g.* Capuzzo et al. (2018)).

Zooplankton collection and analysis

Two zooplankton hauls were conducted at each of the hydrographic stations (Figure 5.3). A Bongo net with two nets and cod-ends of mesh sizes 200 and 68 µm were deployed from the side of the research vessel. For the first haul, the net was thoroughly washed with water to push any planktonic material adhering to the net into the cod-ends. To estimate the volume of seawater that passed through the net during the ascent phase of the haul, a dedicated observer recorded the GPS coordinates and time when the net was first lowered, once it reached the bottom, once it started rising and when at the surface. The bottom and maximum sampling depths of each drop were also recorded, and the vertical net ascent and descent rates ($m.s^{-1}$) calculated using the horizontal drift of the boat, the ascent times and maximum sampling depth. The vertical ascent rates combined with the area of the 40 cm net aperture provides the volume of water sampled by each cod-end. Wet zooplankton samples were weighed within the cod-ends.

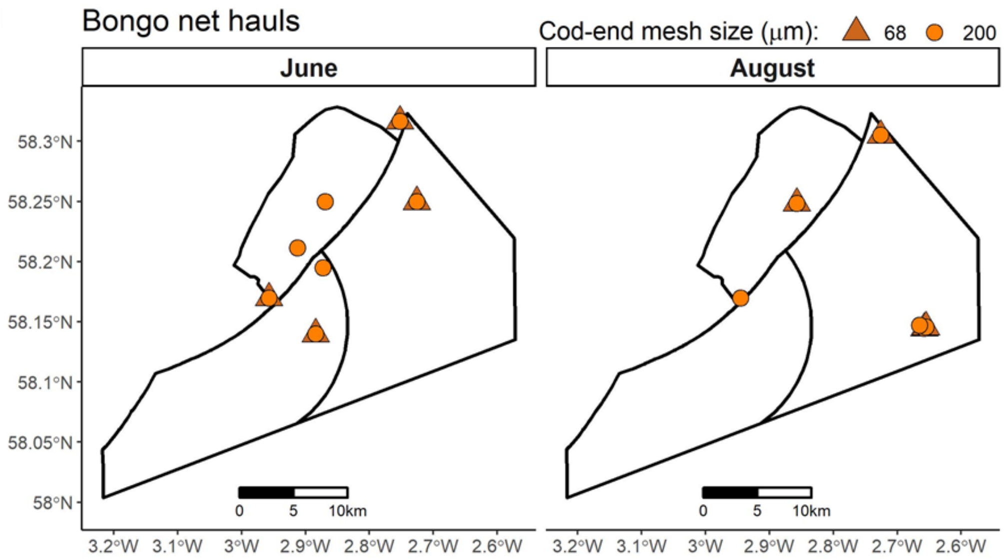

Figure 5.3 Bongo net haul locations for the zooplankton samples collected with cod-end of 68 µm (brown triangles) and 200 µm (orange circles) mesh sizes, within and around the Beatrice offshore windfarm, NE Scotland. The zooplankton from these haul locations were processed to estimate the relative abundance of zooplankton per station.

Three different approaches were used to preserve the zooplankton samples. First, for genetic identification (at the species group level), a 1 mL sample was collected from the cod end and preserved in 50 mL of 70% alcohol. Labelled samples were then preserved at –20°C. Second, for taxonomical identification, the remaining zooplankton from the cod-ends were preserved in 300 mL of 4% formaldehyde. Third, a second haul was carried out to estimate the dead/total ratio of zooplankton individuals at each location. To prevent the haul itself damaging or killing individual zooplankton, the net ascent was controlled to remain under 1 m.s^{-1}, as advised in Elliott and Tang (2009). Unlike the first haul, the net was not rinsed with water. The zooplankton sample was poured into a 500 mL jar containing filtered seawater and 0.75 mL of 1% neutral red stock solution. The jar was placed in a dark chamber for at least 15 min, during which the live individuals became stained. The sample was then filtered through a 50 µm nylon mesh disc, placed in a labelled Petri dish and preserved at -20°C. The sample was analysed within two months of collection. To brighten the neutral red stain, 1M HCl artificial seawater was first poured onto the frozen samples. Additional artificial seawater was then used to thaw and rinse the nylon mesh disc. Depending on the volume of zooplankton in the sample, the sample was diluted in either 200 or 500 mL of artificial seawater. With a pipette, subsamples of 5 mL were taken and put in a threecompartment Petri dish. Dead and alive individuals were identified and counted with two tally-counters in each subsample with a Zeiss Stemi SV6 microscope (1.2 magnification), using the dark field to increase contrast. The relative zooplankton abundance was then estimated for each haul. The zooplankton samples preserved in 4% formaldehyde and 70% alcohol were made available for other Aberdeen University projects.

Top predator visual boat-based surveys

During the fisheries acoustic survey transects, standardized seabird visual observations were recorded in 5- or 10-min intervals following strip-transect counts (Tasker et al. 1984), within 300 m of the vessel by a trained observer from the vessel's deck (5.6 m above sea level). Observations were conducted by naked eye scanning the 180° area ahead of the bow. Observation conditions (Beaufort sea state, cloud cover, swell, glare) were recorded at the start and the end of the survey and every 30 min. Although the weather and sea conditions varied during the two surveys, sea state was always ≤ 4 (Beaufort scale). For each 5- or 10-min interval, the time, species, number of individuals and behaviour were recorded. Seabird sightings were recorded either at the species or family level and categorised into functional and taxonomical groups with contrasting foraging strategies (as in Chapter 4): *Auk species*; *Large gull species* (LG); and, *Tern and small gull species* (TSG). Auks are pursuit divers, chasing prey by active swimming at depths ranging from 10 to 60 m but with occasional dives up to 180 m deep (Piatt and Nettleship 1985). Large gull species are surface feeders, seizing prey while swimming. In multi-species feeding associations, they are kleptoparasites, stealing prey from other birds, and scroungers, exploiting prey patches discovered by other birds (Camphuysen and Webb 1999). Tern and small gull species are surface feeders, mainly seizing prey while flying or shallow plunging. They are catalysts, which means that they usually initiate feeding flocks (Camphuysen and Webb 1999).

To investigate predator-prey distribution, occurrence, and relative abundance within and around the offshore windfarm site, several prey availability measures were summarised for the 500 m buffer around each seabird sampling point. Hereafter these 500 m radius cells are referred to as prey cells.

Habitat characteristics and environmental variables

To identify which environmental/physiographic parameters may influence schooling prey, information on physical and biological habitats was extracted from open-source datasets and summarised for each prey cell. The EMODNet bathymetry was downloaded (EMODNet Bathymetry 2020) as an ASC II file and read with R using the *raster* package (Hijmans 2020) (Figure S5.2A). Using the bathymetry raster file, the mean depth value of prey cell was calculated and allocated to each prey cell's centroid. The slope (Figure S5.2B) and aspect variables (in radians) were calculated based on the bathymetry value of eight neighbouring raster cells, using the function *terrain* from the *raster* package. The slope represents the steepness of a terrain, while the aspect shows the direction of the slope. The aspect was then converted in Northerness (North-South gradient; cos(aspect)) (Figure S5.2C) and Easterness (East-West gradient; sin(aspect)) (Figure S5.2D) to obtain a continuous gradient ranging from -1 to 1. Seabed sediment classification in offshore areas uses the Folk triangle classification (Folk 1954), based on the sand to mud ratio and gravel percentage. Seabed sediment type was obtained from Digimap (1:250,000 scale Offshore Geology Data) as a polygon shapefile (Figure S5.2E). Each prey cell centroid was classified based on the predominant sediment within that prey cell. The sediment variable was then used to assess the suitability of habitat within the prey cell for sandeels (which prefer high proportions of medium and coarse sand (0.25 mm ≤ particle size < 2 mm) with a low silt content (Holland et al. 2005)), and for whiting (*Merlangius merlangus*) (which prefer sandy sediments (Atkinson et al. 2004)).

The European Union Nature Information System (EUNIS) habitat classification, characterised using various parameters such as the depth zone and substrate type, was downloaded as a shapefile (EMODNet Seabed Habitats 2021) (Figure S5.2F). As for sediment, the proportion of habitat type per prey cell was calculated and the predominant habitat type considered as the main habitat type in that prey cell.

Table 5.2 Static environmental variables used to investigate fish schools' habitat preferences in the vicinity of the Beatrice offshore windfarm site.

Variables	Units	Spatial resolution
Depth	m	1/8 arc minutes (~ 230 m)
Slope	radian	1 km
Aspect	radian	1 km
• ***Northerness: cos(aspect)***	# [-1;1]	
• ***Easterness: sin(aspect)***	# [-1;1]	
Sediment type	# (%)	1:250,000
Habitat type	# (%)	100 x 100 m

Modelling

Four Linear Models (LM) were used to compare primary and secondary production between surveys. Response variables were i) thermal stratification (ΔT) derived from CTD casts, ii) chlorophyl α concentrations, iii) the number of > 200-µm zooplankton individuals per haul and iv) the number of > 68-µm zooplankton individuals per haul. The categorical variable *survey ID* (two-levels: June and August) was used as an explanatory variable in all four models but was included as an interaction with the sampling depth in the *chlorophyl model* based on the assumption that if the water column was thermally stratified, chlorophyl α concentrations would vary with sampling depth as well as with season.

To investigate whether the fish school occurrence, vertical distribution and morphology varied between surveys, three LMs were conducted. The first two response variables used were the fish density per EDSU and the mean school depth, and the explanatory variable was the *survey ID*. For the third model, the response variable was the index of aggregation (computed per 1 km cell and 22 m depth bin), which was square-root-transformed to meet normality criteria. The two explanatory variables were the depth layer (8 – 30 m, 30 – 52 m, 52 – 74 m), and the *survey ID*. To compare the index of aggregation between depth layers, pairwise comparisons were conducted using the *emmeans* package (Lenth 2020).

To investigate whether fish school occurrence was related to specific habitat characteristics, the presence/absence of fish schools per EDSU was modelled as a binomial response variable in a Generalized Linear Mixed Model (GLMM) using the *lme4*

R package (Bates et al. 2015). The *cloglog* link function was chosen based on the lowest Akaike Information Criterion (AIC). The categorical covariate *survey ID* was used as a random effect in the model. A categorical covariate indicating location inside/outside the windfarm site and various static environmental variables (see Table 5.2) that may influence prey occurrence were initially integrated into a full model. I then used the *drop1* function which compares all possible models by dropping single covariates (R Core Team 2019) that contribute least to the model until the model with the lowest AIC is identified. Similarly, to investigate whether fish density changed with habitat characteristics, the dataset was filtered to focus on fish-presence only data and the response variable *log (NASC + 1)* (a proxy for fish density) was used in a Linear Mixed Model (LMM). The best fitted model was selected using the same approach as above. The covariates *survey ID* and *hour* of the day were used as random effects to remove any potential confounding effects of survey time on fish density, especially because certain fish species may display diel and seasonal vertical movements through the water column (*e.g.* Freeman et al. (2004)). Finally, to improve model convergence, the bound optimization by quadratic approximation (BOBYQA) method was used with a set maximum of 200,000 iterations.

To study predator-prey interactions, measures of prey availability were related to the relative abundance of auk species, or the occurrence of terns and small gulls (TSG) and large gulls (LG). The measures of prey availability used were initially prey density, the Index of Aggregation, the minimum depth of prey and the depth of maximum fish density (Table 5.1). However, the two latter metrics were colinear and only depth of maximum fish density was included in the final models. Additionally, sea state, distance both from coast and from the nearest offshore structure were included as explanatory variables. This was to account for increased detections in calmer sea conditions and close to breeding colonies (Waggitt et al. 2018), or due to the potential attraction to offshore structures (see Chapter 4). The relative abundance of auk species, expressed as the log-transformed number (N) of auk sightings (*i.e.* $log\ (N_{Auk} + 1)$), was used as the response variable in an LMM, with *survey ID* as a random effect. The presence/absence of terns and small gulls, and of large gulls were modelled as binomial response variables in Generalized Linear Models (GLM) using a *logit* and *cloglog* link function, respectively.

As above, the same model selection approach was used to identify the best fitted models for each of the seabird species groups.

F-tests were calculated for LM. For GL(M)M and LMM, Wald Chi-square tests were calculated using the *car* package (Fox and Weisberg 2019) to assess the significance of fixed effects and their interactions (*cf. chlorophyl model*). For model validation, the uniformity, dispersion and temporal autocorrelation of the model residuals were verified using the *DHARMa* package (Hartig 2020). Finally, for multivariate models, the response variable was predicted, and the uncertainty (95% confidence intervals, CI) estimated using the *make_predictions* function of the *jtools* package (Long 2019).

RESULTS

Variation in water column physical characteristics, and in primary and secondary production

CTD casts were conducted at 17 sites during the two cruises (June n = 9, August n = 8) (Figure 5.4). In June, no transition layer was identified at five of the nine stations, with water column temperature standard deviations at these sites ranging between 0.03 and 0.27°C (Figure 5.4A). The thermal stratification index was estimated for the remaining 12 stations. Overall, the water column was more stratified (+0.37°C) in August than in June ($F_{1,11} = 5.04$, $p < 0.05$) (Figure 5.4B). Additionally, chlorophyll α concentrations varied significantly with depth and month ($F_{1,22} = 5.01$, $p = 0.037$), with higher chlorophyll α concentrations in June than in August. In June, chlorophyll α concentrations ranged between 1.7 and 3.2 µg/L throughout the water column, while in August, lower levels of chlorophyll α were recorded in the lower mid-water layer (< 30 m) than in the upper mid-water layer (Figure 5.4C). Conversely, the estimated number of zooplankton collected in the water column was significantly higher in August than in June (around 2.5 times for 68 µm mesh-size cod-end $F_{1,8} = 9.237$, $p < 0.02$ around 9.7 times for 200 µm mesh-size cod-end $F_{1,12} = 21.34$, $p < 0.001$) (Figure 5.4D).

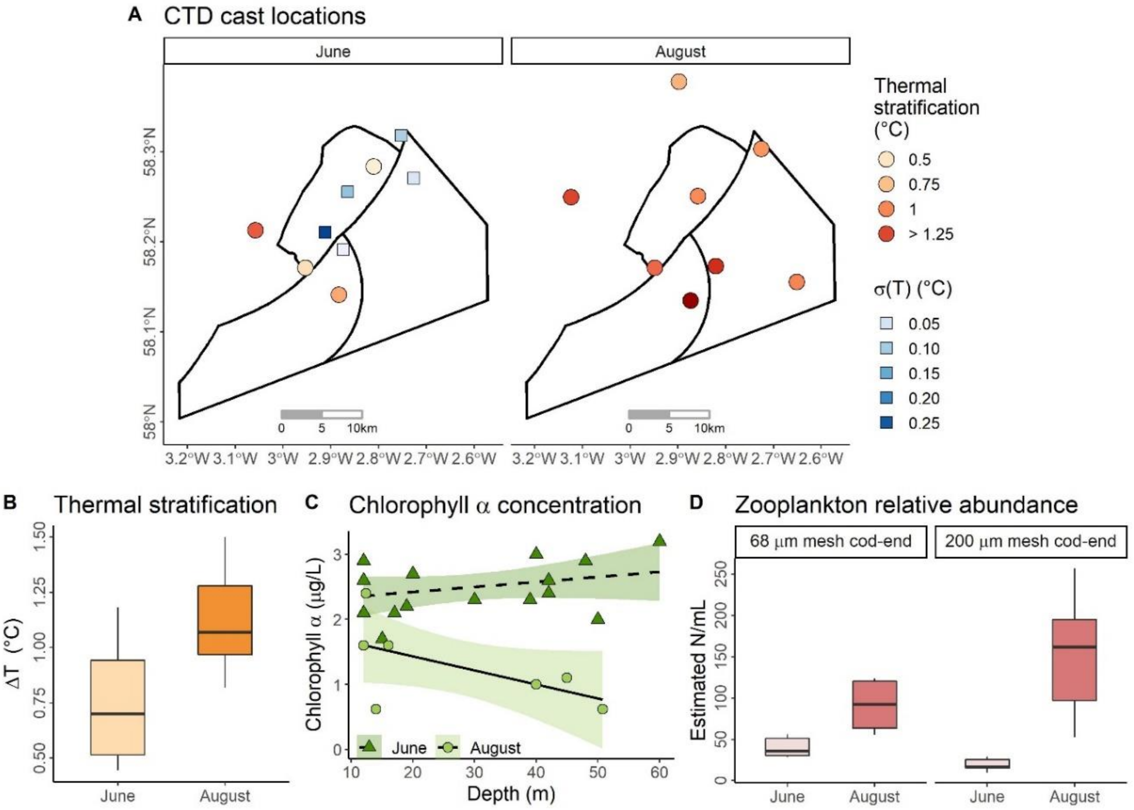

Figure 5.4 A) Thermal stratification (yellow-red gradient circles) and water column temperature standard deviation (σ(T)) (blue gradient squares) per CTD cast in June and August 2019; B) Thermal stratification (ΔT) boxplots per month; C) Chlorophyll α concentration in relation to water depth in June and August 2019 within and around the Beatrice offshore windfarm; D) Relative abundance of zooplankton per cod-end mesh-size and month within and around the Beatrice offshore windfarm.

Summary of fishing hauls

A total of six trawl hauls (four in June and two in August) were taken. In June, one was composed exclusively of lesser sandeels (*Ammodytes marinus*), three were a mixture of sandeels and other species (*e.g.* juvenile Norway pout *Trisopterus esmarkii*). In August, the two hauls were composed of small numbers of juvenile whiting (*Merlangius merlangus*) associated with unidentified jellyfish (Figure 5.5, Figure S5.1 and Table S5.1).

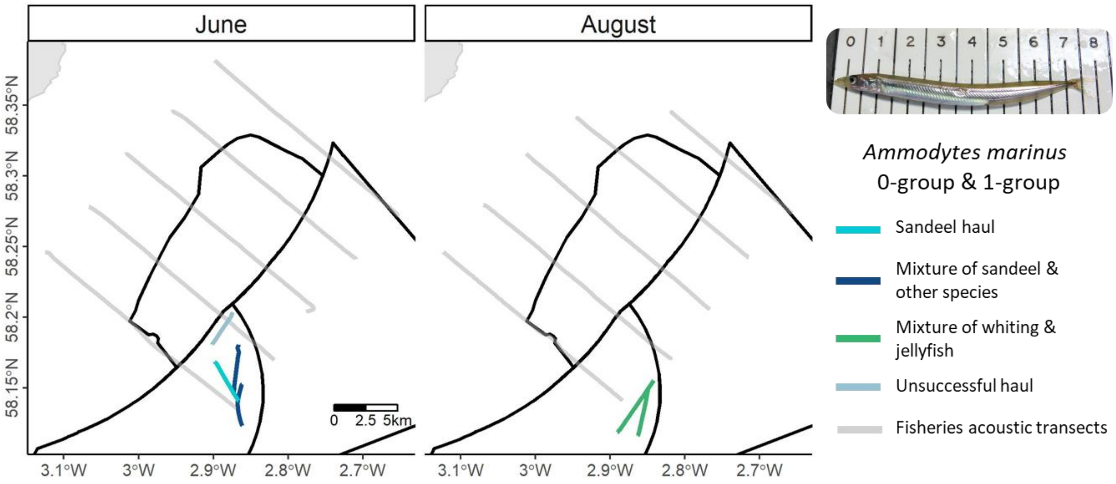

Figure 5.5 Fishing haul locations (coloured lines) and fisheries acoustic transects (grey lines) in June and August 2019 within and around the Beatrice offshore windfarm, NE Scotland.

Variation in schooling prey occurrence, density, and vertical distribution between surveys

Fisheries acoustic survey transects conducted in June and August 2019 each covered a total effort length of around 100 km. Spatial variation in fish school occurrence and density was observed in both surveys, with higher fish density on the western side of the Beatrice offshore windfarm (Figure 5.6A). Fish density was also significantly higher in August than in June ($F_{1,105} = 11.895$, $p < 0.001$) (Figure 5.6C). However, fish schools were present in shallower waters in June (median school depth of 20.6 m) than in August (median school depth of 52.8 m) ($F_{1,398} = 102.34$, $p < 0.001$) (Figure 5.6B). Further, the aggregation level of fish schools significantly changed both between surveys (Wald test: $\chi^2 = 35.236$, df = 1, $p < 0.0001$) (Figure 5.6D), and with water depth (Wald test: $\chi^2 = 21.518$, df = 2, $p < 0.0001$) (Figure 5.6E). Fish school aggregation was higher in June than in August ($z = 4.602$, $p < 0.0001$), and when fish schools were at shallower depths (*8 – 30 m vs 52 – 74 m*: $z = 3.403$, $p < 0.002$; *30 – 52 m vs 52 – 74 m*: $z = 3.43$, $p < 0.002$). Finally, variation in the vertical distribution of scatterers from plankton was observed between in and outside the windfarm, along the transects 3 and 4 in August, (see echograms in Figure S5.3). These patterns were not observed along the transects outside nor at the edge of the windfarm.

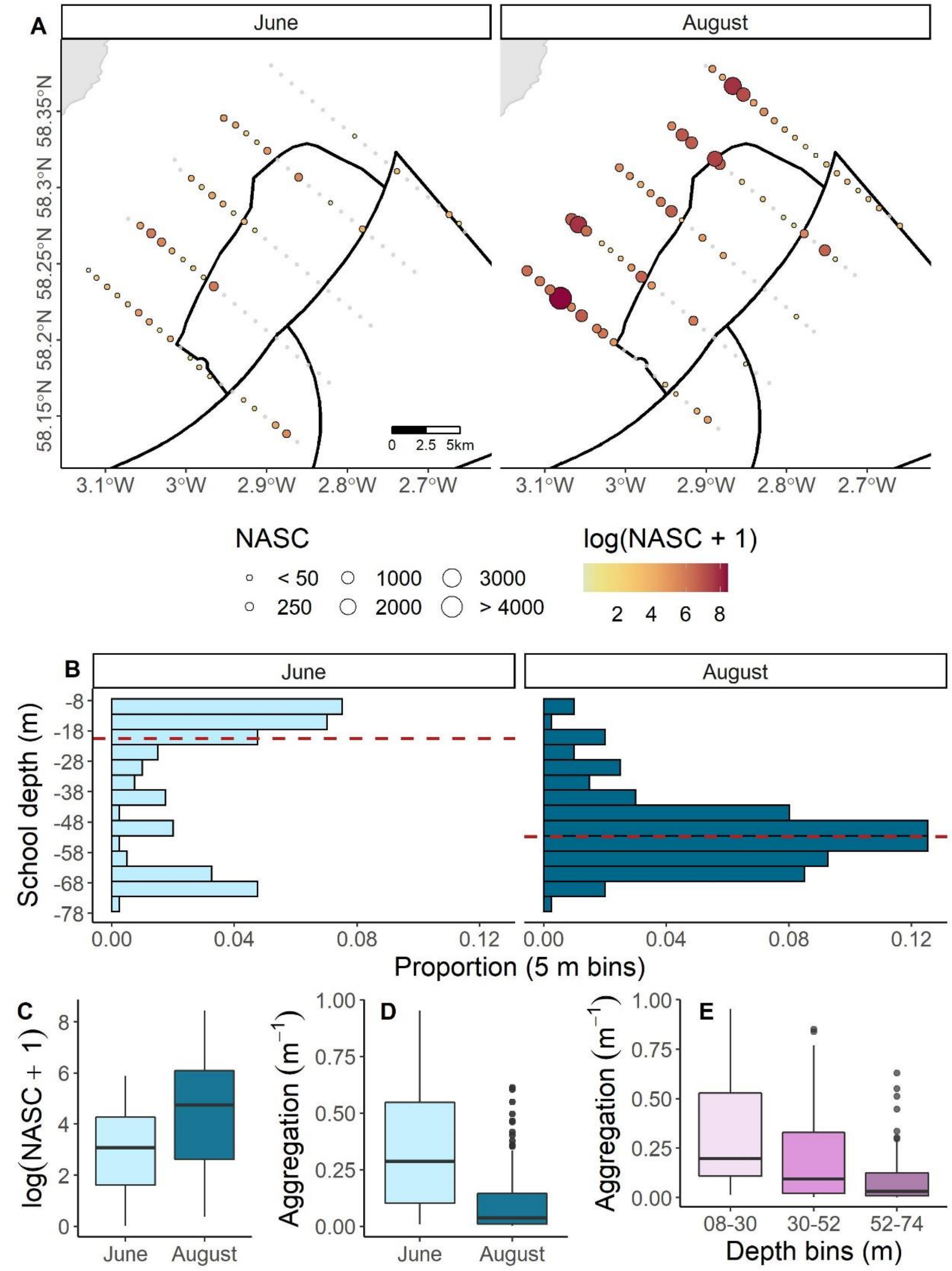

Figure 5.6 A) Spatial distribution of schooling prey density (Nautical area scattering coefficient (NASC) in m^2*.nmi*$^{-2}$*) along the five fisheries acoustic survey transects conducted in June and August 2019 within and around the Beatrice offshore windfarm; B) Vertical distribution of fish schools per 5-m depth bins between June and August 2019; red dotted line represents the median depth of schools per survey; C) Comparison of fish density in the water column between surveys; Comparison of fish school aggregation (high level of aggregation highlight fish school patchiness) D) between surveys and E) between water column depth bins using pooled data from both June and August(i.e. 8 – 30 m, 30 – 52 m and 52 – 74 m).*

The covariates that best fitted the GLMM to explain school occurrence were bathymetry (*i.e.* log-transformed depth), and location with respect to the windfarm (*i.e.* in- *vs* outside windfarm). The habitat type was kept in the best fitted model (with the lowest AIC) but was not significant. The probability of fish school occurrence increased with depth (Wald test: $\chi^2 = 23.265$, df = 1, $p < 0.0001$) (Figure 5.7A) and was higher outside the windfarm site (Wald test: $\chi^2 = 5.017$, df = 1, $p < 0.03$) (Figure 5.7B). When fish schools were present, fish density (*i.e.* log (NASC + 1)) was also significantly higher in deeper waters (Wald test: $\chi^2 = 15.256$, df = 1, $p < 0.0001$) (Figure 5.7C).

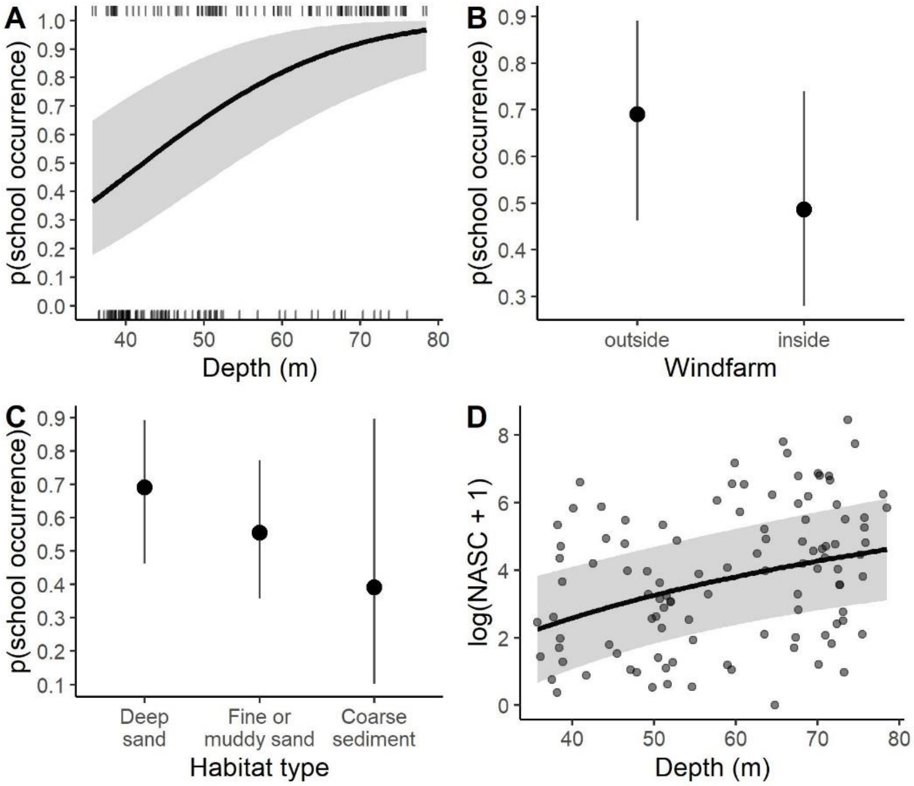

Figure 5.7 Probability of fish school occurrence in relation to A) the bathymetry, B) the windfarm, C) the habitat type (non-significant); D) Estimated fish density in relation to the bathymetry.

Spatial distribution and abundance of seabird sightings during visual surveys

During both cruises, *Auk species* were the most frequently observed species group with 41.7 % of sightings. The two other species groups included in subsequent analyses were the *Large gull species* (LG) and *Tern and small gull species* (TSG), which represented respectively 7.9% and 7.5% of the sightings (Figure 5.8).

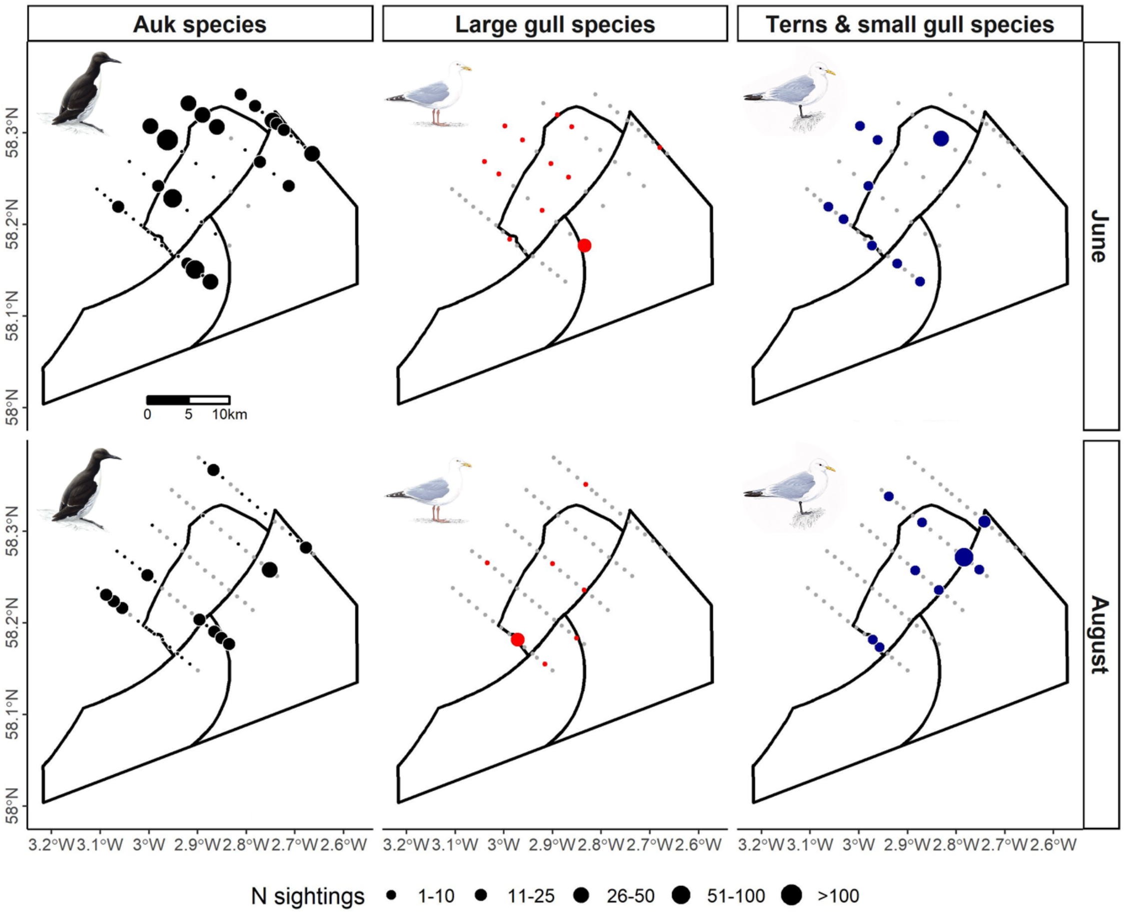

Figure 5.8 Numbers of sightings for three seabird species groups (Auk species black circles, Large gull species red circles and Terns and small gull species blue circles) recorded during the fisheries acoustic transects in June and August 2019, within and around the Beatrice offshore windfarm, NE Scotland.

Predator - prey interaction in the vicinity of an operational windfarm site

To understand whether seabird distribution was influenced by prey availability in the vicinity of an operational windfarm, I investigated the occurrence/abundance of three seabird species groups in relation to measurements of prey availably (Figure 5.9).

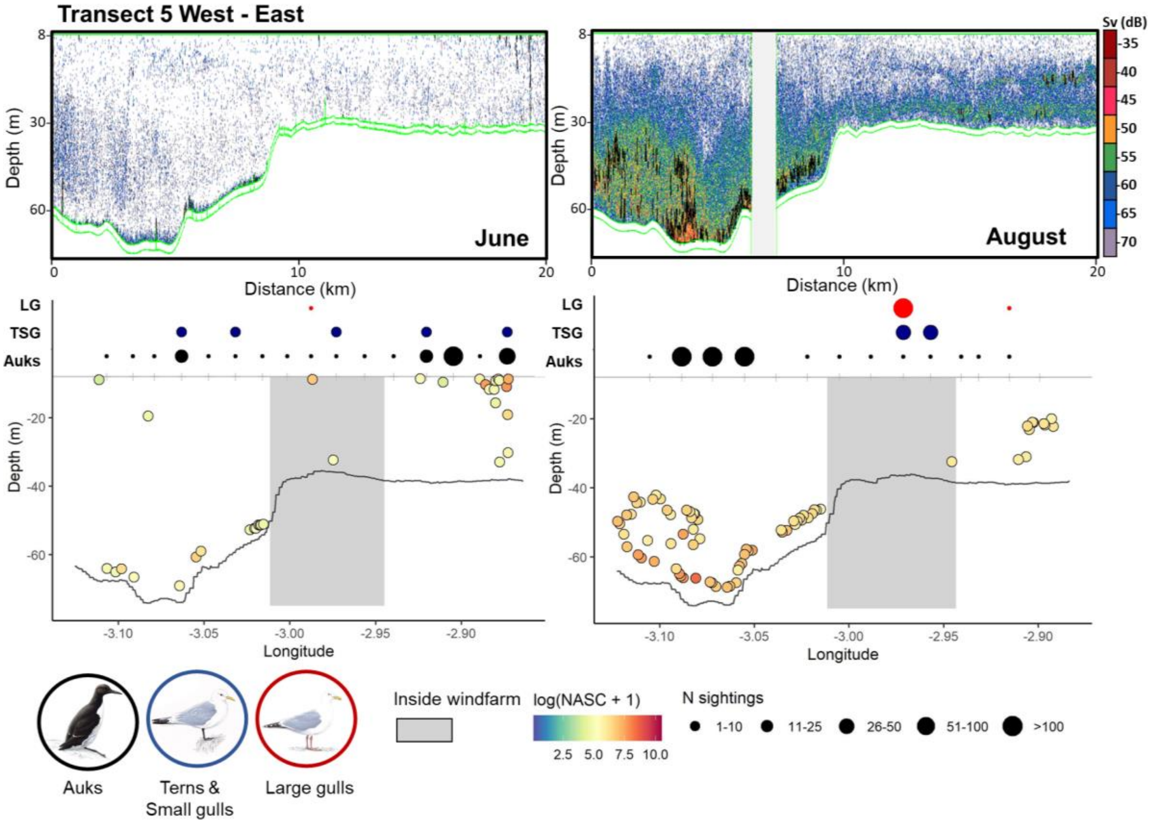

Figure 5.9 Example of the fish school and seabird distribution and abundance/density along, the transect 5 located at the southern boundary of the Beatrice offshore windfarm, NE Scotland, and recorded in June and August 2019. The top panels are subsamples of the 38 kHz echograms in which colours are the mean volume-backscattering strength (S_v) from fish, plankton, and others. The lower panels provide information on the fish school distribution along the transect and the water column; variation in fish density per schools is highlighted with the rainbow colour palette. Acoustic data were processed and analysed between the grey line at 8 m depth and 1 m above the black line which represents the bathymetry. The grey ticks are the counting sampling points, and the circles of variable size are the number of sightings per seabird species group. The grey box represents the windfarm site. See Figure S5.3, Figure S5.4 and Figure S5.5 for the other transects.

When fish schools were present, the abundance of auks increased when fish density was the highest in shallower water depths (Wald test: $\chi^2 = 12.479$, df = 1, $p < 0.001$) (Figure 5.10A). However, auk abundance decreased close to offshore structures (Wald test: $\chi^2 = 11.551$, df = 1, $p < 0.001$) (Figure 5.10B) and their detectability decreased with the Beaufort Sea state (Wald test: $\chi^2 = 11.751$, df = 3, $p < 0.01$) (Figure 5.10C).

When fish schools were present, the probability of terns and small gull species' occurrence increased with the index of aggregation, suggesting their presence overlapped with patchier fish schools (Wald test: $\chi^2 = 4.736$, df = 1, $p = 0.03$) (Figure

5.10D), and when fish density was the highest in shallower water depths, although the latter was not significant (Figure 5.10E).

Similarly, in the presence of fish schools, the probability of large gull species' occurrence increased when fish density was the highest in shallower water depths (Wald test: $\chi^2 = 8.18$, df = 1, p = 0.004) (Figure 5.10F) and near the coast, although the latter was not significant.

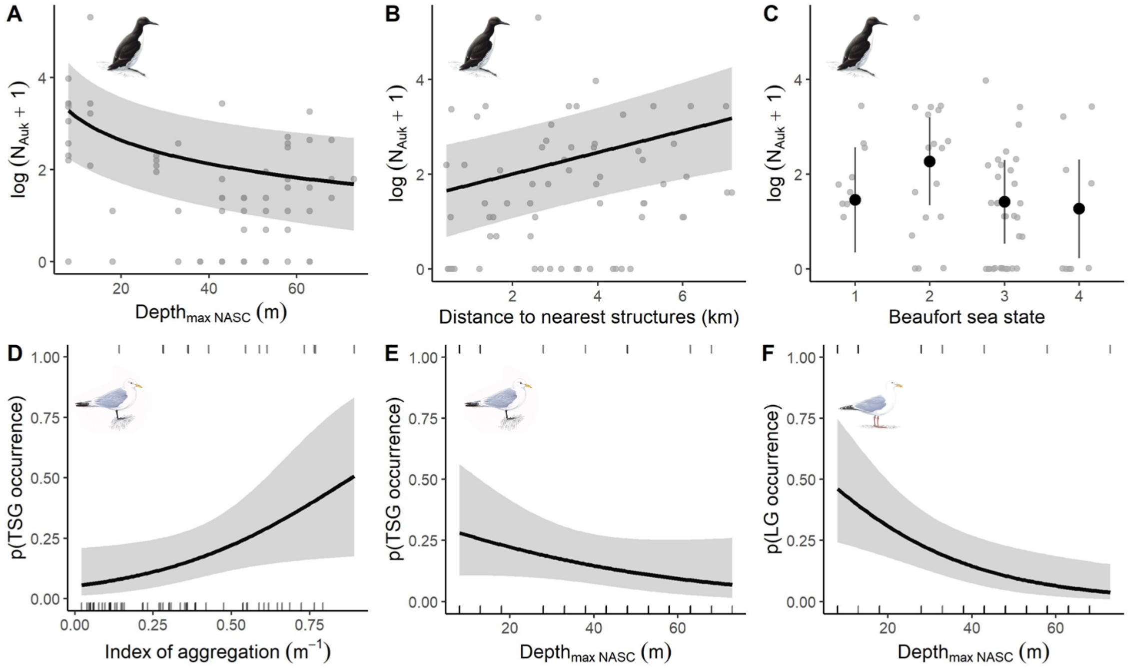

Figure 5.10 Estimated abundance of auk species in relation to A) the depth of maximum fish density, B) the distance to nearest structures, and C) the sea state. Probability of terns and small gull species' (TSG) occurrence in relation to D) the index of aggregation, and E) the depth of maximum fish density (non-significant). F) Probability of large gull species' (LG) occurrence in relation to the depth of maximum fish density. 95% confidence intervals (shaded areas or line range for C) estimated for uncertainty in fixed effects only; grey points (A-C) or ticks (D-F) represent the raw data distribution.

DISCUSSION

Previous studies around operational offshore windfarms have tended to focus on single species (*e.g.* Vallejo et al. (2017)) or functional groups (*e.g.* Bergström et al. (2013); Krägefsky (2014)) rather than investigating the spatial association between top predators and their prey at these development sites. Using a multidisciplinary approach, this study provided insights into the links between predator and prey distributions and abundance, and characterised seasonal changes in the water column structure, and the primary and secondary production within and around the Beatrice offshore windfarm between June and August 2019.

Seasonal primary and secondary production, and ambient hydrography

In June, low thermal variation in the water column (Figure 5.4A) and higher concentrations of chlorophyll α that were distributed homogeneously in the water column (Figure 5.4C) suggest that the water was well mixed. This vertical mixing may enhance nutrient transport and increase the primary production. In August, waters were more stratified following the formation of the seasonal thermocline (Figure 5.4B). Chlorophyll α concentrations were, overall, lower in August, and lowest in the bottom mixed layer, under the pycnocline (Figure 5.4C) (Fernand et al. 2013). The thermocline can act as a barrier to nutrient exchange and it is usually along this transition gradient that the light and nutrients favour phytoplankton growth (Fernand et al. 2013). Although the primary production in this study area was lower in the bottom mixed layer in August, the fisheries acoustic data indicated that there was higher secondary production under the stratified upper layer (*i.e.* in the pycnocline and bottom mixed layers) (see echograms in Figure S5.3). Overall, zooplankton relative abundance was significantly higher in August (Figure 5.4D), which coincides with the seasonal zooplankton peak in the Northern North Sea (Bresnan et al. 2015). Thus, production shifted towards higher trophic levels throughout the summer, with sequential biomass peaks in phytoplankton and zooplankton (Mustamäki et al. 2016). Finally, although I did not directly assess the potential effect of offshore windfarm structures, I did observe some variation in the vertical distribution of zooplankton in relation to the windfarm on echograms, especially along the transects 3 and 4 (Figure S5.3). Within the windfarm site, the zooplankton community seemed located mainly at the bottom, which may be due to changes in the water column structure through enhanced vertical mixing (Carpenter et al. 2016; Floeter et al. 2017). Further work is required to assess the extent to which these offshore structures may induce mixing potentially affecting water stratification, hydrodynamics, and nutrient transport.

Forage fish spatial and seasonal distribution

This study highlighted seasonal and spatial variation in pelagic forage fish distribution and density within and around the Beatrice windfarm. Fish school density was higher in deeper waters outside than within the windfarm site on the Smith Bank (Figure 5.6A) and increased in August (Figure 5.6C). Additionally, the observed difference in fish school vertical distribution between the two surveys was likely due to seasonal changes in the degree of mixing (Figure 5.6B). In mixed waters, in June, fish schools were mainly present in the upper mid water layer (< 20 m depth) or were demersal (> 60 m depth), while in stratified waters, in August, most of the forage fish schools were within the mixed bottom layer (> 40 m depth) (Figure 5.6B). Fish school patchiness was overall higher in June (Figure 5.6D) and at shallower depths (up to 52 m) (Figure 5.6E). These results suggest that fish school structure and behaviour likely changed seasonally and with their position in the water column. These patterns in fish school aggregation level and distribution may be linked to anti-predator strategy (Partridge 1982; Magurran 1990; Urmy and Benoit-Bird 2021), local prey patch distribution (*e.g.* Van Der Kooij et al. (2008); Fauchald (2009); (Benoit-Bird et al. 2013)) and was likely species-specific (van Hal et al. 2017).

Further, although the acoustic processing used did not identify fish schools to the species-level, species compositions are likely to have changed seasonally. When fishing activities were located within the same area during both cruises (outside the south-east boundary of Beatrice offshore windfarm, see Figure 5.5), haul compositions differed dramatically between June and August (Table S5.1). The main pelagic fish species caught, during the survey conducted in June, was the lesser sandeel (*Ammodytes marinus)*, which is a key prey item of the diet of many piscivorous fish (Greenstreet 1998), seabirds (Daunt et al. 2002; Wilson et al. 2004) and marine mammals (Santos and Pierce 2003; Wilson and Hammond 2019). All sandeels caught belong to the 0- and 1-group, age classes known to be taken by seabirds in this region (Lewis et al. 2001; Lewis et al. 2003; Wilson et al. 2004; Daunt et al. 2008). By contrast, in August, pelagic trawls caught only unidentified jellyfish and juvenile whiting (*Merlangius* merlangus). Commensal association between 0-group gadoid fish and medusae has been previously reported in the North Sea where medusae can act as a shelter and enhance whiting feeding opportunities (preying upon zooplankton and the hyperiid parasites of the jellyfish) and survival (Lynam and Brierley 2007). The fact that no sandeels were caught in late August suggests that sandeels may have already entered the overwintering stage during which they remain buried in the seabed (Winslade 1974).

Overall, trawl sampling to characterise pelagic fish schools was challenging, primarily because fishing hauls could not be conducted, along the fisheries acoustic transects within the windfarm. In addition, the trawling depth was mostly < 20 m, and the headline depth of the most successful sandeel haul was around 11 m. In future, pelagic sandeel school sampling could be enhanced by trawling early in the morning, either closer to the surface (around 10 m) or by targeting specific echotraces. A further drawback of pelagic trawls is that they may only catch individuals of a specific age class (*e.g.* 0-group and 1-group) and catches may not represent the whole population nor the structure of schools observed acoustically (Van Der Kooij et al. 2008). Where information on sandeel age structure is required, night-time grab surveys may provide more robust data as all sandeels are buried in the sediment. Due to their high degree of site fidelity, sandeels' foraging behaviour resembles that of a central-place forager (Van Deurs et al. 2013), and their night time benthic distribution is a good proxy for their pelagic distribution in the day (Van Der Kooij et al. 2008). Similarly, diel patterns of sandeels vary seasonally but also with age-class. In summer, sandeels display vertical movements to feed on zooplankton in the upper-mid water layer from dawn (Johnsen et al. 2009). However, adult sandeels, growing less, are likely to reach satiation sooner than juveniles and may rebury in the sediment earlier in the day. Similarly, adult sandeels attain higher body condition sooner, and thus start overwintering earlier in the year (Winslade 1974). Consequently, sampling only the water column may bias both the fishing haul composition and fisheries acoustic data towards juvenile sandeels. Finally, due to the lack of ground-truthing of echotraces and the acoustic characteristics of sandeels (*i.e.* low backscattering strength), identifying sandeel school echotraces, with confidence, proved to be difficult, an issue raised previously in other studies (*e.g.* Hassel et al. (2004); Mosteiro et al. (2004); Mackinson et al. (2005)). In the future, a combination of acoustics, pelagic fishing and night-time dredge sampling may help to provide complementary information on sandeel biomass at different times of the day and year (Mackinson et al. 2005).

Overall, the probability of fish school occurrence (Figure 5.7A) increased with depth and was higher outside the windfarm site (Figure 5.7B) but did not significantly change with habitat type (Figure 5.7C). Similarly, fish density increased with depth (Figure 5.7D). As seen in Figure 5.6A, this suggests that fish school occurrence and density was higher beside and along the edge of the sandbank in deep circalittoral sandy waters (Figure S5.2F). The lower fish density observed on the sandbank may be related to fish species' habitat preferences, fish diel behaviour and distribution and/or to the presence of the windfarm. First, although the habitat type did not seem to significantly influence fish distribution and density in this study, fish species and diversity likely differ between the sandbank and its surrounding habitat. For instance, sand habitat is the preferred sediment habitat type of whiting (Atkinson et al. 2004) and sandeels (Holland et al. 2005), the two main pelagic fish caught during this survey. The three other most abundant species caught were juvenile Norway pout (*Trisopterus esmarkii*), common dab (*Limanda limanda*) and plaice (*Pleuronectes platessa*), common species of the northern North Sea and previously recorded in the Moray Firth (Greenstreet 1998). The Norway pout, a small short-lived gadoid species, generally prefers muddy, sandy sediment habitat (Kutti et al. 2015), and lives pelagically in the first year of life (0-group) feeding on zooplankton (e.g. copepods), before becoming demersal (Robb and Hislop 1980). The two demersal flatfish species (common dab and plaice) are known to occur in the vicinity of the Smith Bank and seem to both prefer coarser sediments (Greenstreet 1997). Second, pelagic schooling fish may be attracted by topographically contrasting habitat, such as the sandbank edges, where the local oceanographic conditions over the bank edges can aggregate food resources and enhance surface prey aggregation (Stevick et al. 2008; Embling et al. 2013). For instance, in summer, sandeels exhibit daily movement from their night-time burrowing sites largely located on the top of shallow sandbanks (at preferred depth range of 30 – 50 m), to deeper waters and along the edge of sandbanks to feed on plankton (Van Der Kooij et al. 2008). Third, the scarce and patchy distribution of small fish schools in June, at the shallower depths, may have made fish schools less detectable during the fisheries acoustic transects over the Smith Bank.

Finally, in the absence of baseline fisheries acoustic data before the windfarm construction, I cannot conclude whether the variation in fish density between in- and outside the windfarm is linked to the naturally contrasting habitat or related to the presence of the windfarm.

Predator-prey spatial association

To have a better understanding of seabird distribution within and around the windfarm, during the breeding season, I characterised the prey field in different ways, to account for prey suitability, which encompasses prey availability, detectability and accessibility to predators that have different foraging strategies and energetic requirements (Benoit-Bird et al. 2013). First, the most commonly used metric quantifying fish density per ESDU (*i.e.* log (NASC +1)) did not appear to significantly explain seabird distribution, as this metric was not retained in the final models. This metric is potentially not a good measure of prey availability as it presumably fails to capture any information on the accessibility of prey to different predators. Similarly, another study investigating the relative importance of biomass and patchiness at different trophic levels found that the spatial variability in biomass, at small-scales, was a better predictor than the standing biomass of organisms to explain bottom-up regulation of a pelagic community (Benoit-Bird and McManus 2012). In my study, the depth with the highest prey density (Depth $_{\text{max NASC}}$) and the index of aggregation were the two prey measures that best explained auk abundance and the occurrence of large gulls and terns and small gulls. Increased prey density in the upper mid-water layer (depth range of 8-20 m) was positively related to an increase in auk abundance and large gull species' occurrence. Using a hull-mounted downward facing multibeam echosounder, I was able to quantify pelagic fish density and distribution across a large area but missed information on the first eight metres below the surface, which may be relatively important for surface-feeders. Although no prey field data were recorded in the upper metres of the water column, the positive association between auk, large gull species and prey density in the upper mid water layer may reflect surface and subsurface prey fields (Benoit-Bird et al. 2013). In the future, a bottom-mounted upward-facing echosounder could be deployed concurrently to validate this assumption, but would also provide fine-scale information on the vertical and diel distribution of the targeted prey species (*e.g.* Axenrot et al. (2004); Urmy and Benoit-Bird (2021)).

Despite different foraging strategies between auks and large gull species, variation in vertical prey distribution played an important role in shaping their distribution and abundance at-sea. Both species groups were positively associated with shallower dense fish schools. Feeding on prey patches near the surface may optimise foraging profitability (Pyke 1984; Zwarts and Wanink 1993; Chimienti et al. 2017) or be linked to the fish size targeted by auks during the breeding season (Wilson et al. 2004), and is consistent with large gulls' feeding methods (Camphuysen and Webb 1999).

Similarly, the probability of terns and small gull species' occurrence was positively related to higher levels of aggregation, which means this species group was mainly present in areas with patchy, small but denser fish aggregations (Urmy et al. 2012). These dense but patchy fish schools were more present in the upper mid-water layer than at deeper depths and in June, in mixed waters, than in August (Figure 5.6D). When forage fish vertically migrate through the water column during the day, to feed at layers with enhanced resource availability, they shoal to dilute predation risk (Fauchald 2009). This dynamic adaptive shoaling behaviour creates an elusive and unpredictable prey field for predators. However once pelagic fish schools are detected either directly or by other foraging predators (*i.e.* local enhancement), catalyst seabirds usually aggregate in flocks or multi-species assemblages which may increase their feeding success through facilitation between predators (Thiebault et al. 2016). For instance, 61% of encounters with terns and small gull species were observed in association with auk species (Figure S5.7, Figure S5.8), which, on some occasions, coincided with the occurrence of near-surface small fish schools (Figure S5.4, Figure S5.5). The commensal association between kittiwakes and auks is common in the Moray Firth and the presence of synchronised diving auks is thought to be of high importance for initiating kittiwake foraging flocks (Camphuysen and Webb 1999) (Figure S5.8). Similarly, large gull species such as herring and great black-backed gulls are also known to join these multi-species feeding associations as kleptoparasites and/or scroungers (Camphuysen and Webb 1999) (Figure S5.7). In this study, during 79% of encounters with large gull species, auks were present, which on some occasions have been linked to the presence of near-surface fish schools, especially in June (see Figure S5.4). Thus, although overall fish density was lower and fish schools patchier and potentially more elusive in June, their natural and/or auk-driven near-surface distribution may increase prey availability for aerial, surface-feeding predators that can only utilize a small proportion of the vertical habitat of their prey (Fauchald 2009).

Improving our understanding of the processes shaping predator and prey distribution and abundance within and around development sites is crucial to be able to assess, more accurately, the potential ecological effects of offshore windfarm developments. This study provided insights into those ecosystem processes and found evidence that patchiness of prey is an important factor influencing top predator distribution in pelagic marine systems. Further work is now required to investigate the effect of offshore structures on fine-scale associations between predators and prey.

APPENDIX

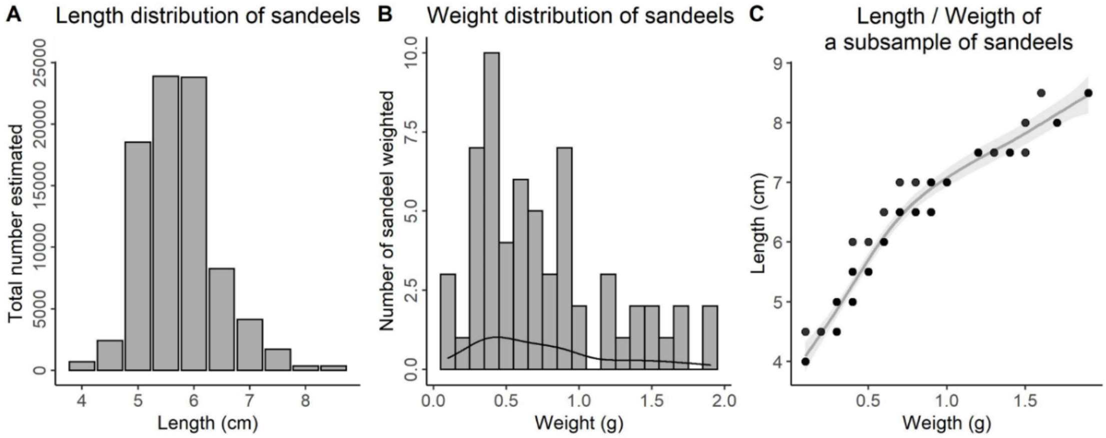

Figure S5.1 A) Length, B) Weight distributions of three subsamples of sandeels (Ammodytes marinus) caught in June 2019, in the Smith bank, NE Scotland. C) Relationship between the length and weight of a subsample of the sandeels caught.

Table S5.1 Haul summary for the Alba na Mara cruises 0919A and 1419A

Date	Haul number	Species	Length (cm)	Total per length	Subsample weight	Subsample N	Total weight	Estimated N	Total N	Raising Factor
08/06/2019	A19.136	Norway pout	3.5	3	92.7	87	2058	67	**1931**	**22.2**
08/06/2019	A19.136	Norway pout	4	6				133		
08/06/2019	A19.136	Norway pout	4.5	7				155		
08/06/2019	A19.136	Norway pout	5	34				755		
08/06/2019	A19.136	Norway pout	5.5	29				644		
08/06/2019	A19.136	Norway pout	6	8				178		
08/06/2019	A19.136	Common dab	11	1					**61**	
08/06/2019	A19.136	Common dab	12	2						
08/06/2019	A19.136	Common dab	13	6						
08/06/2019	A19.136	Common dab	14	13						
08/06/2019	A19.136	Common dab	15	18						
08/06/2019	A19.136	Common dab	16	6						
08/06/2019	A19.136	Common dab	17	12						
08/06/2019	A19.136	Common dab	18	2						
08/06/2019	A19.136	Common dab	19	1						
08/06/2019	A19.136	Plaice	14	1					**25**	
08/06/2019	A19.136	Plaice	16	3						
08/06/2019	A19.136	Plaice	17	2						
08/06/2019	A19.136	Plaice	18	7						
08/06/2019	A19.136	Plaice	19	2						
08/06/2019	A19.136	Plaice	20	1						
08/06/2019	A19.136	Plaice	21	3						
08/06/2019	A19.136	Plaice	22	2						
08/06/2019	A19.136	Plaice	23	2						
08/06/2019	A19.136	Plaice	24	1						
08/06/2019	A19.136	Plaice	26	1						
08/06/2019	A19.136	A. marinus	4.5	5					**64**	
08/06/2019	A19.136	A. marinus	5	1						
08/06/2019	A19.136	A. marinus	5.5	13						
08/06/2019	A19.136	A. marinus	6	17						
08/06/2019	A19.136	A. marinus	6.5	6						
08/06/2019	A19.136	A. marinus	7	20						
08/06/2019	A19.136	A. marinus	7.5	13						
08/06/2019	A19.136	A. marinus	8	7						
08/06/2019	A19.136	A. marinus	8.5	2						
09/06/2019	A19.137	A. marinus	4.5	1	65.5	120	422.8	6	**775**	**6.5**

Date	Haul number	Species	Length (cm)	Total per length	Subsample weight	Subsample N	Total weight	Estimated N	Total N	Raising Factor
09/06/2019	A19.137	A. marinus	5	17				110		
09/06/2019	A19.137	A. marinus	5.5	51				329		
09/06/2019	A19.137	A. marinus	6	38				245		
09/06/2019	A19.137	A. marinus	6.5	10				65		
09/06/2019	A19.137	A. marinus	7	3				19		
11/06/2019	A19.139	A. marinus	4	2	153.8	244	52500	683	**83290**	**341.4**
11/06/2019	A19.139	A. marinus	4.5	7				2389		
11/06/2019	A19.139	A. marinus	5	54				18433		
11/06/2019	A19.139	A. marinus	5.5	69				23553		
11/06/2019	A19.139	A. marinus	6	69				23553		
11/06/2019	A19.139	A. marinus	6.5	24				8192		
11/06/2019	A19.139	A. marinus	7	12				4096		
11/06/2019	A19.139	A. marinus	7.5	5				1707		
11/06/2019	A19.139	A. marinus	8	1				341		
11/06/2019	A19.139	A. marinus	8.5	1				341		
27/08/2019	A19.260	Whiting	3	1					**46**	
27/08/2019	A19.260	Whiting	4	12						
27/08/2019	A19.260	Whiting	5	5						
27/08/2019	A19.260	Whiting	6	3						
27/08/2019	A19.260	Whiting	7	5						
27/08/2019	A19.260	Whiting	8	13						
27/08/2019	A19.260	Whiting	9	5						
27/08/2019	A19.260	Whiting	10	1						
27/08/2019	A19.260	Whiting	11	1						
27/08/2019	A19.260	Lumpsucker	3	1					**1**	
29/08/2019	A19.261	Whiting	3	1					**28**	
29/08/2019	A19.261	Whiting	4	3						
29/08/2019	A19.261	Whiting	5	5						
29/08/2019	A19.261	Whiting	7	7						
29/08/2019	A19.261	Whiting	8	10						
29/08/2019	A19.261	Whiting	9	1						
29/08/2019	A19.261	Whiting	10	1						
29/08/2019	A19.261	Snake pipefish	6	1					**1**	

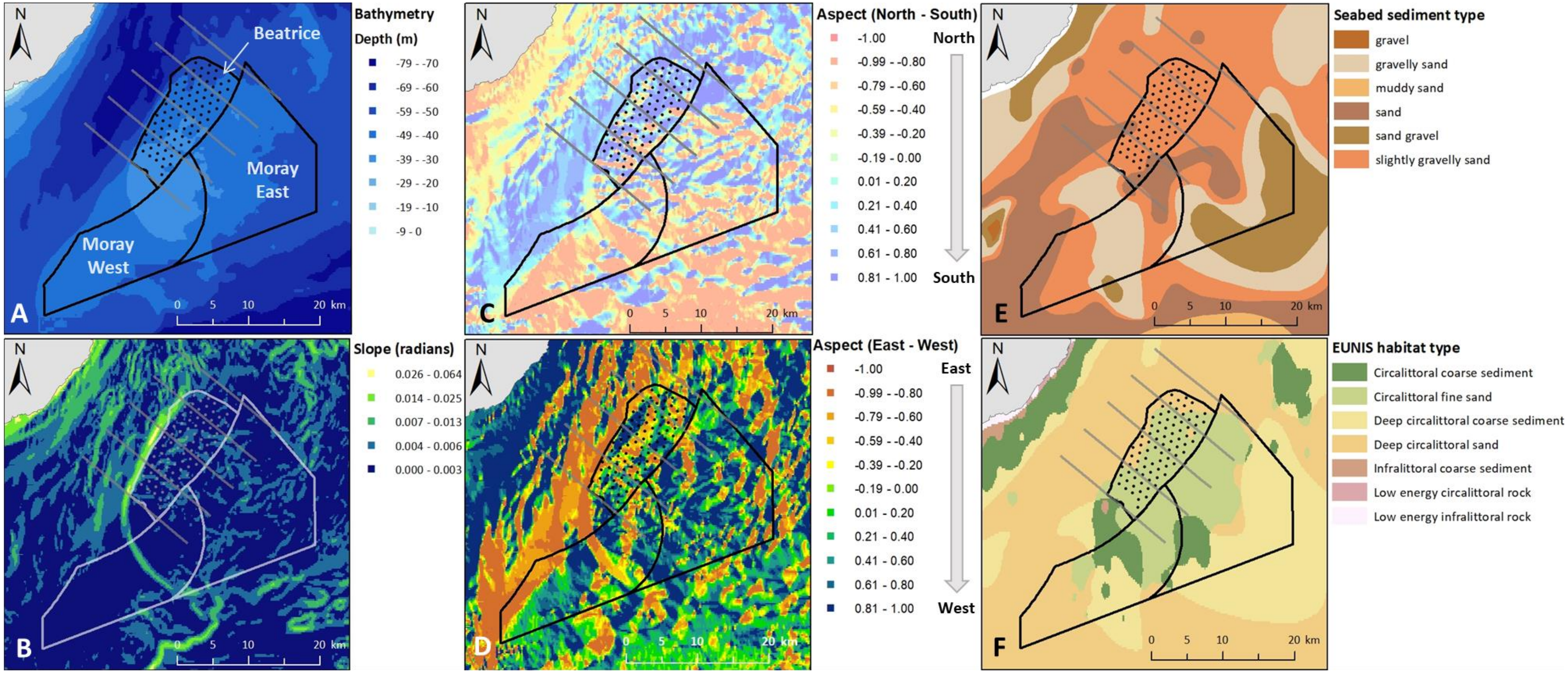

Figure S5.2 Seafloor maps of the Smith Bank (NE Scotland) where the Beatrice, Moray East and Moray West offshore windfarms are/will be commissioned. The grey lines represent the fisheries acoustic survey transects conducted in June and August 2019. A) Bathymetry from EMODnet; B) Slope derived from the bathymetry; C) Aspect quantified as northerness; D) Aspect quantified as easterness; E) Seabed sediment type from Digimap; F) EUNIS habitat type.

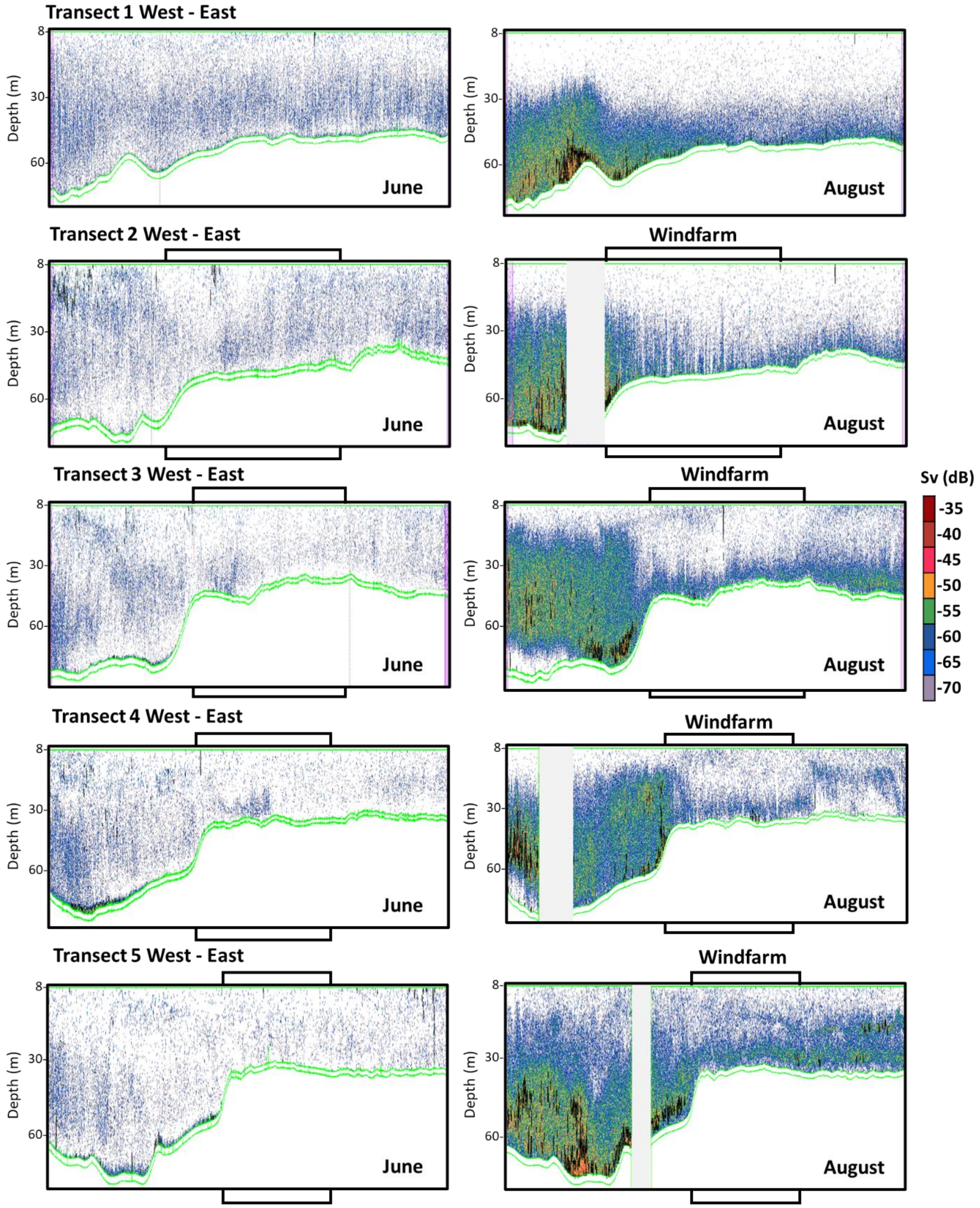

Figure S5.3 38 kHz echograms of each fisheries acoustic transect conducted in June and August 2019 within and around the Beatrice offshore windfarm, NE Scotland. Green lines are the sea bottom (lowest line), 1m above the bottom and 8 m below the surface (upper line). The colour panel highlights the mean volume-backscattering strength (S_v) from fish, plankton, and others. The grey rectangles indicate sections with no acoustic data recorded. The back rectangles in the background give an indication of the windfarm boundary.

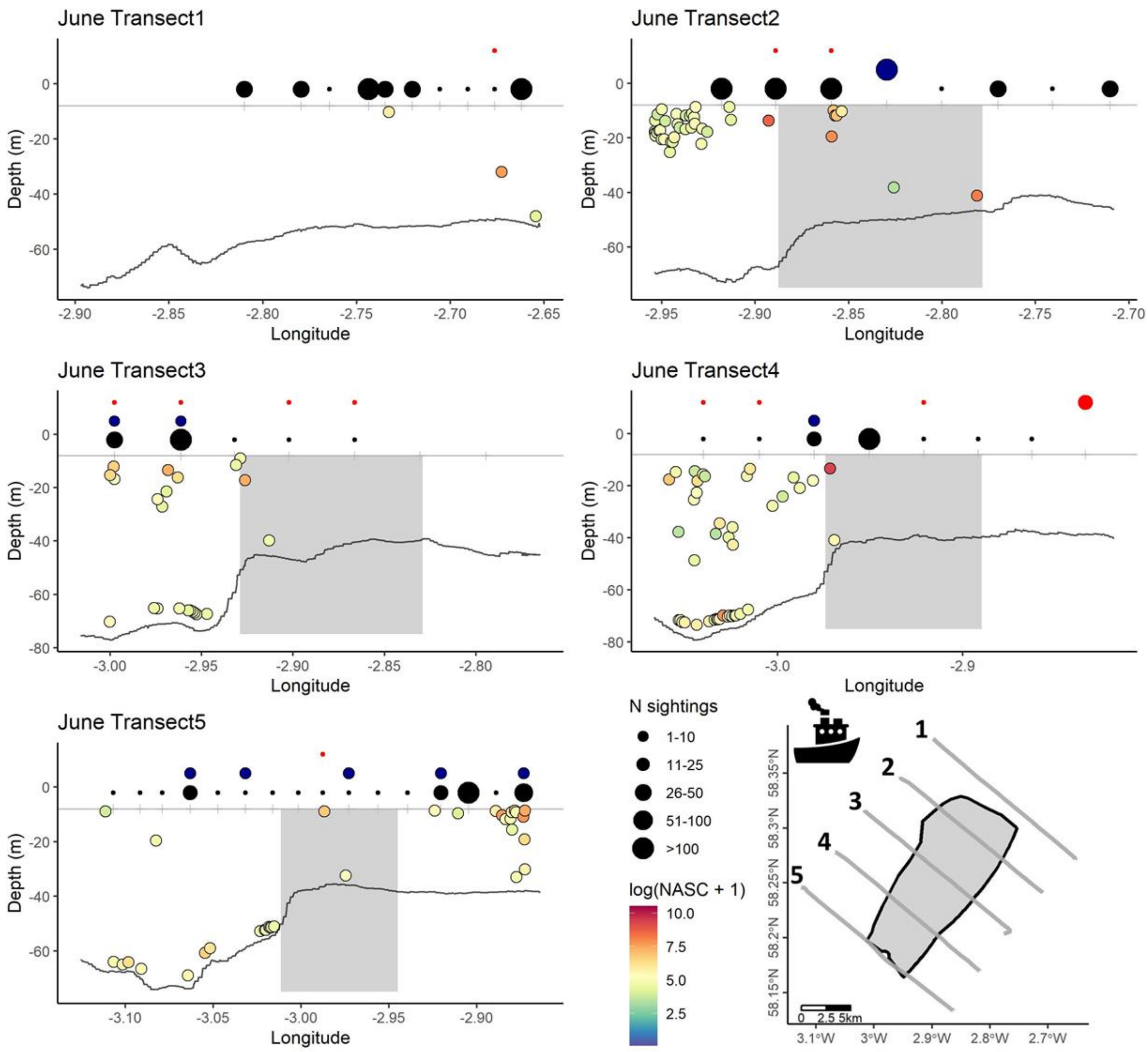

Figure S5.4 Summary of seabird and fish school distribution and abundance along the fisheries acoustic transects and in the water column in June 2019, within and around the Beatrice offshore windfarm, NE Scotland. The variation in fish density per schools (i.e. log (NASC + 1); where NASC stands for Nautical Area Scattering Coefficient) is highlighted with the rainbow colour palette. Acoustic data were processed and analysed between the grey line at 8 m depth and 1 m above the black line which represents the bathymetry. The grey ticks are the counting sampling points, and the circles of variable size are the number of sightings per seabird species groups (auks in black, large gull species in red, and terns and small gull species in blue). The grey box represents the windfarm site.

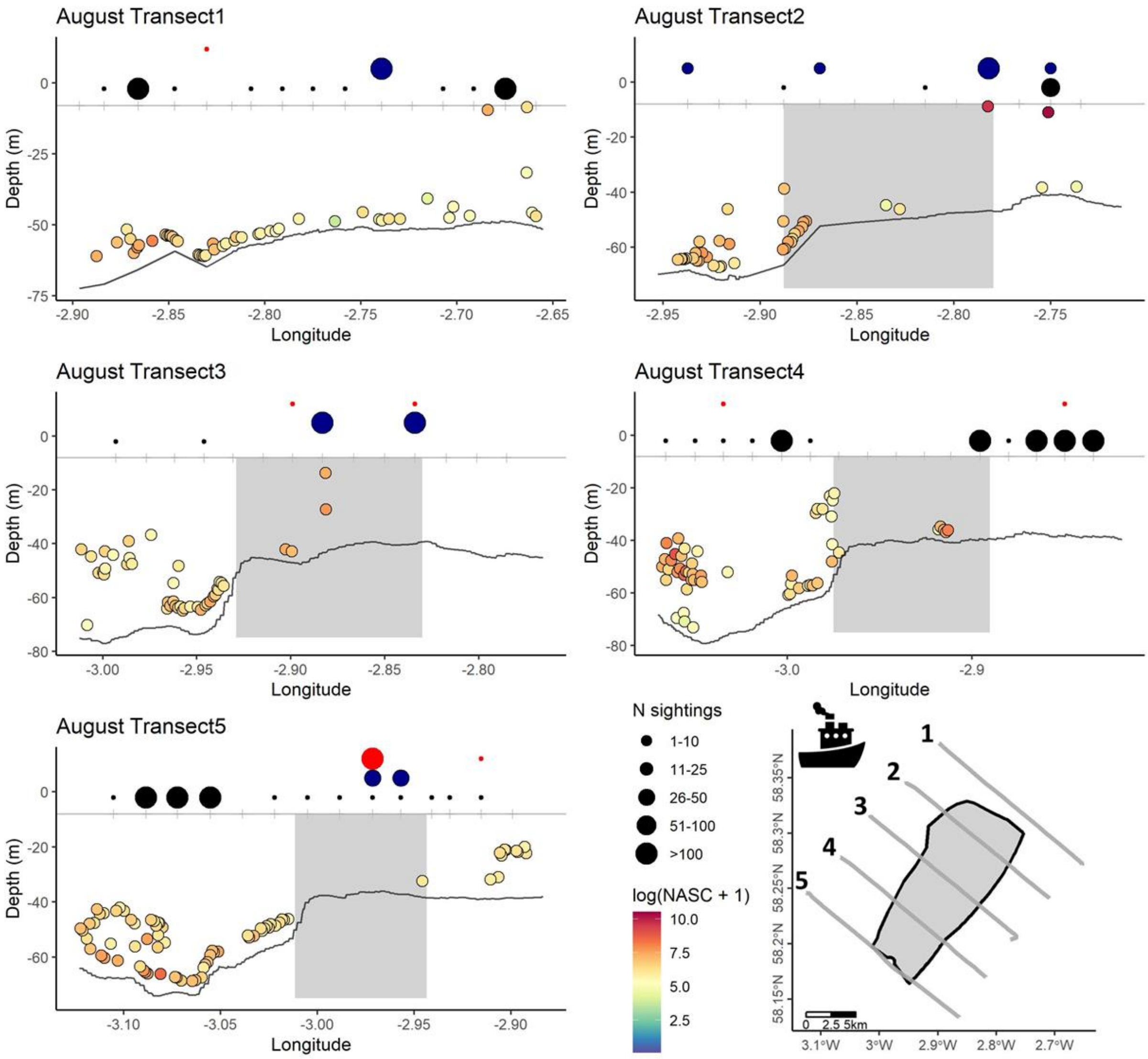

Figure S5.5 Summary of seabird and fish school distribution and abundance along the fisheries acoustic transects and in the water column in August 2019, within and around the Beatrice offshore windfarm, NE Scotland. The variation in fish density per schools (i.e. log (NASC + 1); where NASC stands for Nautical Area Scattering Coefficient) is highlighted with the rainbow colour palette. Acoustic data were processed and analysed between the grey line at 8 m depth and 1 m above the black line which represents the bathymetry. The grey ticks are the counting sampling points, and the circles of variable size are the number of sightings per seabird species groups (auks in black, large gull species in red, and terns and small gull species in blue). The grey box represents the windfarm site.

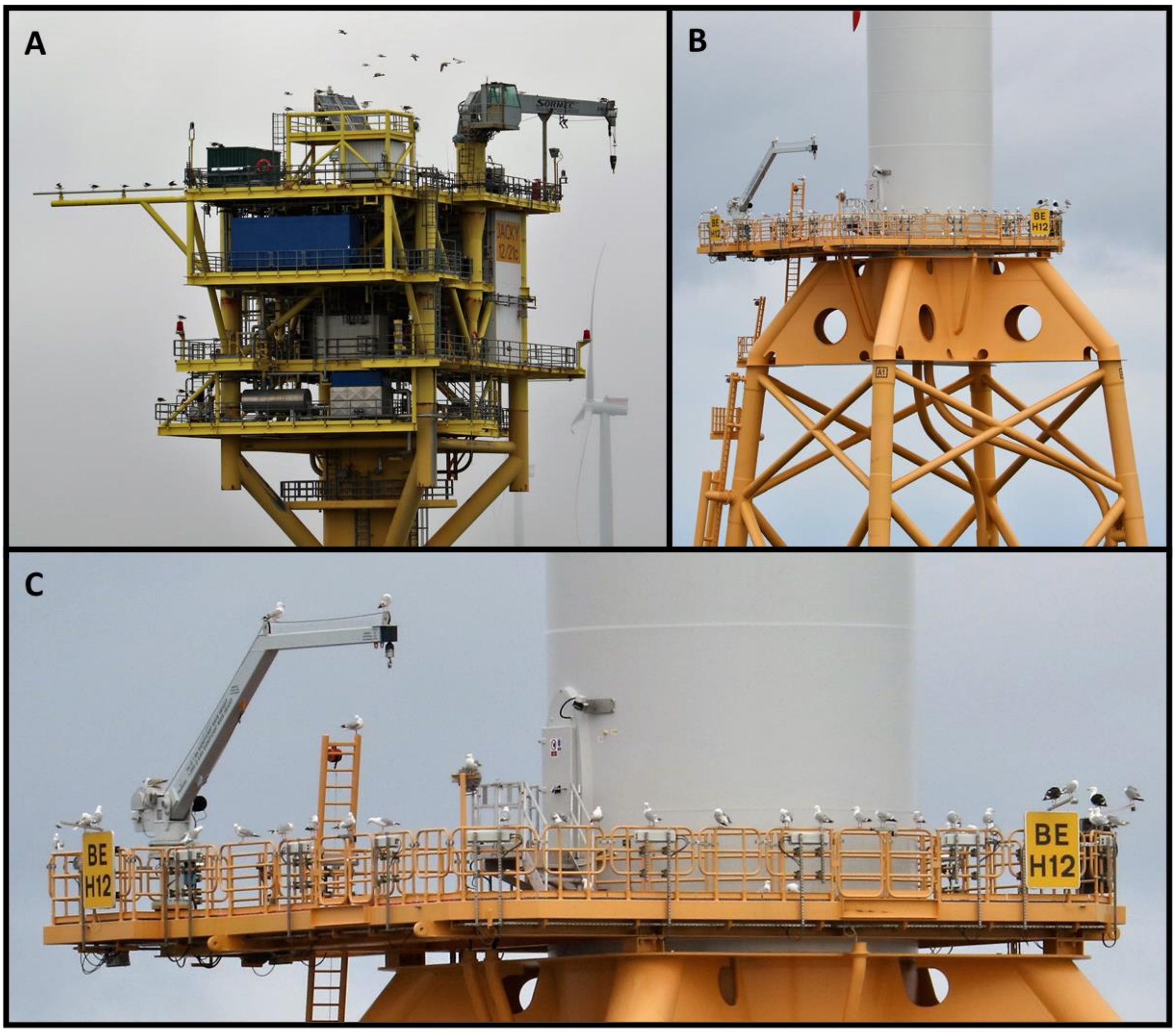

Figure S 5.6 Examples of seabird, mainly large gull species, roosting on (A) the Jacky platform, (B) one of the wind turbine jacket foundations at the Beatrice offshore windfarm, NE Scotland; (C) zoom in of (B) the main large gull species observed roosting are herring gulls and Great black-backed gull. Source: University of Aberdeen.

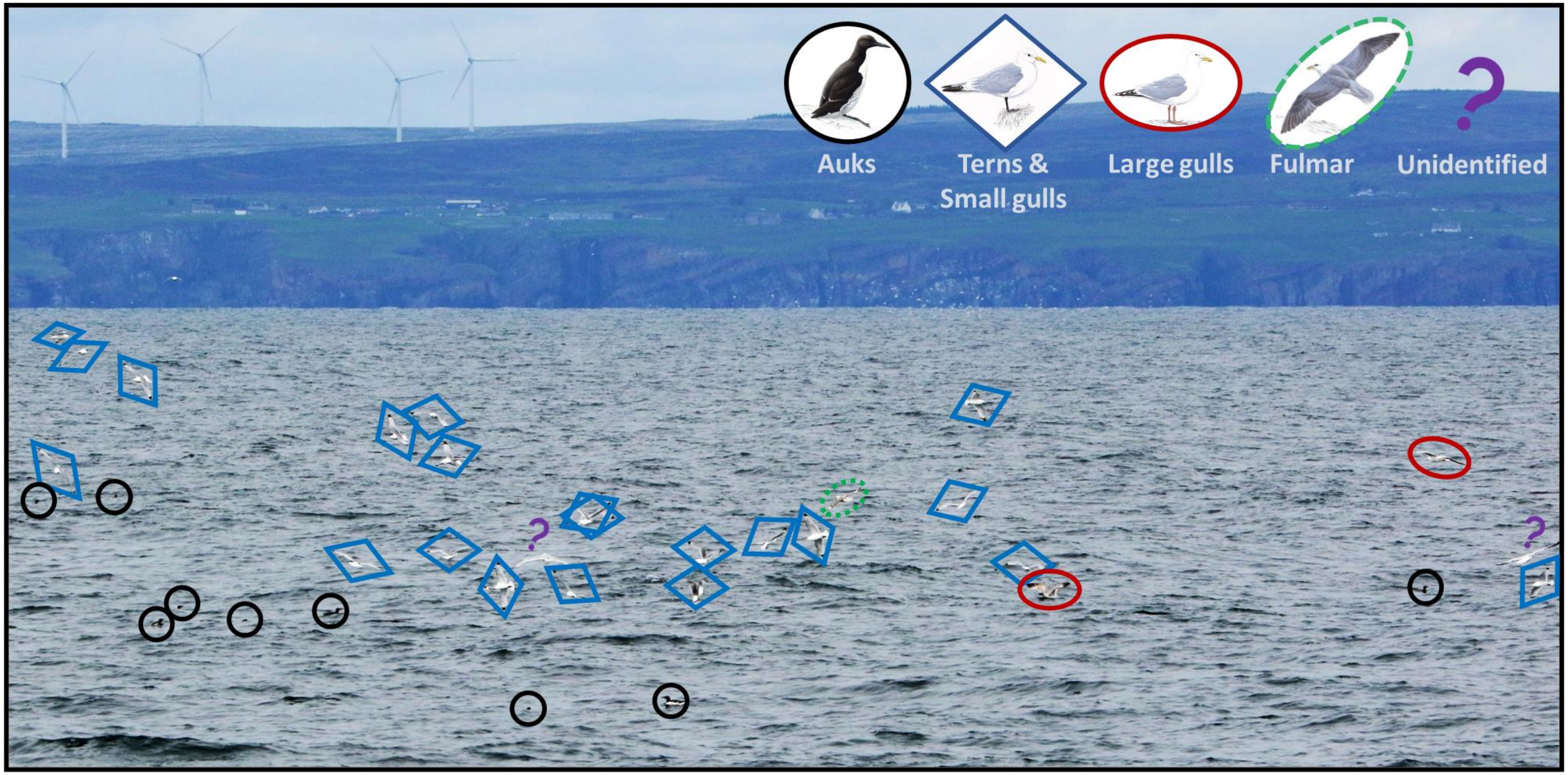

Figure S5.7 Example of multi-species foraging association between auk species (black circles), small gull species (mainly kittiwakes, blue diamond), large gull species (red oval), fulmar (dotted green oval) and unidentified species (purple question mark). This picture was taken in June 2019 at the beginning of the fisheries acoustic transect 2, in the outer Moray Firth, NE Scotland (see Figure S5.4). In the background, another flock of foraging birds can be seen. Source: University of Aberdeen

Figure S5.8 Example of multi-species foraging association between auk species (black circles), small gull species (mainly kittiwakes, blue diamond). This picture was taken in June 2019 at the Beatrice offshore windfarm, NE Scotland.

CHAPTER 6

GENERAL DISCUSSION

CHAPTER 6

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To limit the effects of climate change and reduce carbon emissions, global renewable energy policy has catalysed the windfarm industry sector, leading to the rapid expansion of offshore windfarm developments over the last two decades. However, this has had to be balanced against the potential adverse effects of these developments on marine wildlife, such as marine mammals and seabirds. As such, consented offshore windfarm projects are legally required to monitor, minimise, and when necessary, mitigate the potential impacts of the developments on protected species and populations. Focused research studies are critical for reducing key uncertainties identified during consenting, gathering empirical evidence to assess impacts of these developments and informing decision-makers and future developments. Although significant progress has been made to minimize and mitigate the major environmental stressors related to offshore windfarm developments (*e.g.* intense impulsive piling noise emission, increased collision risk), there is a growing need to assess cumulative effects of these activities to better estimate population level consequences of disturbances. Thus, addressing the gaps in our understanding of offshore windfarm impacts can reduce current barriers to development, improve the scientific knowledge base and allow decision-makers to implement appropriate conservation and mitigation measures.

The principal aim of this thesis was to improve understanding of some of the key impacts associated with offshore windfarm construction and operation on the occurrence, distribution and behaviour of harbour porpoises and breeding seabirds (Chapters 2-4). I also characterised the levels of vessel activity related to offshore windfarm developments (Chapter 2), quantified the magnitude of potential disturbance from different stressors (Chapter 2-3), and put them into the context of conservation and management (Chapters 2-4). Lastly, I intended to provide baseline information on the prey field within and around an operational windfarm, located on a sandbank, and investigated which prey availability measures influenced seabird distribution (Chapter 5).

KEY FINDINGS OF THIS THESIS

As cetaceans rely heavily on sound for navigation, communication, prey and predator detection, substantial changes in their acoustic habitats may interfere with their natural auditory signal processing (Tyack 2008). This may negatively affect behaviour and physiology (Tyack 2008; Duarte et al. 2021), and have potential consequences on both individual and population fitness (Pirotta et al. 2018; Pirotta et al. 2021). While most studies of harbour porpoises have focused on assessing levels of displacement during pile-driving activities (Tougaard et al. 2009a; Dähne et al. 2013; Haelters et al. 2015; Brandt et al. 2016; Graham et al. 2017b; Brandt et al. 2018; Graham et al. 2019), less attention has been paid to the potential effects of piling on the foraging activity of those porpoises that remain in the construction area. Similarly, behavioural impacts of other construction activities, which do not generate acute noise but are associated with increased shipping, have not been considered in environmental assessments, even though they can disturb marine mammals (Dyndo et al. 2015; Culloch et al. 2016; Marley et al. 2017; Oakley et al. 2017; Nabi et al. 2018; Putland et al. 2018; Wisniewska et al. 2018; Benhemma-Le Gall et al. 2021). In Chapters 2 and 3, I addressed these knowledge gaps using passive acoustic monitoring to detect echolocation clicks from porpoises and measure broadband noise levels. I also characterised vessel traffic within offshore windfarm construction sites and highlighted the diversity of vessel type involved in preparation (Chapter 3), construction and operation (Chapter 2), using AIS vessel-tracking data and engineering records from the piling vessels. These data highlighted that vessels within these construction sites can be classified into three categories. First, vessels directly involved in construction, which were mostly stationary and included the piling vessel, offshore supply vessels for jacket and turbine installation, and laying cables etc. Second, vessels supporting construction vessels, such as anchor handling tugs, barges, guard vessels and crew-transfer vessels, that were mostly in motion sometimes transiting from and to the construction sites. Third, vessels unrelated to construction that continued to transit through the windfarm outside small exclusion zones around specific activities. These included fishing vessels, research vessels, bulk carrier, and cargo vessels etc. The first two categories, *i.e.* the construction-related vessels, spent a lot of time within the windfarm, contributing to the high levels of vessel intensity observed. Due to the environmental risk associated with intense impulsive noise (Southall et al. 2008; Popper and Hastings 2009; Southall et al. 2019), piling activities has been the focus of research and mitigation measures. However, the piling campaign represents only a third of the overall windfarm construction phase. While the jacket and turbine installation activities may not have the potential to inflict acute injury, the high levels of vessel intensity associated with these other activities, throughout the whole construction phase, have the potential to affect the behaviour of a greater number of individuals.

In Chapters 2 and 3, I provided strong evidence that harbour porpoises respond to both piling and vessel activities during offshore windfarm construction. More specifically, I showed that porpoise occurrence decreased not only during pile-driving activities but also during wind turbine installation (Chapter 2) and preparation work, in the 12 hours before initiation of piling (Chapter 3). Harbour porpoise displacement was estimated at up to 12 - 17 km from the piling vessel and was not apparent beyond 4 km from construction vessels (Chapter 2). These results were in line with other studies that investigated harbour porpoise responses to pile-driving at other North Sea offshore windfarms (Tougaard et al. 2009a; Dähne et al. 2013; Haelters et al. 2015; Brandt et al. 2018; Nabe-Nielsen et al. 2018) and to vessels (Wisniewska et al. 2018). Further, I argued that even though harbour porpoise overall occurrence was higher during the Moray East piling campaign, the magnitude of response was similar (of the same order) at both windfarms, with a decline in detections between 8 and 14 % during piling (Chapter 2) and of 20 % during preparation work (Chapter 3). These findings suggest that despite the differences in installation techniques, vessel type and intensity between the two piling campaigns, the decline in porpoise occurrence was likely driven by the cumulative exposure to various disturbance sources. These results may be transferrable and help predict levels of porpoise behavioural response to construction and vessel activities for future offshore windfarm developments in the North Sea.

Behavioural response, such as avoidance, is frequently the measurable metric used to assess effects of anthropogenic activities on animals. However, anthropogenic disturbance may not always lead to avoidance as exposed animals may have to make trades-offs between leaving a disturbed area at an energetic cost or tolerating high levels of disturbance while remaining in a suitable area (Gill et al. 2001). In fact, if individuals are in a poor condition or live in a high-quality habitat, they might be less inclined to move away from an exposed area, especially if neighbouring areas are less suitable, highly competitive, or with high predation risk (*e.g.* Williamson et al. (2021)). Moreover, these trade-offs may influence the individual’s activity budget and indirectly affect both individual and population fitness, via the energetic cost of risk avoidance (Frid and Dill 2002). Similarly, disturbance associated with offshore windfarm construction activities have the potential to affect the activity budgets of animals exposed and remaining in offshore development sites. In addition to characterising displacement levels, I also demonstrated that porpoises remaining in the ensonified area during piling activities decreased their foraging activity. The causes of this remain unclear but could be due either to reduced foraging efficiency (Kastelein et al. 2019b) or reduced prey availability. On the other hand, porpoises deterred from the exposed area appeared to increase their foraging activity, potentially compensating for lost foraging opportunities or increased energy expenditure. The effects of piling on harbour porpoise foraging activity were detected up to 12 km at both windfarms. Similarly, porpoise foraging activity decreased with increasing levels of vessel intensity and broadband noise, highlighting that porpoises may adjust their activity budget, avoiding noise sources from vessel (Wisniewska et al. 2018) and development activities (Pirotta et al. 2014a; Sarnocińska et al. 2020; Benhemma-Le Gall et al. 2021) and spending less time foraging and/or socialising (Sorensen et al. 2018). However, for the same broadband noise levels, buzzing activity was higher during rather than in the absence of piling activities (Chapter 2; Figure 2.9B). This observed increase in porpoise buzzing activity, during impulsive noise activities, may be indicative of adaptative behavioural changes in echolocation activity to compensate for increased noise levels (Branstetter et al. 2018; Fernandez-Betelu et al. 2021), or linked to enhanced foraging opportunities due to potential changes in the structure and dynamics of schooling prey (Herbert-Read et al. 2017).

The BACI analyses in Chapter 2 also identified higher foraging activity in the impact block during the Moray East piling campaign. This could also be a response to cumulative changes in habitat quality and prey communities as this impact block encompassed both the Beatrice and Moray East windfarms. In 2019, during the Moray East piling phase, the northern part of the impact block contained 86 1-2-year-old offshore structures at Beatrice. Thus, this contrasting behavioural response may have been driven by enhanced habitat quality and prey availability within the study area (Scheidat et al. 2011), as previously recorded around oil and gas platforms (Todd et al. 2009; Clausen et al. 2021). These findings also suggest that porpoise responsiveness to pile driving may be reduced if buffer areas around construction sites provide better foraging opportunities. However, this now needs testing more explicitly (see below) to better understand cumulative impacts of construction when piling occurs near existing operational windfarms.

Once commissioned, potential impacts of operational windfarms are generally of greatest concern for seabirds, due to their risk of colliding with turbines and displacement (Bailey et al. 2014). Ornithology monitoring studies typically conduct pre- and post-construction digital aerial surveys to assess broad-scale (at-sea) seabird abundance and distribution within and around the development site (Webb and Nehls 2019). In Chapter 4, I used digital aerial imagery data collected before, during and after windfarm construction, to assess changes in the meso-scale (< 1 km) horizontal distribution, abundance, and behaviour of seabirds in the vicinity of wind turbine structures. I found that during construction, before the final installation of turbines, large gull species, especially herring gulls, were strongly attracted to jacket foundations with 72% of birds observed perched on jackets. This roosting behaviour was not observed after final construction, but high numbers of large gulls were observed on the water < 250 m from turbines. Attraction of gulls, such as lesser black-backed gull (*Larus fuscus*), to subsea structures has been previously reported at other windfarm sites (Dierschke et al. 2016; Vanermen et al. 2020). Despite the ecological benefits of this observed attraction to jacket foundations, such as increased roosting possibilities and enhanced feeding opportunities, this behavioural response may put seabirds at greater risk of colliding with turbines (Vanermen et al. 2015). Unforeseen by EIAs, this shifting baseline distribution and abundance of large gulls, part-way through construction, has not previously been reported and should be accounted for in collision-risk frameworks.

Identifying the natural drivers of seabird distribution within and around a newly commissioned site may also help contextualise the potential behavioural responses of seabirds to operational windfarms. The distribution and abundance of suitable prey is recognised to be a key natural driver of top predator distribution, but direct studies of these relationships are rare (Benoit-Bird et al. 2013). Mobile predators such as seabirds are also expected to forage where prey is most available to them, and this is not necessarily where prey is most abundant (Benoit-Bird et al. 2013). Areas with topographically contrasting features may aggregate food resources and plankton, and therefore attract forage fish and their predators (Stevick et al. 2008). However, in the marine environment, pelagic schooling prey can be a highly elusive and patchy resource (Fauchald and Erikstad 2002). As a result, foraging predators may also be patchily distributed, through local enhancement (Fauchald 2009). Anthropogenic activities, such as offshore windfarm developments may influence both predator and prey distributions due to changes in habitat conditions or increased levels of disturbance. Disentangling the natural variation in the spatial distribution of predators with the direct or indirect effects of windfarm development on predators remains challenging (Webb and Nehls 2019).

In Chapter 5, I characterised the underlying physical and biological processes influencing variation in prey and seabird distribution in the vicinity of an operational offshore windfarm, during the breeding season. I demonstrated that throughout the summer, production shifted towards higher trophic levels. The observed seasonal variation in the water column structure likely shaped distribution and abundance of primary and secondary producers (Fernand et al. 2013; Bresnan et al. 2015). Further, pelagic fish density and aggregative behaviour varied temporally (*i.e.* between surveyed months) and spatially (across the study area and within the water column). To briefly summarise, in June mixed waters, fish schools were patchy and mostly distributed near the surface, which may have enhanced foraging opportunities for surface-feeder seabirds. In August, waters were stratified, and overall fish density was higher, less aggregated and mostly distributed in the lower mixed water layer. Fish school prevalence and density was usually higher in deeper waters, outside the windfarm, off the sandbank. In the presence of fish schools, the distribution of three focal seabird groups with contrasting foraging strategies, were linked to two metrics of prey availability. Auks and large gulls were spatially associated with dense prey schools distributed near the surface and upper mid-water layer, while terns and small gulls preferred highly patchy prey schools. Exploiting prey patches close to the surface or sparsely distributed may optimise foraging profitability (Doniol-Valcroze et al. 2011; Chenoweth et al. 2021), be linked to species' foraging performance (Spitz et al. 2014), feeding method (Camphuysen and Webb 1999; Camphuysen et al. 2006), and prey selection or preferences (Wilson et al. 2004; Thaxter et al. 2013). Finally, this study demonstrated that seabird distribution was influenced by localised enhanced prey availability and not specifically by overall prey biomass, which is in accordance with previous studies (*e.g.* Benoit-Bird and McManus (2012)).

WIDER IMPLICATIONS FOR CONSERVATION AND MANAGEMENT

Understanding of the natural and anthropogenic drivers underlying the distribution and behaviour of marine top predators is essential to assess the environmental impacts of offshore windfarm developments on protected species and populations. Through strategic research work carried out in this thesis, I have provided evidence-based information on changes in the spatio-temporal distribution and behaviour of top predators in response to varying levels and sources of disturbance. These findings can be used by a wide range of stakeholders, to reduce uncertainty in assessing in-combination and cumulative impacts of offshore windfarm developments and optimise conservation and management of protected species, here in Scotland and in other parts of the world.

One of the key findings that may have implications for developers and marine spatial planners is the need to consider the potential environmental impacts of vessel traffic associated with windfarm construction and operation. Anthropogenic underwater noise is considered as a point-source pollutant and as such should be monitored and mitigated (Duarte et al. 2021). Shipping is one of the main components of anthropogenic noise in the ocean. To date, there is overwhelming evidence that marine mammals and other species can display a wide range of behaviours in response to vessels (Erbe et al. 2019). Although chronic exposure to ship noise may affect animals' behaviour and physiology and thus have consequences for individual and population fitness, shipping activity has not usually been considered in environmental impact assessments (EIA) for offshore windfarms. As highlighted in Chapter 2, increased vessel activity was recorded within and around offshore windfarm developments sites, along export cable routes and between local harbours and construction sites. The spatial coverage of disturbance associated with construction-related vessel noise may have extended beyond the windfarm site. In addition, increased vessel intensity resulted in a decline in porpoise occurrence and foraging activity (Chapters 2 and 3). Currently, when assessing the population consequences of disturbance from windfarm construction, EIA typically consider only piling activity. My results highlight that population modelling frameworks (*e.g.* PCOD and DEPONS) (Booth et al. 2017; Nabe-Nielsen et al. 2018; Nabe-Nielsen 2020; Dunlop et al. 2021; Pirotta et al. 2021) should be extended to also consider effects of vessel disturbance to better understand 1) the long-term fitness consequences of these different sources of disturbance, and 2) the cost-benefits of different mitigation measures to reduce disturbance effects.

As seen in Chapters 3 and 4, marine top predator baseline distributions and abundance can shift before the start of activities of greatest concerns (*i.e.* pile-driving for marine mammals and operational turbines for seabirds). By accounting for these potential changes in distribution and abundance, developers and other stakeholders may be able to predict, more accurately, the number of individuals at risk of physical injury or instantaneous death. For instance, ornithology monitoring studies would typically conduct pre- and post-monitoring surveys, but none is required during construction. Consequently, even though herring gull overall abundance within Beatrice windfarm was 124 times higher after than before construction, the small sample size of sightings before construction prevented a statistically robust assessment of behavioural responses of herring gulls to turbines (MacArthur Green 2021). Whenever possible, integrating data collected as part of ornithological monitoring studies from neighbouring offshore windfarm developments may provide further insights into seabird distribution and behaviour during construction (or other development stages) of the focal windfarm. This requires collaboration between developers and an extension of the survey coverage, which was the case with the Moray Firth windfarm developers.

Further, to reduce risks of near-field injury of harbour porpoises when pile-driving, the deployment of acoustic deterrent devices (ADDs) were integrated into engineering installation procedures (see Thompson et al. (2020) for further details). However, Thompson et al. (2020) demonstrated that the use of ADDs resulted in broad-scale displacement of porpoises beyond that required to mitigate injury. Accounting for the observed gradual decrease of up to 20 % in harbour porpoise occurrence before piling and deterrence activities may help to refine estimates of the number of animals within the injury zone to optimize mitigation measures, while minimizing broad-scale disturbance of acoustic deterrent devices on porpoises and/or non-target species (Thompson et al. 2020; Findlay et al. 2021). Similar assessments should be conducted in other regions to ensure these estimates are transferrable to other harbour porpoise populations or other cetacean species.

Further, prey availability and profitability are important drivers of predator behaviour and distribution. The results of Chapter 2 suggest that habitat quality and prey availability and biomass may be changing around constructed windfarms which, in return, may affect the magnitude of porpoise responses to disturbance. The prey fields described in Chapter 5 provide baseline knowledge for further work to explore the potential ecosystem benefits of operational windfarms which will permit more balanced EIA, accounting for, both the potential impacts and ecological benefits of man-made structures (*e.g.* Inger et al. (2009)). In addition, integrating empirical prey landscapes into population modelling frameworks may provide more accurate cumulative assessment of offshore windfarm developments on marine mammal populations (Nabe-Nielsen 2020).

DIRECTIONS FOR FUTURE RESEARCH

These key findings have raised a number of new questions about the extent to which cumulative and in combination effects caused by several asynchronous developments may affect protected populations.

Characterizing levels of vessel density and intensity in the Moray Firth and in the vicinity of offshore developments highlighted the extent to which construction vessels contribute to the overall vessel intensity and anthrophony of the Firth. However, other human activities unrelated to windfarm construction *e.g.* fishing, merchandise transport, recreational shipping, also contribute to the soundscape, and cumulatively may have long-term population consequences on protected highly mobile top predators. There is a need to quantify the spatial and temporal footprint of individual construction vessels, and at the fleet-level for each activity type. In return, this may help identify areas of high and lower vessel intensity, find ways to reduce impacts of vessel noise and help to optimise marine spatial planning and vessel management plans in the vicinity of development sites. Additionally, similar to the area/time disturbance thresholds implemented in English waters (JNCC 2020), allocating buffer areas around constructions sites to be quieter and with lower disturbance may reduce the magnitude of cumulative effects on porpoises displaced from ensonified areas.

Filling gaps in our understanding of how acoustic signatures vary in relation to vessel type, behaviour (*e.g.* speed), activity (Erbe et al. 2019) and abiotic parameters influencing underwater noise propagation (Weston 1971) is essential to reduce anthrophony through vessel management and the development of effective mitigation measures (Duarte et al. 2021). The need to reduce shipping noise pollution has been widely recognised by the research community (see Erbe et al. (2019) for a review) but has received, so far, limited high-level policy initiatives (Duarte et al. 2021). However, the COVID-19 pandemic has helped to draw attention to this topic, as national lockdowns drastically reduced human mobility causing global changes in marine traffic (March et al. 2021) and ship noise (Thomson and Barclay 2020; Ryan et al. 2021) permitting an assessment of the effect of this anthropause on wildlife (Rutz et al. 2020). In addition, the European Union's Horizon 2020 research and innovation programme, Saturn (2021-2025), bringing together a wide range of stakeholders, aims to quantify underwater radiated noise, assess noise exposure and responses of key receptors at the individual and population levels, and assess benefits and costs of mitigation solutions (Saturn 2021).

As highlighted throughout this thesis, prey availability is an important component driving top predator distribution, abundance, behaviour and responses to disturbance. However, given the challenge of studying the dynamic interactions between predators and their prey at appropriate spatial and temporal scales, monitoring studies often focus impact assessment on single receptors without integrating empirical information on their prey. During offshore windfarm construction, variation in sound pressure and particle motion from anthropogenic noise can affect fish behaviour, *e.g.* by disrupting communication (Siddagangaiah et al. 2021), school structure and dynamics (Herbert-Read et al. 2017) or influencing fish school vertical distribution in the water column (Kok et al. 2021) and movement (Mueller-Blenkle et al. 2010). Changes in natural patterns of fish distribution and behaviour may have indirect positive or negative effects on their predators. For instance, a lack of cohesion in fish schools may enhance predator’s foraging efficiency and opportunities, while variation in fish school vertical distribution as a response to impulsive noise may influence prey accessibility differently for predators with contrasting foraging abilities.

During windfarm operation, the introduction of hard structures and boulders on sandy bottom may locally modify hydrodynamic conditions with enhanced vertical mixing downstream of structures, potentially enhancing food resources near the surface (van Berkel et al. 2020). In addition, changes in the benthic and pelagic ecosystem including a gradual colonization of structures, a shift in fish assemblages and enhanced larval settlement is likely to increase production and foraging opportunities for forage and piscivorous fish and their predators (Reubens et al. 2014; Degraer et al. 2020). Evidence-based research has already demonstrated the ecological role of oil and gas offshore platforms as artificial reefs, through localized enhancement of biodiversity and biomass (Todd et al. 2009; Delefosse et al. 2018; Tassetti et al. 2020; Wright et al. 2020; Clausen et al. 2021; Todd et al. 2021). Similarly, offshore windfarm infrastructures may provide ecosystem services and ecological benefits and have effects on the trophic web structure and functioning (Raoux et al. 2017; Degraer et al. 2020). While changes in fish communities and biomass within operational windfarms has been highlighted (Andersson and Öhman 2010; Bergström et al. 2013; Krägefsky 2014; van Hal et al. 2017), the ecological benefits of increased biomass at these sites for top predators remain, to date, poorly documented. Evidence of attraction by pinnipeds to individual wind turbine subsea structures was reported (Russell et al. 2014). Finally, given the increase in harbour porpoise foraging activity around offshore oil and gas platforms (Todd et al. 2009; Clausen et al. 2021), a similar attraction to offshore windfarm structures is expected. Thus, further empirical information is required to improve our understanding of the reef effects around individual wind turbine structures for mobile fish and marine top predators. Concurrent passive and active acoustic monitoring can be used to compare the occurrence, abundance, distribution and behaviour of cetaceans and fish at different spatial (*e.g.* near- and far-field) and temporal scales (*i.e.* diel, tidal, seasonal). Additionally, underwater imagery techniques such as Baited Remote Underwater Video (BRUV) would provide further insights into the richness, abundance and assemblage composition of demersal and benthic fish and motile fauna around structures (*e.g.* Bicknell et al. (2016); Griffin et al. (2016); Bicknell et al. (2019)).

Further, research on the connectivity between individual structures and between windfarms and other offshore developments is required to better understand the magnitude of reef effects, habitat usage, larval dispersal etc. Studying the behaviour and movement of fish within and around the windfarm can be done by tagging fish and tracking their movements with an array of acoustic receivers (Taylor et al. 2018; Secor et al. 2020; van der Knaap et al. 2021). Similarly, animal-borne telemetry and accelerometery tagging techniques can be used to track the movement and behaviour of highly mobile species, such as seabirds, marine mammals and basking sharks (*Cetorhinus maximus*), at sea, in the vicinity of the windfarm, and in response to different disturbance sources (Russell et al. 2014; Thaxter et al. 2018; Wisniewska et al. 2018; Austin et al. 2019; Onoufriou et al. 2021).

Additional (long-term) broad-scale boat-based fisheries acoustic surveys will provide further information on fish biomass and distribution within and around operational windfarms to investigate changes in biomass production overtime, and spatially (*cf.* spill over effect), estimate prey availability for top predators with contrasting foraging abilities and provide prey landscapes to integrate into population modelling frameworks (*e.g.* DEPONS Nabe-Nielsen et al. (2018), iPCOD Booth et al.

(2017), SeabORD Searle et al. (2014)) or validate pre-existing prey habitat association models (*e.g.* Langton et al. (2021)). Fish samples primarily used for ground-truthing echotraces could also provide further information on their energy profiling and foraging ecology (De Troch et al. 2013).

Characterising, concurrently, top predator and prey distribution, abundance and behaviour within individual or a cluster of offshore windfarm developments can reduce uncertainties about potential reef and shelter effects of windfarm structures and improve understanding of the degree to which these moderate cumulative anthropogenic impacts of offshore windfarm developments. Thus, further strategic research work is required to assess the extent to which top predator responses are driven by direct disturbance from offshore windfarm developments or indirectly via changes in prey availability, and its consequences on population fitness.

This thesis highlighted changes in the baseline distribution of predators before the start of potentially harmful activities. A shift in the baseline is even more likely when clusters of windfarms are developed sequentially, next to each other, as is the case in the Moray Firth. This emphasizes the need to move towards cumulative impact assessment frameworks, integrating empirical data on prey landscapes and vessel intensity into pre-existing population models (Searle et al. 2014; Nabe-Nielsen et al. 2018), to improve predictions of population consequences of disturbances associated with developments.

Finally, the next round of offshore windfarm developments in Scotland (Scotwind) will deploy floating windfarms further offshore, making impact assessment and monitoring even more challenging. Early collaboration between developers, regulators and other key stakeholders should facilitate data collection at these future windfarm sites. For instance, working closely with engineers during the design of structures may enable the deployment of monitoring sensors on or below the structures. These sensors could collect data continuously to investigate the gradual and modulated changes in prey and predator behaviour and detections in relation to the environment around man-made structures (*e.g.* Marini et al. (2018); Baidai et al. (2020)). Fine-scale resolution data could be processed automatically, using emerging supervised learning techniques and other algorithms to extract key metrics that can be sent to researchers (*e.g.* Yang et al. (2019)). Similarly, autonomous vehicles, such as Unmanned Surface Vehicle (USV), would be able to pass near man-made structures and record oceanographic and acoustic data along specific transects to produce broad-scale prey and habitat landscapes (see Verfuss et al. (2019a) for a review). Finally, for windfarms using crew-transfer vessels for maintenance activities, developing monitoring techniques could also be valuable to deploy equipment, and collect data (*e.g.* marine mammal sightings, CTD casts (Carlson et al. 2021)) from these opportunistic platforms, all year-round. In return, all these innovative techniques may increase safety, and, in the long term, reduce overall cost of projects, carbon emissions and processing time for researchers.

IMPACT OF MY RESEARCH

The research conducted in this thesis have addressed key knowledge gaps and reduce uncertainty over the potential impacts of offshore windfarm developments on marine mammals and seabirds. Part of this work has been (Chapter 2) or will be (Chapters 3 and 4) published in peer-reviewed journals and Chapter 3 will be incorporated in the Moray East construction Marine Mammal Monitoring Programme. Additionally, I organised and delivered a workshop which aimed to inform and discuss the wider implications of the results found in Chapter 3 with developers, regulators and other statutory stakeholders. I have also presented parts of my research to several conferences and symposiums, through oral presentations, poster and videos, to both academic and non-academic audiences. Finally, Chapter 5 will provide baseline knowledge for PrePARED, a collaborative research project, funded by the Offshore Wind Evidence and Change programme and Crown Estate Scotland, to concurrently study predators and prey distribution and behaviour in the vicinity of offshore windfarms.

CONCLUSION

This thesis aimed to provide evidence-based estimates of marine top predator behavioural responses to various disturbance sources induced during offshore windfarm construction and operation. Using a wide range of monitoring devices and techniques, this thesis has successfully addressed key knowledge gaps on the natural and anthropogenic drivers of predator responses. For the first time, this study highlighted the magnitude of vessel activity at offshore development sites, provided evidence of changes in harbour porpoise foraging behaviour in response to vessel and construction activities, due to increased soundscapes. I also demonstrated that predator distribution and behaviour can shift before the start of activities likely harmful for the focal species, which is, to date, not accounted for in piling or collision risk assessments. Finally, I described the environmental drivers underlying fish school distribution and density within and around an operational windfarm, and identified key measures of prey availability that explain the distribution and prevalence of seabirds with contrasting foraging abilities. Further work is now required to integrate these key findings into cumulative impact assessment and population modelling frameworks to quantify the population consequences of cumulative impacts of offshore windfarm developments. This will help reduce current uncertainties, optimise mitigation measures and de-risk consenting process, while minimising the impacts on protected populations and increasing renewable energy supply to meet national decarbonization targets.

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