Co-designing opportunities towards the development of Irish offshore wind

Project acronym: EirWind
Collaborative project
Start date: 01st August 2018
Duration: 02 years

Work Package 4: Biological

Deliverable D4.13 Impacts from Offshore Wind Farms on Marine Mammals and Fish – A review of the current knowledge

Delivery date: 31st July 2019
Dissemination level: Public

By: William Hunt and Mark Jessopp

This project has received funding from the industry partners, Science Foundation Ireland (SFI) under Grant No 12/RC/2302, and University College Cork, Ireland

Disclaimer

The content of the publication herein is the sole responsibility of the authors and does not necessarily represent the views of the industry partners, Science Foundation Ireland, University College Cork or their services.

Without derogating from the generality of the foregoing neither the EirWind Consortium nor any of its members, their officers, employees or agents shall be liable for any direct or indirect or consequential loss or damage caused by or arising from any information advice or inaccuracy or omission herein.

Attribution - Please cite the report as follows:

W. Hunt, M. Jessopp (2019). *Impacts from Offshore Wind Farms on Marine Mammals and Fish – A review of the current knowledge*, EirWind Project Deliverable D4.13 Report, MaREI Centre, ERI, University College Cork, Ireland. DOI: http://doi.org/10.5281/zenodo.3948460

Document control:

Version	Date	Document History	Prepared by	Reviewed by	Approved by
01	26/04/2019		William Hunt	Mark Jessopp	Val Cummins
01.1	24/06/2019		William Hunt	Emma Critchley	Val Cummins
02	26/06/2019		William Hunt	Mark Jessopp	Val Cummins
02.1	19/08/2019	Additions to the Executive summary	Emma Critchley	Val Cummins	Val Cummins

Executive Summary

This report addresses the need for a robust overview of the science, in order to understand current knowledge of the impacts of offshore wind development, as the sector develops in Ireland. As such, the report summarises an extensive literature review of the known impacts from construction and operation of offshore wind farms (OWFs) on marine mammals and fish/shellfish. In addition, an overview of the currently available mitigation methods has been provided. The chief potential stressors of marine life from offshore wind farms have been identified as 1) noise, 2) built structures, 3) vessel activity, and 4) cables, comprising both power cables and mooring lines.

Noise is a known stressor of both marine mammals and fish and is currently the most significant stressor from OWFs. Impacts include acoustic trauma, hearing impairments, masking of biologically important acoustic signals, behavioural changes, and physiological stress. How these impacts manifest varies between fish and marine mammals, will often vary between species within these groups, and between size/age of individuals within a single species. Marine mammals detect noise through sound pressure alone, crustaceans likely detect noise solely through particle motion, fish may use either one or both. Recently revised thresholds for marine mammals in the literature may influence changes in Irish guidelines for permitted anthropogenic noise emission levels. Guidelines for thresholds for fish could also be adopted from the available literature.

Man-made structures have the potential to negatively impact marine life through habitat displacement, pollution, and behavioural disturbances. However, they have also proven to benefit ecosystems through the addition of complex hard substrate which increases biodiversity and may provide greater foraging opportunities, as well as shelter from predators, in addition to shelter from fishing activities and shipping noise. While localised increases in productivity and abundance of fish have been recorded around turbines, there is not yet evidence of increased productivity at a regional scale from the expansion of OWFs in European waters.

Vessels, whether commercial, shipping, or recreational, may impact both marine mammals and fish through noise, localised chemical pollution from leaks and spillages, possibility of ship-strikes, and as vectors for invasive species. Impacts include disruption to functional behaviours such as resting, foraging, or communication. In general, marine mammal numbers decrease in response to increased vessel numbers, and vessels travelling at speed have greatest impact. Fish display lower anti-predator responses, or spend more time guarding nests than feeding in the presence of increased vessel activity, and few reports exist of fish being struck by vessels.

The potential impacts from cables include possible entanglement, localised changes in the electro-magnetic field (likely negligible impact for marine mammals, local moderate impact for elasmobranchs and possibly some crustaceans), sediment suspension, noise/vibration, heat, and reef/reserve effects. However, this stressor is poorly studied, and more research would provide greater clarity on several of these issues.

Impacts will be best mitigated by appropriate site selection and assessment of potential windfarm sites, using the best available data on species spatial and temporal distributions or conducting additional surveys where necessary. Additional mitigation measures including time-area restrictions, the controlled use of acoustic deterrence devices, and other noise mitigation devices should be implemented to reduce impacts from noise – most importantly pile driving, but also vessel and operations noise. An update of Irish guidelines for permitted anthropogenic noise emission levels would help to inform these measures. Where possible, if one option has been shown to have lower negative impact over another available option, the first should be selected. Future OWF developments

should take the findings of this report in to consideration, and further research should be carried out to fill in the gaps in knowledge.

List of Abbreviations	
EMF	Electro-magnetic field
EPA	Environmental Protection Agency
GHG	Greenhouse gas
NMS	Noise mitigation system
NPWS	National Parks and Wildlife Service
OWE	Offshore wind energy
OWF	Offshore wind farm
OWT	Offshore wind turbine
SAC	Special Area of Conservation

Table of Contents

1 Introduction

1.1 Offshore wind

The burning of fossil fuels in the energy and transport sectors are the primary contributors to global greenhouse gas (GHG) emissions, and there is a growing need to reduce these (IPCC, 2014). This will require a shift towards renewable sources of energy, of which wind energy forms a viable and increasingly consistent option. Offshore wind energy (OWE) in particular is an attractive option to industry and regulators alike, due to more consistently high wind speeds and lower visual impacts (Higgins and Foley, 2014). While costs have been prohibitive in the past, these are continuing to fall (WindEurope, 2019), and the first non-subsidised offshore wind farm (OWF) has recently been granted approval for development in the Netherlands (WindEurope, 2018). Wind energy can assist Ireland in reaching targets for GHG emission reductions set by the EU and in transitioning towards greater energy independence (Department of Communications Energy and Natural Resources, 2014).

The first utility-scale offshore wind farm was established at Middelgrunden, Denmark in 2001 (EWEA, 2011). In the 18 years since then, Europe has become the world leader in OWE production (**Figure 1.1**) (International Energy Agency, 2018). Ireland's large continental shelf, with associated shallower waters (Dorschel et al., 2010) and consistent wind resource, make it ideally situated to harness OWE (Department of Communications Energy and Natural Resources, 2014). Despite this, Ireland has been slower to establish OWE than other EU nations, including the UK, Belgium, Germany, and Denmark. Recently published projections from the Environmental Protection Agency (EPA, 2019) show that Ireland will not meet the 2020 GHG emission reduction targets set by the European Commission (2009). If the country is to meet the EU's 2030 GHG emissions reduction targets, the generation of energy via renewable sources must be accelerated (Department of Communications Climate Action & Environment, 2019). OWE has the potential to make a significant contribution as part of the renewable energy mix. This has been recognised in the recent Climate Action Plan with a target for offshore wind energy to contribute 3.5 GW to Ireland's electricity by 2030 (Department of Communications Climate Action & Environment, 2019).

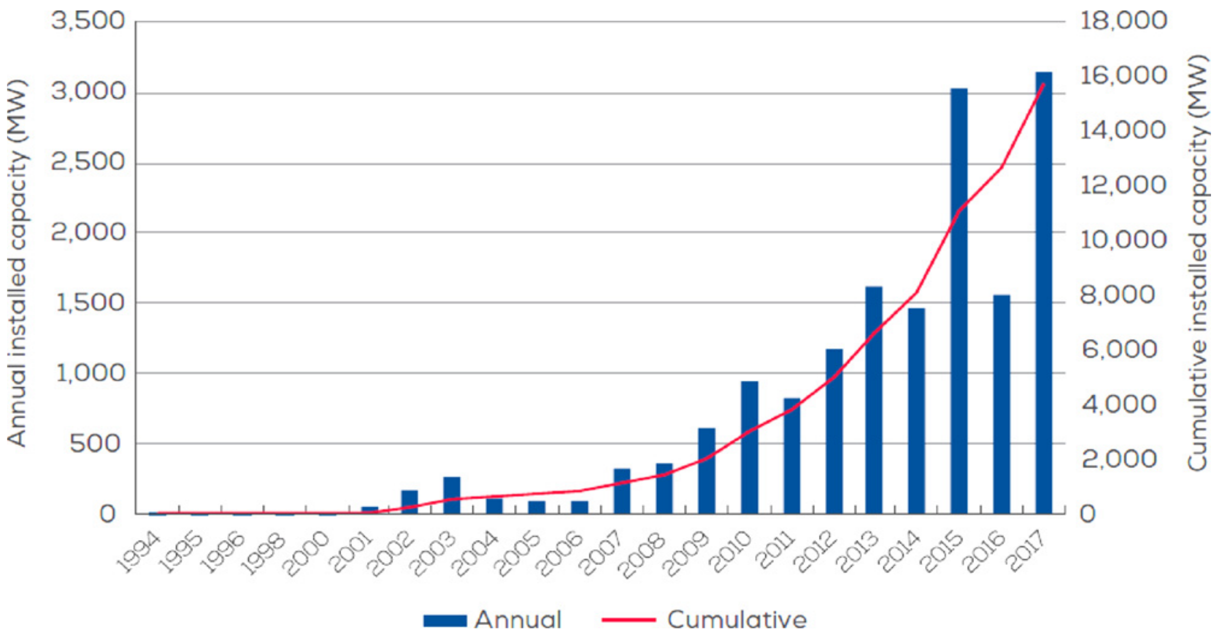

Figure 1.1: Offshore wind power installation in Europe 1994 – 2017 (WindEurope, 2018a).

However, such an expansion does not come without costs, including the potential for environmental impact. The ever-increasing footprint of humanity includes the proliferation of built structures within the marine environment, which has been termed 'ocean sprawl' (Duarte et al., 2013; Sanabria-Fernandez et al., 2018). Both construction and operation of OWFs has the potential to elevate anthropogenic noise levels above ambient ocean noise, the effects of which are being increasingly recognised as harmful to marine life (Popper and Hawkins, 2019; National Research Council, 2003; Williams et al., 2015). In order to reduce and, where possible negate this cost, we must identify what the risks are, how they might impact on marine life, and what mitigation can be applied.

Using searches for keywords in relevant online databases (Tethys, Science Direct, Scopus, and Google Scholar) this report has reviewed the available peer-reviewed and grey (technical reports and environmental impact statements) literature concerning potential impacts from the installation and operation of offshore windfarms on marine mammals and fish (including commercially relevant shellfish). It also provides an overview of the currently available mitigation methods.

1.1.1 OWF arrays

OWFS have typically been limited to installation at sites of <50 m depth, and more commonly in depths of 30 m (Higgins and Foley, 2014). The vast majority of OWFs to date are fixed-bottom, with the most common foundation types comprising gravity-based, monopile, and tripod or steel jacket types (**Figure 1.2:** Schematic of offshore wind turbine foundation types (Oh et al., 2018).). Depth of water and the associated environmental processes are the primary limiting factors when considering foundation type (Oh et al., 2018; Arrambide, Zubia and Madariaga, 2019; Wu et al., 2019). Scour protection is added to foundations to decrease substrate erosion at the base of the foundation arising from hydrodynamic processes. This additional hard substrate increases the heterogeneity of the hard structures and may prove beneficial to marine life (Stenberg et al., 2015).

Gravity-based foundations are the cheapest type and do not require any form of pile-driving during installation, significantly reducing their potential to impact on the marine environment. However, they can be limited by water depth and typically are not installed beyond depths of greater than 15 m, with a few exceptions such as those on the Thornton Bank (28 m) and in the Karehamn OWF in Sweden (Oh et al., 2018). Monopiles are by far the most commonly used foundation type in European waters given the shallow depths and general mix of sand and gravel substrate composition which make them easier to pile in (Oh et al., 2018). Their design can reduce the manufacturing cost, but their installation involves heavy-duty pile driving with an associated noise emission that is environmentally significant and can potentially impact negatively on marine life (Madsen et al., 2006; Brandt et al., 2018; Reyff, 2016; Popper and Hawkins, 2019; Bailey, Brookes and Thompson, 2014).

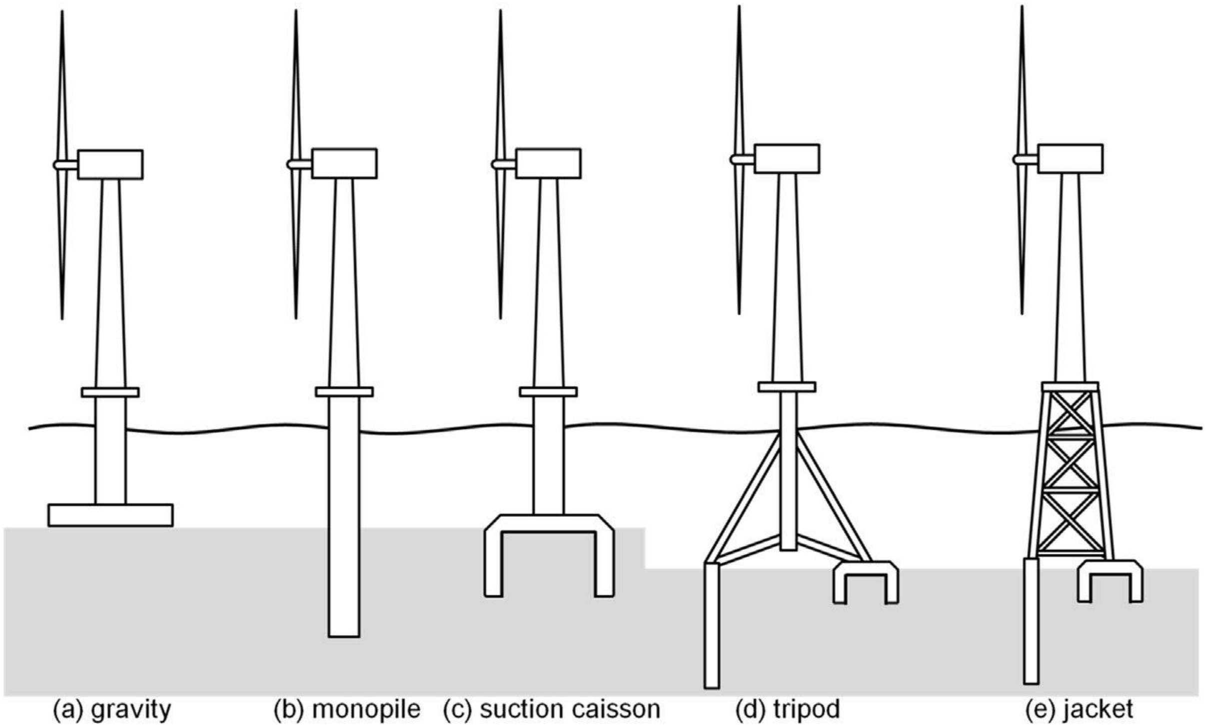

Figure 1.2: Schematic of offshore wind turbine foundation types (Oh et al., 2018).

Suction caisson foundations are designed with 'upside down bucket' base that has been demonstrated to require little time to install, significantly reduces the cost of material in their manufacture and does not require heavy piling (Oh et al., 2018). Thus, their environmental impact is greatly reduced compared to monopile foundations. However, there are concerns regarding their suitability for installations in water depths greater than 30 m due to the increased aerodynamic and hydrodynamic loads.

Multipod foundation types include tripod and jacket types (Oh et al., 2018; Wu et al., 2019). These are typically used in waters from 30 – 50 m depth and provide the turbine with sufficient strength and stiffness to overcome the greater loads associated with these depths, while keeping their costs relatively low. They are generally fixed to the seabed using smaller diameter piles driven less deeply than would be required for a monopile, but there is the potential to utilise the suction caisson base in future.

Floating wind turbine platforms are being developed to harness the greater and more consistent wind energy resource located further offshore in typically deeper waters (> 50 m) (Oh et al., 2018; Wu et al., 2019; Castro-Santos and Diaz-Casas, 2015; Sclavounos, Lee and DiPietro, 2010). Platform designs chiefly comprise three concepts: semi-submerged platforms (or "spar-submersible"); spar-buoy; and tension-leg platforms (**Figure 1.3**). These platforms will be tethered to the seabed using one of various anchor types: gravity anchors (deadweight anchors), anchor piles, drag embedment anchors (DEAs), vertical loaded anchors (VLAs), suction caissons, suction embedded plate anchors (SEPLAs), torpedo anchors, and deep penetrating anchors (DPAs) (or dynamically embedded anchors) (Wu et al., 2019). The environmental impact of these devices is generally unknown due to their nascent development. However, it should be much reduced on conventional, fixed-bottom OWTs due to lower installation-associated disturbances. While some noise disturbance may occur due to pile-driving anchor piles or possibly deep penetrating anchors it will likely be less than that from pile-driving noise emissions which occur during the installation of monopile OWTs. Furthermore, it remains to be seen which anchor type will become the prevalent system within the floating OWF sector.

Figure 1.3: Concept floating wind turbine platforms (IRENA, 2016)

1.1.2 Legislation

Under European legislation, developers are required to undertake Environmental Impact Assessments (EIA) to assess the potential effects of development and suggest appropriate mitigation (European Commission, 2011). The first step of such an EIA is to understand the risks that result from the proposed activity to the environment, identifying which components of the environment are most at risk, how that risk manifests itself, and what actions can be taken to eliminate, reduce, mitigate, or compensate for that impact.

In 1991, the entire Irish EEZ was designated a sanctuary for whales and dolphins (Rogan and Berrow, 1995). Further protections are afforded to marine mammals through the OSPAR Convention and various pieces of EU legislation including the Habitats Directive (European Commission, 1992) which, alongside the EU Birds Directive (European Commission, 2010), was consolidated into Irish law in the European Communities (Birds and Natural Habitats) Regulations 2011. Under the Habitats Directive, protection is afforded to species listed on Annex II, including the requirement for designation of Special Areas of Conservation (SACs) to protect core use habitats for native species. In an Irish marine context, species of relevance include harbour porpoise (*Phocoena phocoena*), bottlenose dolphin (*Tursiops truncatus*), grey seal (*Halichoerus grypus*), and harbour seal (*Phoca vitulina*). **Figure 1.4** displays current SACs designated for the protection of these species in Irish waters.

The Marine Strategy Framework Directive is also of relevance for the establishment of OWFs. This Directive provides a legal framework for an ecosystem-based approached to the management of human activities, supporting the sustainable use of marine goods and services. The primary objective is to achieve 'Good Environmental Status' within all EU waters by 2020, and specifically mentions noise from energy as a pollutant which requires regulation (European Commission, 2008).

In order to implement effective policies and manage marine areas to incorporate the conservation of marine life it is important to distinguish between effects and impacts. Boehlert and Gill (2010) proposed a framework for evaluating environmental effects in which they define effects as modifications of environmental parameters, termed "stressors", such as the substrate type, hydrodynamics, water temperature, noise, or electromagnetic fields beyond the range of natural variability. In contrast, an impact is a change that has occurred at the "receptor" level, i.e. on the different ecosystem compartments, or levels (community, populations) or some ecological processes within marine ecosystems (trophic interactions). Impacts may be positive or negative, although this distinction remains subjective (Taormina et al., 2018). This report looks at the potential impacts which may occur to the receptors of marine mammals, fish and shellfish resulting from the stressors occurring due to OWFs.

1.2 Marine life

1.2.1 Marine mammals

At least 25 species of marine mammals have been recorded in Irish waters (Rogan et al., 2018). These include species of cetaceans (whales, dolphins, porpoises) and pinnipeds (seals). Many of these species are transient, passing through Irish waters on their migration to and from feeding/breeding grounds (Reid, Evans and Northridge, 2003; Wall et al., 2013). Some of these migrants (fin whales, humpback whales) forage in Irish coastal and offshore waters during their passage, particularly in early spring and autumn to take advantage of plankton blooms and spawning and feeding aggregations of forage fish such as herring, sprat, and mackerel (Wall et al., 2013; Ryan et al., 2014, 2013). Other species are resident and present year around, and can be found in coastal inshore waters, offshore, or both (Reid, Evans and Northridge, 2003; Wall et al., 2013). Locations that have been identified as important breeding and calving habitats or haul out sites for some of these speices have been afford protection through the designation of SACs (**Figure 1.4**).

All marine mammals are vulnerable to disturbance from anthropgenic activities. Their slow maturation and low fecundity limits the rate of population growth, while many of the large baleen whale populations are still recovering from centuries of whaling activity. Coastal populations of cetaceans are particularly at risk from habitat loss and encroachment, contamination from chemical pollutants and toxins, bycatch in fisheries, and noise disturbances including chroinic stress from sources such as ship noise.

Figure 1.4: Designated SACs for marine mammals within the Irish EEZ. Data from the 'Protected sites' theme accessed through Ireland's Marine Atlas at http://atlas.marine.ie/, 18/06/2019

1.2.2 Fish

Irish waters also host highly productive ecosystems which support internationally important fisheries (White, Mohn and Orren, 1998; Marine Institute, 2018; Marine Institute and Board Iascaigh Mhara, 2017; Cummins, Lewis and Egan, 2016). Important spawning, nursery and fishing grounds are contained within the Irish EEZ (**Figure 1.5** and **Figure 1.6**) for commercial species including Atlantic mackerel (*Scomber scombrus*), Norwegian lobster (*Nephrops norvegicus*), Atlantic herring (*Clupea harengus*) and Atlantic cod (*Gadus morhua*) (Marine Institute, 2018). For the Republic of Ireland, the two most economically important fisheries are mackerel and nephrops (Marine Institute, 2018). In 2018, the Irish landings for mackerel was €59 m, while landings of nephrops were worth more than €56 m, and in 2017 the value of those two fisheries comprised just over half (52 %) the total value of Irish landings (O'Higgins and O'Hagan, 2019)

A number of these species are naturally vulnerable to environmental fluctuations (Pikitch et al., 2012; Rose, 2004). While many have an inbuilt resilience to small-scale climatic variations, their ability to recover may be compromised in the presence of additional pressures. High intensity fishing during unfavourable environmental conditions has contributed to collapses in certain stocks in the past (Pikitch et al., 2012; Pershing et al., 2015; Rose, 2004; Overholtz and Friedland, 2002). While there are efforts being made to reduce overfishing worldwide, the vast majority of assessed fisheries remain fully or overexploited (FAO, 2018) and the uncertain dynamics of climate change further fuels this unstable mix (Cheung et al., 2012). It is therefore vital that managers are cognisant of the potential for further, compounding, pressures which may arise.

Figure 1.5 and **Figure 1.6** display biologically sensitive areas for 4 commercially important fish and shellfish species. Herring (**Figure 1.5**) and mackerel (**Figure 1.6**) are important target species for the pelagic fishing fleet, and also play a vital role in the wider ecosystem as forage fish (Pikitch et al., 2012; Overholtz and Link, 2007; Ryan et al., 2014). Forage fish refers to the small pelagic species which act as a conduit for energy between the different trophic levels (Pikitch et al., 2012). They feed on zooplankton and are in turn preyed upon by a wide range of species including larger predatory fish (Rose and O'Driscoll, 2002), dolphins, seals, seabirds, and baleen whales (Ryan et al., 2014, 2013).

For many coastal small-scale fishing communities, cod (**Figure 1.5**) is an iconic target species, important both culturally and economically. Furthermore, they are an intrinsic component of the ecosystems in which they occur. Their role as a top predator helps to regulate the structure of those communities, and their loss to an ecosystem could result in a species regime shift (Rose and Rowe, 2015). Other commercially important members of the wider cod-family (the gadidae) include pollock (*Pollachius pollachius*), haddock (*Melanogrammus aeglefinus*), whiting (*Merlangius merlangus*), saithe (*Pollachius virens*), and hake (*Merluccius merluccius*) (Marine Institute, 2018).

Nephrops fishing grounds (**Figure 1.6**) are of vital importance for the economic success of the Irish polyvalent fleet. Commercially known as Dublin Bay prawn, langoustine, or Norwegian/Norway lobster, this species live in burrows in sandy/muddy sediments, with geographically distinct populations separated by isolated habitats and hydrodynamics (O'Sullivan et al., 2014). Low fecundity and sensitivity to temperature changes during their larval stages make them vulnerable to disturbance (O'Sullivan et al., 2014).

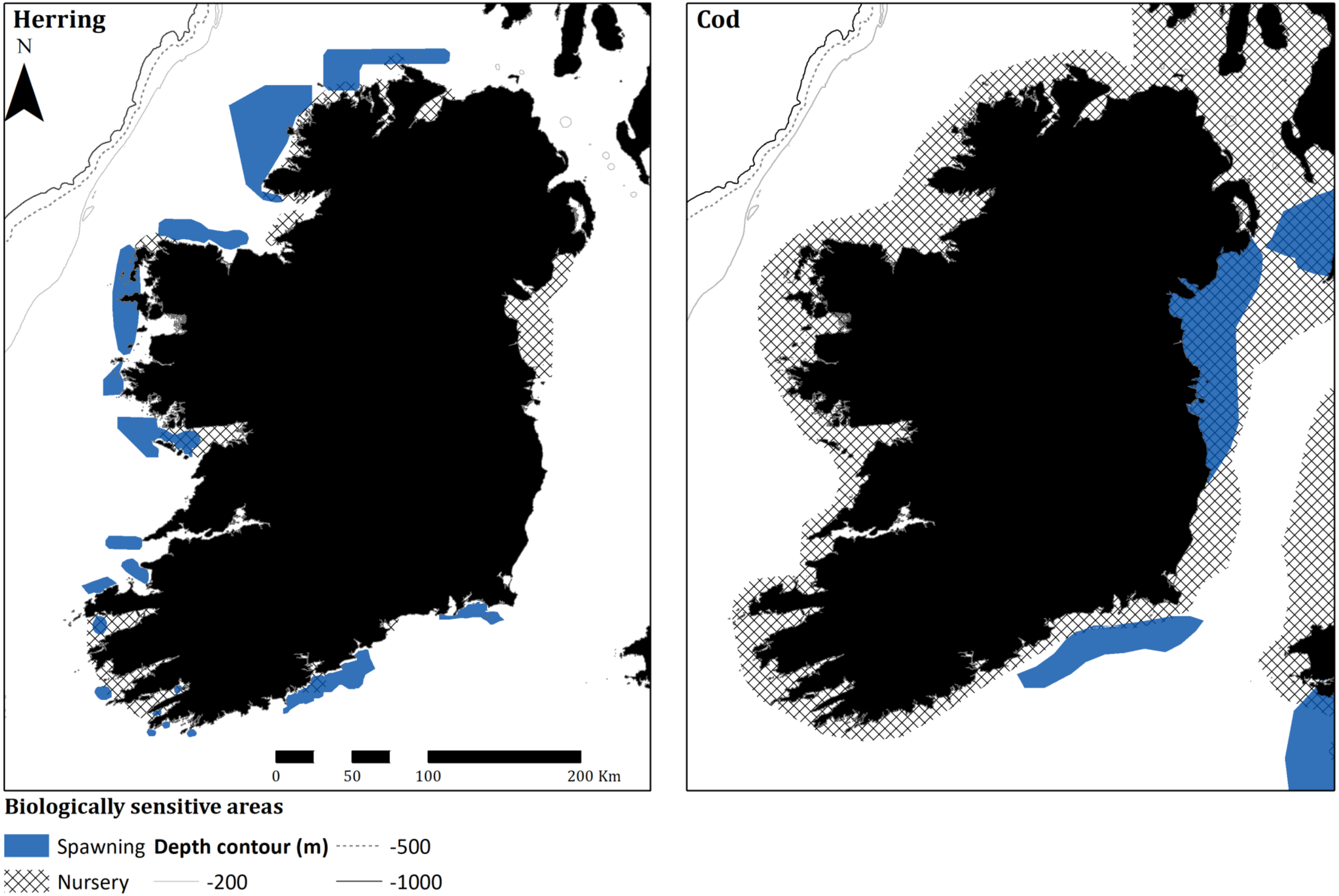

Figure 1.5:: Biologically sensitive areas. Herring and cod spawning locations and nursery grounds. Data from the 'species distribution – sea fisheries' theme accessed through Ireland's Marine Atlas at http://atlas.marine.ie/, 18/06/2019.

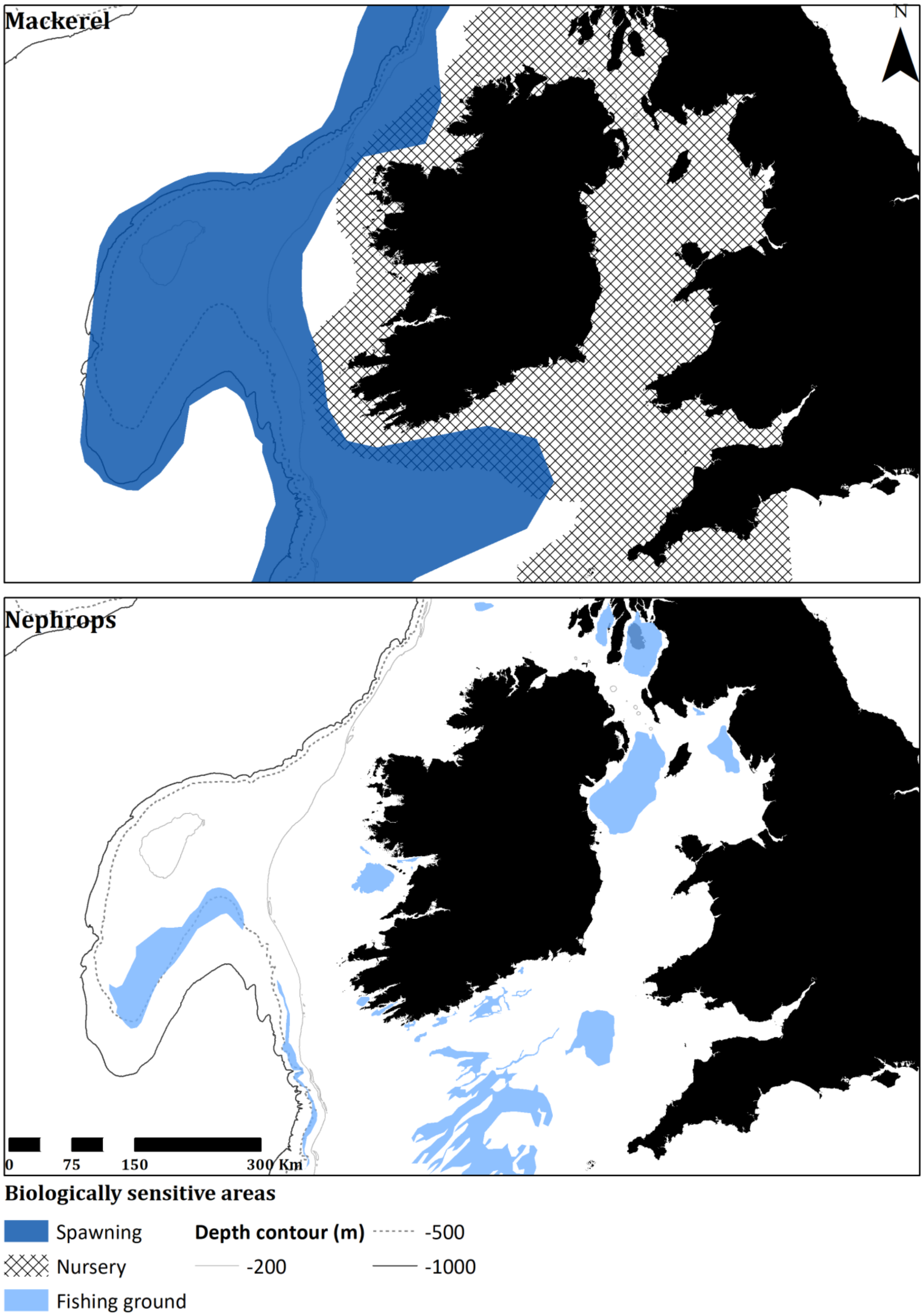

Figure 1.6: Biologically sensitive areas. Mackerel spawning locations and nursery grounds, and nephrops fishing grounds. Data from the 'species distribution – Sea Fisheries' theme accessed through Ireland's Marine Atlas at http://atlas.marine.ie/, 18/06/2019

2 Noise

2.1 Summary

2.1.1 Marine mammals

Sound plays a pivotal role in the life of marine mammals. It is their primary tool for understanding their environment, and is used for communicating, navigating, and foraging. Consequently, they are highly sensitive to acoustic signals and have well-developed sensory organs adapted to the physical properties of underwater noise. Sudden changes to the underwater soundscape beyond their natural hearing range is likely to cause disturbance. Noise is viewed as the most significant potential stressor for marine mammals from offshore wind farm developments.

The intense, percussive, high-frequency noise emitted by pile-driving activities is deemed the most likely to cause disturbance (through hearing impairment, masking of biologically important acoustic signals, habitat displacement, behavioural changes, physiological stress). Pinniped hearing range also overlaps with the low frequency noise emitted during the operation of offshore wind turbines (OWTs), as well as with ship engine noise. While direct, physical injuries as a result of noise are unlikely to occur, there is a significant threat of hearing impairment, either temporary or permanent, if mitigation methods are not employed. Furthermore, masking of biologically sensitive cues, habitat displacement, and behavioural disturbances may occur up to 25 km from the source of pile-driving noise, depending on the sensitivity of the species present and physical characteristics of the local environment.

2.1.2 Fish and shellfish

Anthropogenic noise has been shown to cause harm to fish. While there is little evidence for death as a direct result from man-made sound, there is some risk of direct acoustic trauma, and a high risk of hearing impairments, masking of important biological signals, changes to behaviour, and physiological responses to noise. It is clear that exposure to short-duration, high-intensity noise events causes spikes in stress responses. However, the current state of knowledge suggests that these stress responses revert to baselines levels relatively soon following the cessation of the noise emitting activity. However, it is also clear that chronic exposure to long-duration noise, even at low levels, can result in pervasive and long-lasting stress responses. While in some cases habituation may occur, negative impacts on reproduction and general fitness have also been observed.

The majority of the studies to date have been carried out in laboratory environments, and the response of wild fish to noise may differ in the field, where they are unconfined. Furthermore, particle motion is an important component of sound and is the primary means of detecting sound for most fish species yet is conspicuous by its absence in many studies assessing the impact of sound on fishes. This is primarily due to difficulties in measuring particle motion, particularly in the field. This has also led to very few studies on how sound impacts shellfish, as it is currently believed that most if not all crustaceans detect sound only via particle motion.

Impulsive noise sources such as pile-driving are the most studied threat to date; however, low-level continuous noise such as that from shipping and operational OWT is becoming increasingly recognised as a potentially significant stressor on marine life, including fish and shellfish. While this field has seen a large growth in studies since the late-2000s, there is still much to be done in order to appropriately quantify the risk from anthropogenic sound to fishes and shellfish.

2.2 Receptors

The literature has identified a number of potential impacts from noise on receptor species including marine mammals and fish (**Table 2.1**). Noise impacts during the construction phase have been considered most critical, whilst noise generated from operational wind turbines has so far been deemed negligible, with the caveat that vibration into the seabed may disturb some benthic species. Lindeboom *et al.* (2011) found that during relatively low wind speeds (1.8 – 9.7 ms^{-1}) turbine noise emissions were in the 875 – 1500 Hz frequency range, varied between 125 – 130 dB re 1 $\mu Pa^2\ s^{-1}$, and were undetectable beyond 300 m from source. While these values will increase with stronger wind speeds, these values are far below those which may be considered harmful to most fish based on the current knowledge.

Table 2.1: Potential effects of anthropogenic sound on animals adapted from Popper and Hawkins, (2019).

Effect	Description
Death	Sound exposure results in instantaneous or delayed mortality.
Physical injury & physiological changes	Physical injury results in temporary or permanent impairment of the structure and functioning of some parts of the body. Physiological changes result in increased stress or other effects that can lead to reduced fitness.
Hearing threshold shift	Loss of hearing, temporarily or permanently, results in decreased ability to respond to biologically relevant sounds.
Masking	Noise results in a decrease in detectability of biologically relevant sounds (e.g., sounds of predators and prey, sounds of conspecifics, acoustic cues used for orientation).
Behavioural responses	Behavioural responses include any change in behaviour from small and short-duration movements to changes in migration routes and leaving a feeding or breeding site. Such responses are likely to vary from species to species, depending on numerous factors such as the animals' normal behavioural repertoire, motivational state, time of day or year, age of the animal, etc. Some changes in behaviour, such as startle reactions, may only be transient and have little consequence for the animal or population.
No obvious behavioural responses	Animals may show transient or no responses, even if they detect the sound (e.g., to a very low-level sound) or habituation may take place. However, even if there is no response, there is always the possibility that physical injury and physiological changes may take place without the animal showing overt changes in behaviour

2.2.1 Marine mammals

Noise is viewed as the most significant potential stressor for marine mammals from offshore wind farm developments (Lovich and Ennen, 2013; Bailey, Brookes and Thompson, 2014; Bergström et al., 2014; Hooper, Beaumont and Hattam, 2017; Nabe-Nielsen et al., 2018). However, while impacts from noise such as habitat displacement and hearing damage have been relatively well described, very little is known concerning other physiological impacts on marine mammals, or the effect of chronic exposure to low-level noise pollution. Chronic exposure may occur during the operational phase of OWFs. However, of greater concern is the noise produced during pile-driving foundations for wind turbine pylons. The noise levels and frequencies at which this impulsive, percussive sound is emitted can cause temporary and permanent hearing loss in some marine mammals at close range (Lucke et al., 2009) and also affect their behaviour at greater distances. Piled anchors used for floating wind turbine arrays (Wu et al., 2019) may also be of concern, but very little is currently known regarding this specific scenario. In order to assess any potential impact from noise on marine mammals at both the individual and population level, it is necessary to be able to measure sound levels accurately and ascertain at what intensity they pose a risk to the health of marine mammals. This necessitates both an understanding of marine mammal hearing and how sound is transmitted through water.

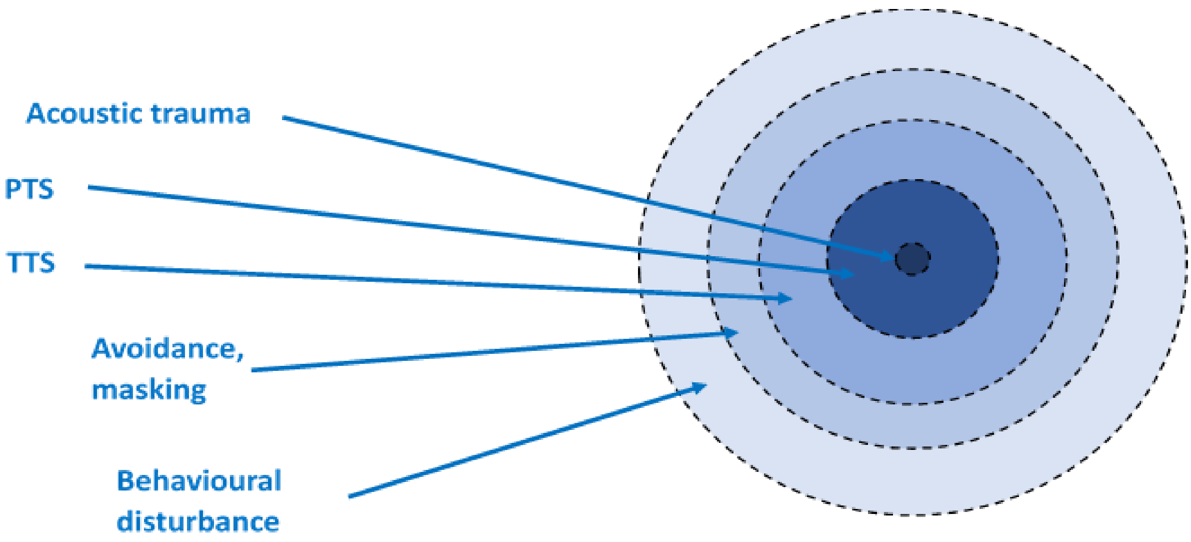

Figure 2.1: Schematic representation of increasing impact zones relative to a point source of impulsive noise

Southall *et al.* (2007) previously described a dual criterion for assessing the levels of noise to which marine mammals are exposed: unweighted peak sound pressure level (SPL), and the acoustic energy of sound, referred to as the sound exposure level (SEL). SPL is a ratio of the absolute sound pressure and a reference level, for example the lowest intensity sound that can be heard (Richardson et al., 2013). SEL is a frequency-weighted metric that describes the amount sound pressure to which an animal is exposed over a given time (Southall et al., 2007). Another term of importance is the received level (RL), which is the level of sound energy when it is heard by the receptor, in this case a marine mammal. In addition, frequency-weighting functions, called M-weighting, based on the known or estimated auditory sensitivity at different frequencies (rather than vocal characteristics) was defined for marine mammals. These thresholds, defined by Southall *et al.* (2007) have been used by the Irish government to provide guidance on accepted levels of noise exposure from anthropogenic sound sources (National Parks & Wildlife Service 2014), which are contained in **Table 2.2.**

Table 2.2: NPWS criteria and values for TTS-onset (single pulse only) and disturbance/behavioural response (multiple pulses and non-pulses). SEL: frequency weighted sound energy level; SPL: unweighted peak sound pressure level; RL: received energy level. Adapted from National Parks & Wildlife Service (2014).

Noise type	Cetaceans			Pinnipeds	
	Low-frequency	*Mid-frequency*	*High-frequency*	*Water*	*Air*
Single Pulse	224 dB SPL 183 dB SEL	224 dB SPL 183 dB SEL	224 dB SPL 183 dB SEL	212 dB SPL 171 dB SEL	109 dB SPL 100 dB SEL
Multiple pulse	120-180 dB RL Not applicable	120-180 dB RL Not applicable	Data unavailable Not applicable	150-200 dB RL Not applicable	Data unavailable Not applicable
Non-pulse	120-160 dB RL Not applicable	90-200 dB RL Not applicable	90-170 dB RL Not applicable	100+ dB RL Not applicable	110-120 dB RL Not applicable

* Units of measurement:
Sound Pressure Level, SPL (in water): measured in dB re 1 µPa (peak) (flat)
Sound Exposure Level, SEL (in water): measured in dB re 1 $\mu Pa^2 s^{-1}$
Sound Pressure Level, SPL (in air): measured in dB re 20 µPa (peak) (flat)
Sound Exposure Level, SEL (in air): measured in dB re 20 $\mu Pa^2 s^{-1}$

Recent studies have tended towards using SEL as the most appropriate parameter to measure the intensity of noise that marine wildlife is subjected to (Dähne et al., 2013; Tougaard, Wright and Madsen, 2015; Finneran, 2015) due to better correlation with the onset of hearing loss in marine mammals (Tougaard and Mikaelsen, 2018). Furthermore, the cumulative sound exposure level (SEL_{cum}) is considered best when assessing the risk from repeated, impulsive, sound emissions such as non-vibratory pile-driving activity (National Marine Fisheries Service, 2016). SEL_{cum} is the total amount of acoustic energy an animal may be exposed to from multiple noise events (e.g. multiple strikes of a hammer on a pylon) during a specified period, usually 24 hours (but it may be shorter depending on species and scenario-specific circumstances; greater research is needed in this area), from either a single continuous event, or multiple events with short breaks between them. However, Southall *et al.*, (2019) recently published an extensive update to their earlier review. In it, they maintain the relevance of both parameters: SEL only for non-impulsive or broadband noise sources; with dual metrics for impulsive noise criteria: frequency weighted SEL and unweighted peak SPL.

2.2.2 Fish and shellfish

All fish studied to date are able to hear sound (Slabbekoorn et al., 2010). Over 800 species from 109 families are also known to produce sounds, and many more are suspected of being capable of doing so (Radford, Kerridge and Simpson, 2014). Sound is used to communicate aggression and courtship behaviours (Bass and Ladich, 2008; Kasumyan, 2008), to detect predators and prey, for orientation, and in primitive forms of echolocation (Andersson, 2011; Fay and Popper, 2000; Popper and Fay, 2011, 1993). The enormous diversity of fish – 32,000 described species, more than half of all known vertebrates (Helfman et al., 2009) – makes it nearly impossible to derive species-specific guidelines for reducing the potentially harmful impacts from anthropogenic noise.

Fish detect noise in two ways: air-filled cavities such as swim bladders detect sound pressure changes and specialist organs or structures (otolith complexes and lateral lines) detect particle motion. Either or both may occur in a single species. Certain specialised structures, e.g. the connection between swim bladder and inner ear in clupeids (e.g., herring and sprat), lead to greater sound sensitivity in some species (Popper and Hastings, 2009a; Popper and Hawkins, 2019). Popper *et al.* 2014 described 4 groups to which the vast majority of fishes could be assigned in terms of how they perceive noise: 1) fishes lacking swim bladders that are sensitive only to sound particle motion; 2) fishes with a swim bladder where that organ does not appear to play a role in hearing; 3) fishes with swim bladders that are close, but not intimately connected to the ear; and 4) fishes that have special structures mechanically linking the swim bladder to the ear. **Figure 2.2** illustrates the hearing ranges for a representative species for each of these groups, and clearly demonstrates the differing levels of hearing sensitivity.

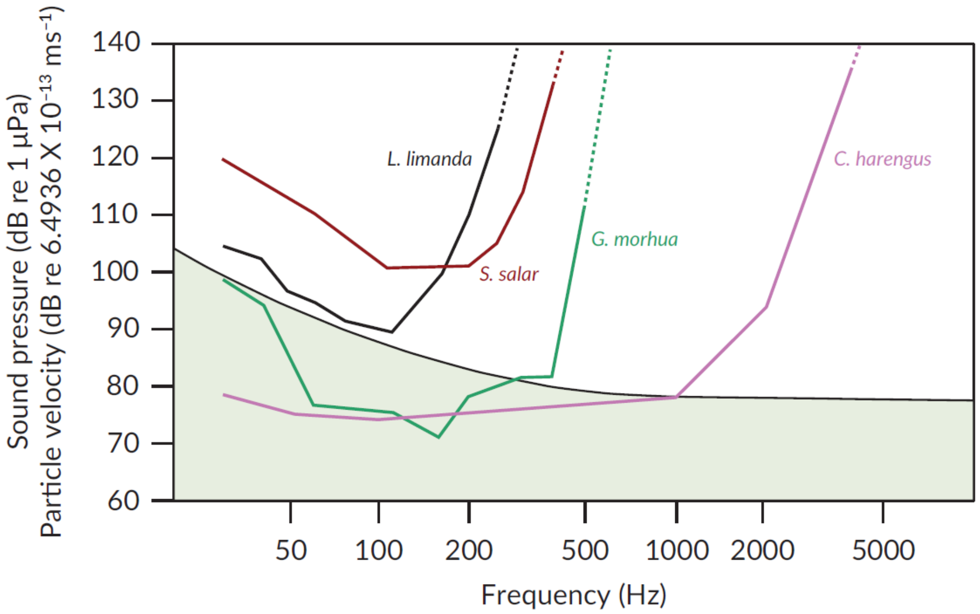

Figure 2.2: Fish hearing sensitivity (thresholds) obtained under open sea, free-field, conditions in response to pure tone stimuli at different frequencies (Popper and Hawkins, 2019). The lower the thresholds (y-axis), the more sensitive the fish is to a sound. Thus, *Clupea harengus* has the best hearing of all of these species over a wider range of frequencies. Note that the thresholds in *Gadus morhua* and *C. harengus* obtained under quiet conditions may be below natural ambient noise levels, especially at their most sensitive frequencies. In the presence of higher levels of noise, the thresholds would be raised, a phenomenon referred to as masking. *Gadus morhua* and *C. harengus* are sensitive to both sound pressure and particle motion, whereas *Limanda limanda* and *Salmo salar* are only sensitive to particle motion. The reference level for the particle velocity is based on the level that exists in a free sound field for the given sound pressure level. *n.b.*, For the particle velocity levels in this figure to match the sound pressure levels in a free sound field it is necessary to calculate an appropriate particle velocity reference level. If the standard reference levels are used, then the curves will not match one another and so they are not included here to keep the figure relatively simple. Fig. © 2018 Anthony D. Hawkins, all rights reserved.

Noise-generating human activities in aquatic environments, such as commercial shipping, recreational boating, pile-driving, seismic exploration, and energy production, are widespread and occur with increasing frequency (Radford, Kerridge and Simpson, 2014). Indeed, the expansion of offshore wind farms in to already noisy areas, such as shipping zones, have been found to increase the levels of noise that may be perceived by fish (Andersson, Sigray and Persson, 2011). In their 2009 review of impacts on fish from such human-generated noise, Popper & Hastings found a paucity of relevant studies available: while many potential impacts had been described, very little had been empirically proven. Since then, a body of work has steadily increased to experimentally demonstrate some of these effects.

However, as Popper & Hawkins (2019) emphasise in their recently published review of fish bioacoustics and the impacts of anthropogenic sounds on fishes, much of these studies have been performed in laboratory conditions or enclosures which do not accurately reflect natural environmental conditions; on species which may not be truly representative of populations encountered in the wild, or had been entirely lab-reared; and the majority of studies consider sound pressure alone rather than also measuring particle motion, which is of greater importance for most fish species. While lab-based studies can provide great detail, the inability to replicate in-the-field environments reduces their power to accurately predict effects on marine life. Furthermore, it is important to note that different behavioural responses may occur depending on level of sound, the level of ambient background noise, what the fish is doing at the time of the noise, and its previous experience with that sound and other sounds (Popper, Hawkins and Halvorsen, 2019). Each of these caveats are of significance when determining if the results of studies are capable of being extrapolated to wild populations. For example, all studies of stress responses to human-generated noise have been conducted on fish in confinement. The results have been variable, but many have recorded some degree of physiological or behavioural stress response. However, it is possible that these stress responses may have been induced by the inability to flee the noise source rather than by the noise source itself, or a combination of both.

2.3 Death

Very little evidence of instantaneous death or mortal injuries has been provided by experimental studies (Popper and Hawkins, 2019). To date, the only report of direct mortality comes from fish in very close proximity to pile driving activities in a single study (California Department of Transportation, 2001). Debusschere *et al.*, 2014, found no differences between control and exposure groups of juvenile European seabass (*Dicentrarchus labrax*) exposed to pile-driving noise occurring 45 m distance. These findings agree with controlled laboratory experiments (Halvorsen et al., 2012c; Casper et al., 2017, 2013a, 2012; Bolle et al., 2012).

In a review by Edmonds *et al.* (2016) it is noted that blue crabs (*Callinectes sapidus*) have suffered mortality as a result of exposure to close range underwater explosion. While no lethal effects have been observed among the three main commercial species in UK and Irish waters (*Cancer pagurus, Hommarus gammarus, Nephrops norvegicus*), sublethal physiological effects have been reported for *N. norvergicus*.

2.4 Acoustic trauma

2.4.1 Marine mammals

Acoustic trauma is direct damage to tissues caused by an acoustic blast (Tougaard and Mikaelsen, 2018). These blasts arise from very loud, impulsive noises which may occur during pile driving activity, however, acoustic trauma is more likely associated with military sonar activities (Parsons et al., 2008; Dolman et al., 2011; Cox et al., 2006; Tyack, 2009; DeRuiter et al., 2013). To date, no study has attributed acoustic trauma to pile-driving, nor has there been any reported mortalities.

2.4.2 Fish and shellfish

Sudden changes in water pressure, such as those caused by the percussive impact of pile-driving, may cause acoustic trauma (or barotrauma) to fish (Halvorsen et al., 2012c; a; Casper et al., 2013b; Halvorsen et al., 2012b; Casper et al., 2017). Fishes which have gas-filled cavities (i.e., swim bladders) are of chief concern. Impulsive sound energy causes the walls of such bladders to oscillate, creating friction between the bladder and neighbouring organs (e.g., liver, kidney, gonads). This may lead to cell and tissue damage of those organs, and in extreme cases could be lethal (Popper and Hawkins, 2019). Species such as Atlantic cod (*Gadus morhua*) and Atlantic herring (*Clupea harengus*) are more susceptible to acoustic trauma impacts (Hammar, Wikström and Molander, 2014; Thomsen et al., 2008; Popper and Hawkins, 2019) than flatfish such as common sole (*Solea solea*) or European plaice (*Pleuronectes platessa*) (Popper and Hawkins, 2019), which don't have swim bladders and use particle motion to detect sound.

Recent studies have shown that such injuries occurred only above a certain threshold. While there has been some variation in the exact value of this level, it is always at SEL_{cum} levels >203 dB re 1 µPa^2 s^{-1} (Casper et al., 2013a; Halvorsen et al., 2012c). In addition to damage caused by the oscillation of gas bladder walls, impulsive noise levels have been demonstrated to damage the sensory hair cells which line the inner ears of fish (Halvorsen et al., 2012c; Casper et al., 2017, 2013a, 2012). In all cases of cell and tissues damage, recovery and regeneration of damage occurred within 10 days – but these studies were carried out in laboratory environments only. Fish with such injuries in the wild may be susceptible to predation and disease until fully recovered.

The percussive, impulsive noise levels generated by unmitigated pile driving are of sufficient intensity to cause harm (**Table 2.5**). While it is very possible that fish will either avoid the construction site or be able to flee the impact zone before harm is caused, no direct studies to experimentally prove this have been carried out. However, operational wind turbine noise has not been shown to cause any form of direct acoustic trauma to fish. Hammar, Wikström and Molander, (2014) found that the construction phase of OWFs in the Kattegat posed a high risk to Atlantic cod due to their vulnerability to both pile driving and vessel noise; however, risks from the operational phase (including from turbine noise but also from EMF and lubricant spills) were considered low to zero.

2.5 Hearing loss

2.5.1 Marine mammals

The inner ear of marine mammals is the organ most likely to be injured by the type of impulsive sound generated by pile-driving activity during construction of OWFs (Tougaard and Mikaelsen, 2018; Southall et al., 2007; David, 2006). As such, it is likely that auditory injury will occur at lower sound levels than for other body tissues (Southall et al., 2007). Exposure to loud sounds has the potential to shift the threshold at which animals can hear noise (i.e. cause hearing loss), and damage/threshold shift may be temporary (TTS), lasting for only a few minutes or up to days, or permanent (PTS). Shifts may occur due to exposure either to a short-period, high intensity sound source, or longer-periods of lower intensity noise (Finneran, 2015; Tougaard, Wright and Madsen, 2015; Mooney et al., 2009; Southall et al., 2007). Loss of hearing will impact an animal by potentially disrupting their ability to forage, communicate, and navigate.

Both TTS and PTS refer to the 'new' level of noise that can be heard, measured in decibels (dB). According to the US National Marine Fisheries Service (2016) an initial TTS of 40 dB or higher significantly increases the risk of permanent hearing damage. PTS occurs when the baseline threshold at which an animal can hear noise is permanently elevated. Furthermore, regular, repeated exposure to levels of noise which causes TTS may result in PTS. For a more in-depth discussion on TTS and PTS see the relevant sections of both National Marine Fisheries Service (2016) and Tougaard and Mikaelsen (2018).

Lucke *et al.* (2009) demonstrated that TTS could be induced in a harbour porpoise from a single broadband acoustic pulse at a receieved, unweighted SEL of 154 dB re 1 $\mu Pa^2 s^{-1}$. This value has since informed regulations surrounding acceptable noise levels during wind farm construction e.g., in Germany impulsive sound levels recorded up to 750 m from the sound source must not exceed 160 dB re 1 $\mu Pa^2 s^{-1}$ SEL for greater than 5% of piling time (SEL_{05}) (German Federal Ministry for the Environment and Nuclear Safety, 2013). However, the most recent Irish guidelines (National Parks & Wildlife Service, 2014) still utilise the higher values (183 dB re 1 $\mu Pa^2 s^{-1}$) first described by Southall *et al.* (2007) as the recommended limit for single strike noise emissions. **Table 2.2** displays these criteria for marine mammal groupings for single pulse, multiple pulse, and non-pulse (i.e. broadband) sound sources. It should be noted that these impulsive or non-impulsive noise criteria are applied based on signal features at the sound source. This is important as the recent update to this extensive review, Southall *et al.* (2019), instead applies the respective noise exposure criteria based on signal features likely to be received by animals.

In the review by Southall *et al.* (2019) scientific recommendations have been revised, including the threshold levels at which marine mammals experience disturbance. These recommendations follow the methodology set out by Finneran (2015) and adopted by the National Marine Fisheries Service (2016). It is likely that these revised thresholds will eventually be incorporated into the Irish guidelines.

Table 2.3 summarises the updated TTS and PTS thresholds for *non-impulsive* noise. Dual criterion for this noise type has been removed in favour of using only the single frequency weighted SEL. The low energy nature of these noise types, which include sources such as vessel engine noise and OWT noise, ensures that there are virtually no scenarios for which the SPL threshold would be breached prior to the SEL threshold (Southall et al., 2019). The increase in research in this field since 2007 has led to more accurate thresholds being derived. For example, whereas the earlier thresholds were applied to all pinnipeds, there have now been thresholds developed specifically for phocids, or the true seals. In addition, ranges have been replaced by specified values.

Table 2.3: TTS- and PTS-onset thresholds for marine mammals exposed to **non-impulsive** noise. Adapted from Southall *et al.* (2019)

Marine mammal hearing groups		TTS onset: SEL (weighted)	PTS onset: SEL (weighted)
Cetaceans	*Low frequency*	179	199
	Mid frequency	178	198
	High frequency	153	173
Phocids	*Water*	181	201
	Air	134	154

* Units of measurement:
Sound Exposure Level, SEL (in water): measured in dB re 1 $\mu Pa^2 s^{-1}$
Sound Exposure Level, SEL (in air): measured in dB re $(20\ \mu Pa)^2 s^{-1}$

Similarly, **Table 2.4** summarises the onset thresholds for TTS and PTS for marine mammals exposed to *impulsive* noise sources. While not specific to wind farm activities, Mooney *et al.* (2009) were able to demonstrate TTS onset in bottlenose dolphins across a range of exposure durations and sound levels, with the results indicating that shorter duration exposures require higher sound energy levels to induce TTS than longer duration exposures. They suggest that this is likely true for many other odontocetes. For this reason, it may be important for any pile-driving activity to factor in regular pauses to reduce the likelihood of negative impact on marine mammals. Indeed, this is one of the recommendations Russell *et al.* (2016) make in a study on harbour seal avoidance of wind farms. Breaks in piling should be of sufficient duration to reduce the overall SEL_{cum} animals are exposed to.

Table 2.4: TTS- and PTS-onset thresholds for marine mammals exposed to **impulsive** noise: SEL thresholds in dB re 1 µPa^2 s^{-1} under water and dB re (20 µPa)^2 s^{-1} in air; and peak SPL thresholds in dB re 1 µPa under water and dB re 20 µPa in air. Adapted from Southall *et al.* (2019)

Marine mammal hearing groups		TTS onset: SEL (weighted)	TTS onset: Peak SPL (unweighted)	PTS onset: SEL (weighted)	PTS onset: Peak SPL (unweighted)
Cetaceans	*Low frequency*	168	213	183	219
	Mid frequency	170	224	185	230
	High frequency	140	196	155	202
Phocids	*Water*	170	212	185	218
	Air	123	138	138	144

* Units of measurement:
Sound Pressure Level, SPL (in water): measured in dB re 1 µPa (peak) (flat)
Sound Exposure Level, SEL (in water): measured in dB re 1 µPa^2 s^{-1}
Sound Pressure Level, SPL (in air): measured in dB re 20 µPa (peak) (flat)
Sound Exposure Level, SEL (in air): measured in dB re 20 µPa^2 s^{-1}

Harbour seals exposed to pile-driving noise in controlled experiments by Kastelein *et al.* (2018) display onset of TTS at SEL_{cum} of around 192 dB re 1 µPa^2 s^{-1}. Hastie *et al.* (2015) demonstrated, using tags and sound propagation models, that half the seals tagged were exposed to SELs during pile-driving activity at an OWF which exceeded the PTS threshold levels described by Southall *et al.* (2007). This clearly illustrates the potential for population level impacts from OWF construction activities should appropriate siting and mitigation measures not be utilised.

Site-specific sound propagation models should be used in order to effectively determine the distance at which TTS onset is likely to begin (National Research Council, 2003). Several factors will influence this range, including the amount of energy required to pile into the seabed type present, the local bathymetry and associated topography, and the local oceanographic conditions, such as temperature, salinity, currents, and fronts (Sutton et al., 2013). In addition, the influence of any dampening devices, or noise mitigation systems, should be factored in.

2.5.2 Fish and shellfish

The inner ear of fishes are lined by many sensory hair cells that are similar to those found in mammals (Popper and Hawkins, 2019). Exposure to short, intense sounds such as those from pile-driving or to longer exposure to less intense noise have been observed to damage these cells and thus impair the ability of fishes to hear (Casper et al., 2013b; Smith et al., 2011). This may lead to a temporary hearing loss (TTS). However, there is no evidence of permanent hearing loss (Popper and Hawkins, 2019); due to their ability to replace or repair cells, it is likely that any hearing loss will be temporary. Laboratory studies have shown that recovery can occur from minutes to days following the cessation of the noise source (Casper et al., 2013a). During a period of reduced hearing ability, however, fish may become more susceptible to predation, be unable to communicate effectively, and have difficulty in assessing their environment (Slabbekoorn et al., 2010; Popper and Hawkins, 2019). Popper *et al.*, 2014b developed interim criteria to be used as guidelines concerning noise threshold levels at which fish may be impacted during pile driving events. These were modified in Popper and Hawkins, 2019 (**Table 2.5**).

Table 2.5: Proposed interim criteria for mortality and recoverable injury from exposure to pile driving signals are based on 960 sound events at 1.2 s intervals (Halvorsen et al., 2012c; b). Temporary threshold shift (TTS) based on Popper *et al.* 2005. The same SPL_{peak} levels are used both for mortality and recoverable injury since the same sound exposure level (SEL_{ss}) was used throughout the pile driving studies. All criteria are presented as sound pressure even for fishes without swim bladders since no data for particle motion exist. Relative risk (high, moderate, low) is given for animals at three distances from the source defined in relative terms: N, near; I, intermediate; F, far (from Popper *et al.*, 2014). Table adapted from Popper & Hawkins 2019.

Receptor	Mortality/ potentially mortal injury	Impairment			Behaviour
		Recoverable	*TTS*	*Masking*	
Fish: no swim bladder (particle motion detection)	> 219 dB SEL_{cum} > 213 dB SPL_{peak}	> 216 dB SEL_{cum} > 213 dB SPL_{peak}	> 186 dB SEL_{cum}	(N) Mod (I) Low (F) Low	(N) High (I) Mod (F) Low
Fish: swim bladder is not involved in hearing (particle motion detection)	210 dB SEL_{cum} > 207 dB SPL_{peak}	203 dB SEL_{cum} > 207 dB peak	> 186 dB SEL_{cum}	(N) Moderate (I) Low (F) Low	(N) High (I) Moderate (F) Low
Fish: swim bladder is involved in hearing (primarily pressure detection)	207 dB SEL_{cum} > 207 dB SPL_{peak}	203 dB SEL_{cum} > 207 dB SPL_{peak}	186 dB SEL_{cum}	(N) High (I) High (F) Mod	(N) High (I) High (F) Mod
Eggs and larvae	> 210 dB SEL_{cum} > 207 dB SPL_{peak}	(N) Moderate (I) Low (F) Low	(N) Moderate (I) Low (F) Low	(N) Mod (I) Low (F) Low	(N) Mod (I) Low (F) Low

*Units: SPL_{peak} and SPL_{RMS} = dB re 1 µPa; SEL = dB re 1 µPa² s⁻¹.

2.6 Masking

Masking refers to the disruption of auditory signals and communication due to increased levels of sound above the ambient background noise, and may be the most pervasive impact from noise pollution in the marine environment (Hawkins and Popper, 2018; Popper and Hawkins, 2019; Dooling, Leek and Popper, 2015; Peng, Zhao and Liu, 2015; Pine et al., 2016; Cholewiak et al., 2018). Sources of noise may be short-term and impulsive, or low-level and continuous, e.g. vessel noise and operational wind turbines. **Figure 2.3** illustrates the different pathways along which noise from an OWT may travel and potentially disturb fish and other marine life.

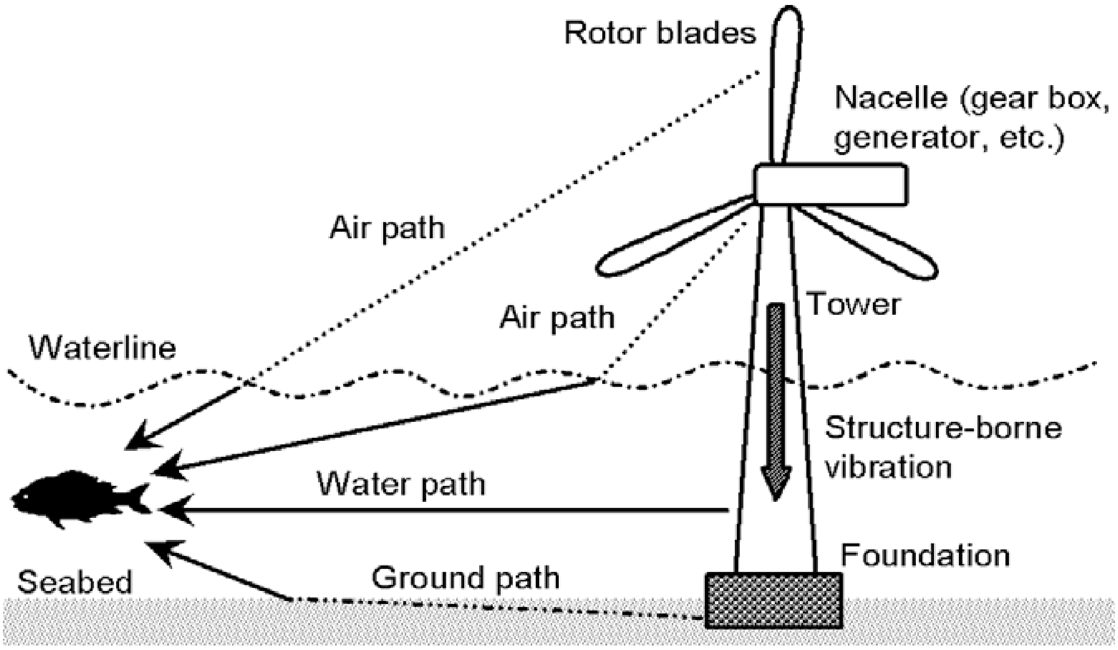

Figure 2.3: Transmission pathways for noise from OWTs (Kikuchi, 2010).

2.6.1 *Marine mammals*

Auditory masking occurs when the "the ability to detect or recognise a sound of interest is degraded by the presence of another sound" (Erbe et al., 2016). It is likely that temporary auditory masking of biologically important acoustic signals for marine mammals will occur during pile driving, and possibly due to ship noise during construction. Pinnipeds may experience very localised masking around an OWT.

Sources of operational noise at wind farms may include vibration noise from turbines, vessel traffic, and maintenance activities (Bailey, Brookes and Thompson, 2014; Bergström et al., 2014; Nabe-Nielsen et al., 2018; Hammar, Perry and Gullström, 2016; Schuster, Bulling and Köppel, 2015). Noise from operational turbines primarily derives from the movement of wings and gears. The sound frequencies of these noises are too low to be heard by cetaceans such as harbour porpoise and bottlenose dolphins, but do overlap with the hearing range of phocid seals such as harbour seals and grey seals (Tougaard and Mikaelsen, 2018; Tougaard, Wright and Madsen, 2015; Tougaard and Beedholm, 2019). It is therefore possible that chronic low levels of noise associated with wind turbines and vessel traffic from shipping, fishing, and maintenance vessels may result in auditory masking for species which hear at low frequencies. However, persistent noise pollution from OWF must be put into the context of the local ambient soundscape. If the level of noise emissions from OWF do not exceed existing background noise levels, they are unlikely to cause further disturbance to marine mammals. To date, no studies have demonstrated such a negative impact from operational wind farms on cetaceans (Bailey, Brookes and Thompson, 2014; Lovich and Ennen, 2013; Schuster, Bulling and Köppel, 2015; Nabe-Nielsen et al., 2018), while only two studies have indicated possible but very limited auditory masking on harbour seals at short distances (see Schuster, Bulling and Köppel, 2015).

2.6.2 *Fish and shellfish*

Masking may lead to interference with fish behaviours such as foraging and reproduction, predator/prey avoidance/detection, and the masking of auditory cues that aid in navigation and orientation (Popper and Hawkins, 2019). Limited evidence suggests that some fish will alter their own vocalizations to counteract the effect of masking (Radford, Kerridge and Simpson, 2014), and this may lead to a reduction in their overall condition.

De Jong *et al.* (2018) found that gobies exposed to noisier environments led to reduced communication during courtship and spawning success, and suggest that reproductive success may be sensitive to the masking effects of noise. Thomsen *et al.* (2008) suggested that Atlantic cod and herring (both having swim bladders) would be able to detect pile driving noise from 80 km distant, and masking of auditory signals may occur within 4 km of an operational turbine. They also found that dab and Atlantic salmon (*Salmo salar*) (both species detect sound via particle motion only) would be able to detect pile-driving noise from an unspecified but considerable distance, and that masking may occur within 1 km of an operational turbine in these species. However, Wahlberg and Westerberg (2005) note that noise from an operational wind turbine may be detected out to 25 km distance during wind speeds of 23 ms^{-1}, highlighting the importance of local conditions on noise propagation. More recent studies on fish which detect sound through particle motion suggest that masking effects do not extend beyond 10 m from a wind turbine foundation (Andersson, 2011; Sigray and Andersson, 2011). It is worth noting that the level of wind turbine noise increases with the size of the turbine and foundation. Recent wind farm installations have tended towards using larger turbines with subsequently larger foundations (Higgins and Foley, 2014; Igwemezie, Mehmanparast and Kolios, 2019), which may result in a larger radius of impact.

Whitfield and Becker (2014) reviewed the available literature on impacts from recreational motorboats on fishes and found a consensus for the probability of masking effects on fish communication by engine noise. However, effects were found to vary between species and between sizes within species. This demonstrates the importance of considering not only interspecific variation but also intraspecific variations in susceptibility to noise disturbances when assessing any potential impacts from anthropogenic noise pollution.

2.7 Behavioural response

2.7.1 Marine mammals

Behavioural responses to acoustic disturbance are more likely to occur during pile-driving activity, although ship noise may also have some impact (Oakley, Williams and Thomas, 2017; Merchant et al., 2014; Pirotta et al., 2015). The most obvious response is an abrupt escape behaviour. Animals swim rapidly away from the source of noise at an accelerated pace (Kastelein, Van de Voorde and Jennings, 2018). However, disruption to foraging and social behaviours may also occur.

A number of studies have shown that harbour porpoise leave the vicinity of construction activities during pile-driving events (Brandt et al., 2011; Carstensen, Henriksen and Teilmann, 2006; Graham et al., 2017; Dähne et al., 2014, 2017; Kastelein, Van de Voorde and Jennings, 2018; Vallejo et al., 2017; Dähne et al., 2013; Teilmann and Carstensen, 2012). Displacement of up to 22 km has been recorded in some cases (Brandt et al., 2011). In all studies bar one (Teilmann and Carstensen, 2012), harbour porpoises quickly returned to the habitat, with baseline levels of occurrence observed usually no later than 48 hours, and often as short as a 6 hours after the cessation of pile driving. The slow return of harbour porpoise to numbers recorded during baseline surveys at the Nysted Offshore Wind Farm has yet to be sufficiently explained. However, there has been a gradual increase in numbers in the 10 years following the establishment of the windfarm. This increase may be due to habituation of the porpoises to the wind farm or to enrichment of the site following reduced fisheries and an artificial reef effect (Teilmann and Carstensen, 2012). At the Egmond aan Zee project, porpoise acoustic activity following construction exceeded that which was recorded during baseline surveys (Scheidat et al., 2011). Again, while there is no evidence to attribute this increase to any single factor it is believed that habitat enrichment, increased foraging opportunities, and the quiet soundscape found within the OWF (due to reduced vessel activity within the OWF) may all contribute.

Brandt *et al.* (2018) investigated disturbance of harbour porpoise during construction of the first seven offshore wind farms in Germany. Six of these projects employed noise mitigation systems (see **Table 6.1**), primarily of the big bubble curtain type, during construction. They found that declines in harbour porpoise detections occurred at SEL_{05} exceeding 143 dB re 1 $\mu Pa^2 s^{-1}$, at distances up to 17 km away from piling events. In addition, declines occurred more abruptly during unmitigated piling events. The SEL which provoked this behavioural response is a more specific estimate than the SEL range (139 – 152 dB re 1 $\mu Pa^2 s^{-1}$) given by Dähne *et al.* (2013), and thus may provide better guidance to the degree of noise level which provokes an initial disturbance response. Both studies recorded the return of harbour porpoises within several hours post cessation of piling events.

Merchant *et al.*, (2014) and Pirotta *et al.*, (2015) demonstrated how both boat noise and presence disrupted foraging activity levels of bottlenose dolphins by up to 49%, which may have a significant impact on energy balance. Frequent disruptions to foraging and or social behaviours such as mating may be detrimental to long-term health and survival, and could have implications at the population level (Bailey et al., 2010; Bailey, Brookes and Thompson, 2014; Thompson et al., 2010; Nabe-Nielsen et al., 2018). Increased levels of vessel traffic during construction phases of OWF development thus need to be accounted for in any impact assessment. Conversely, the likely drop off in vessel activity in any exclusion zone surrounding an operational OWF facility may be of benefit to local marine mammal populations (Lindeboom et al., 2011; James, 2013).

There have been similar results from studies on the impacts of pile-driving on harbour seals. Russell *et al.* (2016) described a significant reduction (19 – 83 %) in habitat usage up to 25 km from piling activity, with displacement beginning at predicted received SPLs of between 166 – 178 dB re 1 µPa. Despite this, two hours after the cessation of piling activity harbour seal occurrence had returned to that observed pre-pile driving. However, notwithstanding the use of a soft-start to piling activities, Hastie *et al.* (2015) predicted almost half of the seals monitored in their telemetry study were exposed to sound levels above the threshold at which permanent auditory damage may occur. This suggests that a soft start alone may not give seals sufficient time to vacate the area of piling activity. There is far less information available on grey seals, but their similar physiology and foraging activities would suggest they would behave in a similar manner to harbour seals. In a review of impact monitoring and mitigation at offshore wind farms Verfuss *et al.* (2016) noted that harbour seal numbers took two years to recover in the UK. However, during the same period at the same location, grey seal numbers increased year on year with no initial drop off during construction, and the authors suggest this increase in grey seal abundance may have had a more significant influence on the reduction in harbour seal numbers than the construction activities.

The return of marine mammals to habitat following the cessation of pile driving implies that either operational noise levels are of no or little impact, or that the conditions found within wind farms are more favourable (e.g. less exposure to shipping or fishing activities). More studies are required to confirm or reject either hypothesis.

2.7.2 Fish and shellfish

Popper, Hawkins and Halvorsen (2019) note that, in terms of harmful impacts on fish, assessing changes to behaviour as a response to noise pollution is more relevant than assessing injuries. Due to the lower sound level required to elicit behavioural responses it is likely that such responses will occur earlier, within a greater portion of the population exposed to the noise source, and over a much greater area. It has been generally assumed that fishes with greater hearing sensitivity are more likely to display behavioural responses to anthropogenic sound than species with less sensitive hearing abilities (Popper and Hawkins, 2019). However, some studies have shown this is not always the case. The responses of two species of differing hearing sensitivities were compared in a lab environment (Shafiei Sabet et al., 2016), with results showing that exposure to anthropogenic noise caused both similar and species-specific responses, which were not obviously related to their hearing abilities. This highlights the need for further research at the species level to better elucidate which species are more vulnerable to behavioural disturbance and any intra-specific differences in response. Such information is significant when determining the degree of environmental impact.

Behaviours such as foraging, predator avoidance, migration, and spawning may all be impacted by increased sound levels in fish and shellfish (Popper and Hastings, 2009a; Popper, Hawkins and Halvorsen, 2019; Hawkins and Popper, 2018; Cox et al., 2018). Schooling behaviour of certain species may also change in response to impulsive sound sources (Popper and Hawkins, 2019). The behaviour of sprat (*Sprattus sprattus*) and Atlantic mackerel (*Scomber scombrus*) was observed using sonar/echo sounder when exposed to playback of simulated pile driving noise (Hawkins, Roberts and Cheesman, 2014), with behavioural responses found to increase with increasing sound level. Sprat, which are very sensitive to sound pressure, were observed to disperse or change schooling density, while mackerel, which likely use only particle motion to detect noise, were observed to change depth with some scattering recorded. The results suggest that these behavioural changes may occur many kilometres from the sound source. It is also worth noting sprat schools break up at night, and there was no measurable response from individual fish when exposed to playbacks.

Studies assessing the behavioural response of European seabass (*Dicentrarchus labrax*) to playback of impulsive and continuous sound found that pile-driving noise can result in longer-lasting behavioural effects than those observed when exposed to continuous sound sources such as shipping noise or noise emitted by turbines in operational wind farms (Neo et al., 2014). The type and intensity of the response may be influenced by the amplitude and pulse rate intervals of the sound (Neo et al., 2015). Seabass held in an outdoor floating pen displayed no difference in behavioural response to playback of mitigated (using ramp-up procedures) and unmitigated pile driving activities (Neo et al., 2016). Neo *et al.* (2018) observed seabass responding more strongly to impulsive noises at night, but also detected a decrease in response over time. The study suggested this may be due to either habituation to the noise after repeated exposure, or due to a temporary threshold shift in their hearing levels. Kastelein *et al.* (2017, 2015) used playback of pile driving sounds to European seabass to determine acoustic-dose relationships and found that smaller fish were more acoustically sensitive than larger fish. The study concluded that if seabass were exposed to similar sounds levels in the wild, there would be limited adverse effects on their ecology due to the short-lived nature of their response. However, it should be noted that these latter studies were conducted in a shallow pool with unnatural acoustics and on fish which had been reared in captivity, and so may not accurately reflect the behaviour of wild population of the same species in the field. Juvenile, laboratory-bred European seabass exposed to playbacks of pile driving and seismic sounds in a laboratory based study (Radford et al., 2016) displayed increased ventilation rates in contrast to exposure to a continuous noise source (playbacks of ship noise). Fish exposed to piling/seismic playbacks for 12 weeks no longer responded with increased ventilation, indicating either habituation or a temporary threshold shift caused by repeated exposure to the sound source. Similarly, Spiga, Aldred and Caldwell, (2017) exposed juvenile European seabass to recordings of both pile-driving and of drilling noises in a confined, shallow tank. They observed a startle response in the seabass at onset of piling but no immediate response during playback of either drilling or ambient noise. In addition, an increase in ventilation rate and reduced anti-predator response was observed during playback of piling and drilling noise.

A study using acoustic telemetry during pile driving within a harbour environment (Iafrate et al., 2016) found no effects on the residency of sheepshead (*Archosargus probatocephalus*), but grey snapper (*Lutjanus griseus*) were more likely to depart the area during the onset of pile driving. The study concluded that snapper may be more easily displaced by sound pollution and sheepshead were more at risk to behavioural disturbance due to their greater site fidelity.

Mueller-Blenkle *et al.*, (2008) observed significant avoidance behaviour in Atlantic cod to low frequency sound resembling wind turbine noise emissions. The authors suggest that this behaviour is not likely to lead to permanent avoidance of the area, particularly if the habitat is attractive to cod. This is supported by findings assessing the impact of artificial structures on the aggregation, abundance, and foraging behaviour of cod (see Hammar, Wikström and Molander, 2014; Reubens *et al.*, 2014, 2013b; a; Reubens, Degraer and Vincx, 2014). Atlantic cod and common sole exposed to playback of pile driving noise while held in two large (40 m) net pens (Mueller-Blenkle et al., 2010) found significant movement away from the sound source in both species at relatively low received sound pressure levels (sole: 144 – 156 dB re 1 μPa_{Peak}; cod: 140 – 161 dB re 1 μPa_{Peak}, particle motion between $6.51x10^{-3}$ and $8.62x10^{-4}$ ms^{-2} peak). However, there was high variability in behavioural responses across individuals, and a decrease in response behaviours following multiple exposures.

A baited underwater video (BRUV) study (Roberts, Pérez-Domínguez and Elliott, 2016) recorded startle and directional avoidance responses during playing back of impulsive sounds with a received level of sound pressure level 163 – 167 dB re 1 µPa (peak-to-peak) in two-spotted goby (*Gobiusculus flavescens)*, pollack (*Pollachius pollachius)*, and thicklip grey mullet (*Chelon labrosus*). Playback of shipping noise of received SPL_{RMS} 142.7 dB re 1 µPa was sufficient to startle thicklip grey mullet repeatedly. However, the study also noted that the playback noise, while accurately reflecting the water borne component of pile-driving could not replicate the substrate-borne vibrations that pile-driving produces. Purser and Radford, (2011) showed no significant impact on the total amount of food eaten, but increased food handling errors and reduced discrimination between food and non-food items at exposure to increasing levels of sound. Lower survival of larval fish on coral reefs exposed to playback of motorboat noise has also been observed (Nedelec et al., 2017). In this instance, parental behaviour is modified, with more time spent in defensive behaviour and less spent on feeding and interacting with the larvae by the male parent. Engine noise exposure may also affect the feeding behaviour of fish, with roach (*Rutilus rutilus*) and perch (*Perca fluviatilis*) showing species-specific responses, with habituation and the presence of other species likely to modify the effects (Magnhagen, Johansson and Sigray, 2017).

The density of crustaceans' bodies is nearly the same as that of water, and they lack a gas-filled chamber. Therefore, it is likely that crustaceans rely solely on particle motion to detect sound (Breithaupt, 2002; Breithaupt and Tautz, 1988, 1990; Popper, Salmon and Horch, 2001; Monteclaro, Anraku and Matsuoka, 2010; Goodall, 1988; Heinisch and Wiese, 1987). Crustaceans may also detect vibrations that are waterborne e.g., Norwegian lobster (Popper, Salmon and Horch, 2001) or felt through the sediment, e.g., the European hermit crab (*Pagurus bernhardus*) (Roberts et al., 2016). Tidau and Briffa (2016) presented a review of behavioural responses to anthropogenic noise among crustaceans finding evidence of behavioural impacts associated with seismic surveys, pile-driving, vessel noise, and white noise. American lobsters (*H. americanus*) have an increased feeding rate following exposure to seismic airgun noise, a response similar to that observed in humans following brain trauma (Payne et al., 2008). Commercial catch rates of rock lobster (no species noted) did not differ before and after seismic surveys (Parry and Gason, 2006), however, there was a high degree spatial separation between fishing areas and seismic survey areas, which likely reduced the statistical robustness of the study. Similarly, catch rates of three shrimp species 12-36 hours post exposure to seismic surveys identified no significant decreases, nor were there significant changes to density observed (Andriguetto-Filho et al., 2005). However, it is has previously been noted that catch rates of invertebrates following explosions are difficult to interpret as crustacean species may be attracted to scavenge on dead or injured animals.

N. norvegicus were observed to burrow less deeply, flush their burrows less regularly, and be considerably less active when exposed to pile driving noise (Solan et al., 2016). Evidence for impacts of chronic, continuous noise, i.e. shipping related noise, has also been found on *N. norvegicus* with repression of burying, bioregulation, and locomotory behaviour recorded in one study (Solan et al., 2016). This disruption could have wider ecosystem implications due to the important role this species plays in mixing upper sediment layers which prevents the suspension of sediment materials. Pile driving was also shown to alter the chorus of three snapping shrimp species (*Athanas nitescens, Alpheus macrocheles,* and *Alpheus glaber*) (Spiga, 2016), resulting in an increase in the number and amplitude of acoustic signals during the highest levels noise playback.

Kastelein (2008) exposed the common cockle (*Cardium edule)* to sediment-borne vibrations, increasing the frequency and amplitude until the cockles retracted their siphons and closed their shells. If vibrations from anthropogenic noise sources occur frequently, this could limit the time available to cockles to feed, with negative consequences for growth and reproduction.

It is clear that behavioural responses to anthropogenic noise associated with construction and operation of OWF can result in reduction of overall survival rate of some species. Should noise be chronic and pervasive, it is possible that the impact at an individual level could lead to population level consequences. Therefore, it will be necessary to survey any potential wind farm sites to ascertain the community assemblages present and use appropriate mitigation measures based on noise propagation models (Hawkins and Popper, 2017) developed for both the physical characteristics of that site and the hearing abilities of the fishes present.

2.8 Physiological stress

Stress may be defined as any state of biological strain or tension resulting from adverse circumstances (Popper and Hawkins, 2019). Hormonal, autonomic, immune, and behavioural responses may initially allow animals to adapt to adverse conditions. However, some stressors may change the state of physiological processes and affect homeostasis, thus having an adverse effect on the animals' health and well-being. Such consequences can include increased susceptibility to disease, reduced fecundity, decreased longevity, and physiological dysfunction (Houser et al., 2016; Tennessen, Parks and Langkilde, 2016).

2.8.1 Marine mammals

Acoustic disturbance may induce a physiological stress response in marine mammals. Such responses include elevated respiration or the increase in the stress related hormone cortisol (Houser et al., 2016). However, such responses are very difficult to measure in the wild (Cato et al., 2016), and experiments conducted on animals in captivity risk conflating the observed stress response of the measured effect with the stress response of being confined (Marino et al., 2019). As a consequence, research in this field is limited, and more so when restricting the search to noise-related stress responses.

Kastelein, Van de Voorde and Jennings (2018) observed increased respiration in a captive harbour porpoise during playback of audio recordings of pile driving noise. Increased respiration is a visible stress response and leads to a greater consumption of energy. However, other biological stress responses are likely to occur such as the expression of cortisol, with long-term exposure to noise possibly resulting in detrimental impacts. However, in the wild this animal would likely have been able to flee the vicinity of the noise source, and thus lessen the potential impact. Wright, Deak and Parsons (2011) suggest that both deep-diving and coastal dwelling marine mammals may be particularly vulnerable to chronic stress induced by long-term or frequent exposure to noise such as that from whale watching vessels, shipping, seismic surveys, and sonar. If the chronic stress responses observed in humans are replicated marine mammals, there would significant consequences for populations, and may partly explain why certain species have not recovered following the introduction of management measures such marine reserves.

2.8.2 Fish and shellfish

There are concerns regarding the potential for stress-related impacts from noise on fish (Slabbekoorn et al., 2010; Popper and Hawkins, 2019). Studies of stress on other vertebrates have clearly shown the detrimental effect that noise, particularly impulsive noise, has on their well-being (Slabbekoorn et al., 2018). However, there is little information published on noise-related stress responses in fish. To date, research has focused on continuous noise, with very little published on stress from impulsive noise sources (Popper and Hawkins, 2019). Nearly all studies have been on species in confinement, under variable exposure conditions, and responses have varied greatly between species and treatments. Thus, while broadening our general understanding of noise-related stress responses, the applicability of these studies to understanding population-level impacts in the wild is limited.

Atlantic cod exposed to a linear sweep of artificial noise from 100 – 1000 Hz, similar to that which could be emitted by an operational wind turbine (Kikuchi, 2010) and commonly emitted by vessel traffic (Ross, 2005), showed a mild cortisol (stress hormone) increase which was temporary but clearly related to the acoustic emission (Sierra-Flores et al., 2015). Plasma levels of cortisol returned to normal < 1 hr post exposure, but under long-term, daily exposure to this noise source, the brood stock of cod incurred a significant reduction in total egg production and fertilisation rates, which resulted in a 50% reduction in total viable embryos produced. As noted in previous experiments on fish, fish in tanks were unable to flee the noise source, and thus results may not reflect the outcome of a similar exposure in a wild population. European seabass and gilthead sea bream (*Sparus aurata*) exposed to similar levels of artificial noise as above showed a significant increase in motility as well as an increase in lactate and haematocrit levels (Buscaino et al., 2010). In addition, acoustic stimulus produced intense muscle activity, requiring an increase in energetic output, and potentially reducing the fitness of the animal. If animals are exposed to noise during periods when their natural levels of energy are depleted, e.g. post spawning, this could lead to a greater risk of predation or susceptibility to disease.

Playback of pile-driving noise elicited stress related responses in European seabass, black sea bream and European plaice. Seabass in both laboratory and open water conditions had an increased ventilation rate, with a rapid recovery following the cessation of the sound emissions (Bruintjes et al., 2016a). Black sea bream were observed to increase their oxygen uptake during pile driving when compared with ambient noise levels. Plaice, however, displayed no increase in oxygen uptake during the exposure to the same level of pile driving noise (Bruintjes et al., 2016b). This illustrates a potentially greater physiological impact on sea bream and seabass, both of which have gas filled swim bladders capable of detecting sound pressure, than on plaice, which uses particle motion only to detect noise. Debusschere *et al.*, (2016) exposed juvenile European seabass to pile-driving noise as close as 45 m from the emitted sound source. Their results showed significant reductions in secondary stress responses of oxygen consumption rate and whole-body lactate concentrations following exposure to pile driving noise. No tertiary stress responses were observed up to 30 days post exposure. However, a number of limitations should be noted. Firstly, fish were held within 500 mL containers during exposure making them unable to flee the noise. Secondly, fish were submerged at 2.5 m below the surface, much shallower than their natural depth. Finally, their proximity to the surface and the emitted sound source increases the likelihood that particle motion would have played a large role in the perception of sound, but this was not assessed.

Several studies assessing exposure to long term, continuous noise among European eels (*Anguilla anguilla*) have been carried out. While healthy juvenile eels displayed no response to playbacks of ship noise, poor condition juveniles displayed increased ventilation rates and delayed response to a predator mimic in the presence of ship noise (Purser et al., 2016). These results were replicated by Bruintjes *et al.* (2016a) who also observed a rapid recovery in startle response, and noramilisation of ventilation rates 2 minutes post cessation of the emitted noise source, with a similar response and recovery observed for European seabass. In a further study, juvenile European eels exposed to playbacks of passing ship noise were 50% less likely and 25% slower to respond to an 'ambush predator' than eels exposed to ambient noise levels, suggesting a compromised antipredator behaviour (Simpson, Purser and Radford, 2015).

Nichols, Anderson and Širović (2015) found increased levels of cortisol expressed among juvenile giant kelpfish (*Heterostichus rostratus*) exposed to intermittent playback of engine noise, but no significant changes when exposed to either continuous playback or ambient noise. Nedelec *et al.* (2016) observed an increase in tolerance to the noise levels following regular exposure to the playbacks, with a decrease in hiding behaviour after two days exposure while ventilation rate increases were diminished following two weeks of exposure in coral reef fish. Following 3 weeks of exposure, blood plasma cortisol levels were equivalent to those recorded during baseline assessments, providing the first field-based evidence for a tolerance to anthropogenic noise.

A review of reports on crustacean sensitivity to high amplitude noise by Edmonds *et al.* (2016) notes the occurrence of some physiological responses. American lobsters (*H. americanus*) exposed to airgun noise of 202-227 dB re 1 µPa (peak-to-peak) observed changes to feeding levels, serum biochemistry and hepatopancreatic cells of animals exposed for months when compared to controls. Ship playback noise was found to affect the metabolic rate among shore crabs (*Carcinus maenas*) with subjects consuming 67% more oxygen in comparison with playback of general harbour noise. In addition, individuals exposed to playback of ship noise were more likely to suspend feeding behaviour and took longer to return to shelter. The review suggests that underwater noise can influence the physiological regulatory mechanisms that control larval growth within in crustaceans. More research is required to confirm whether such responses also occur among commercially important shellfish species in UK and Irish waters.

In molluscs, Spiga, Caldwell and Bruintjes (2016) observed an increased clearance rate in the mussel (*Mytilus edulis)* exposed to playback of pile-driving noise, suggesting that mussels move from a 'maintenance' state to active metabolism as a stress response to noise emissions.

3 Built structures

Figure 3.1: As offshore wind farms begin to provide an ever-increasing share of our energy needs so too will their physical footprint expand in the marine environment. Offshore wind turbines near the German island of Amrum © Morris MacMatzen / Reuters

3.1 Summary

3.1.1 Marine mammals

The increasing physical footprint of humanity within the oceans has been termed "ocean sprawl" (Duarte et al., 2013; Sanabria-Fernandez et al., 2018). This sprawl consists of built structures and related infrastructure, such as cabling and mooring lines. Offshore windfarms are a growing contributor of this, potentially occupying valuable habitat for marine life. Ocean sprawl may cause spatial or temporal displacement from important foraging or breeding habitats, or disrupt migratory routes. Conversely, introduction of new hard substrate may benefit marine mammals by acting as an artificial reef, enriching the local ecosystem and providing a source of food. In addition, vessel exclusion zones surrounding OWF may become a quiet space, giving marine mammals shelter from vessel traffic, and other source of noise pollution as well as some fishing activities.

3.1.2 Fish and shellfish

Built structures within the marine environment supply additional hard substrate upon which organisms may settle and may be classified as artificial reefs (Bergström et al., 2014; Hammar, Perry and Gullström, 2016; Ashley, Mangi and Rodwell, 2014; Langhamer, 2012). Artificial reefs can potentially provide a wide range of benefits including stock enhancement, species conservation, juvenile nurseries and spawning ground habitats, and as a means to prevent bottom trawling and associated habitat destruction. However, their physical footprint on the seabed may also remove benthic habitat and disrupt existing hydrographic processes, or create new ones (Langhamer, Wilhelmsson and Engström, 2009; Langhamer, 2012). Changes in flow may result in alterations to the distribution of nutrients and associated biota (Floeter et al., 2017). These changes may prove beneficial to marine life, chiefly through the provision of shelter from currents and predation, and greater foraging opportunities.

The increase in artificial structures may also attract reef-forming species (Langhamer, 2012; Langhamer and Wilhelmsson, 2009; Hammar, Perry and Gullström, 2016; Hammar et al., 2017). Furthermore, an OWF with a fisheries exclusion zone may become *de facto* Marine Protected Areas (Ashley, Mangi and Rodwell, 2014; Coates et al., 2016), resulting in increased abundances and greater diversity of associated communities. The degree of any such benefit will vary, and depend greatly on the location, and upon the level of any imposed fishing restrictions (Hammar, Perry and Gullström, 2016). However, the opportunity to colonize introduced hard substrate may also benefit invasive species and enable them to increase their distribution into new habitats.

3.2 Habitat displacement

3.2.1 Marine mammals

The majority of studies have not reported a significant change to the abundance of marine mammals following the establishment of an OWF (Schuster, Bulling and Köppel, 2015; Hammar, 2015; Wingfield et al., 2017). To date, all studies bar one have recorded the return of target species to the habitat at baseline levels, following the cessation of pile-driving activities (Scheidat et al., 2011; Bailey et al., 2010; Tougaard et al., 2009; Brandt et al., 2018; Nabe-Nielsen et al., 2014). This would suggest that there is no acute negative impact from the presence of built structures, or that the habitat enrichment and subsequent greater foraging opportunities they provide are incentive enough for harbour porpoises to endure any negative impacts which may exist. Teilmann & Carstensen (2012) observed the slow return of harbour porpoise to Nysted Offshore Wind Farm in the Danish western Baltic Sea. A significant reduction in porpoise echolocation activity from baseline levels in 2001-02 was observed, and while a gradual increase in activity has occurred since construction ceased, it has only reached 29 % of baseline levels 10 years post-construction.

A large-scale array can have a substantial footprint. Therefore, it is vital that any OWF be sited in habitat that is not critical for marine mammals, i.e. it does not form part of their breeding or calving grounds or is not their primary foraging habitat.

3.2.2 Fish and shellfish

No studies to date have shown a significant displacement of fish or shellfish from the addition of hard substrate (Stenberg et al., 2015; Vaissière et al., 2014; Methratta and Dardick, 2019; Hammar, Perry and Gullström, 2016). While there has been disruption to local soft-bottom communities, the area between turbine foundations appears to be large enough to accommodate populations displaced by the physical footprint of the turbine foundations (Reubens, Degraer and Vincx, 2014; Lindeboom et al., 2011; Stenberg et al., 2015; Wilber, Carey and Griffin, 2018; Vandendriessche, Derweduwen and Hostens, 2015). Coates *et al.* (2015) observed a rapid recovery (within months) of macrobenthic communities following the installation OWT using gravity-based foundations. Despite dredging activities and subsequent loss of benthic habitat following installation of turbines, the communities recovered to baseline levels, suggesting strong resilience within these communities and a negligible long-term impact.

3.3 Reef/reserve effect

3.3.1 Marine mammals

The introduction of hard substrate into a marine environment provides new space and habitat for many organisms to colonize (Langhamer, 2012; Krone et al., 2017). These 'artificial reefs' become populated by different species and functional groups of organisms and may, over time, develop a rich ecosystem. By increasing the habitat heterogeneity of an area, such structures promote species diversity, increased abundances, and greater density. Scour protection, in particular, offers great opportunity for enhanced, heterogeneous habitat which promotes reef growth (Langhamer, 2012). Furthermore, an OWF array may act as a sanctuary for many fish species from trawling activities (Teilmann and Carstensen, 2012; Scheidat et al., 2011). While there is some debate as to whether artificial reefs increase or concentrate existing biomass, it is clear they may provide enhanced foraging opportunities for marine mammals. However, such indirect benefits are difficult to detect (Thompson et al., 2010). Consequently, while many studies have hypothesized the possible benefit to marine mammals, there are few which have directly demonstrated an increase in foraging opportunities.

Some pinnipeds have been reported displaying a preference for foraging behaviour in and around OWFs. Telemetry studies have shown harbour seals to trace the physical footprint of OWF in Scottish waters while foraging (Russell et al., 2014), and Australian fur seals have displayed a preference for foraging in the vicinity of sea floor components of MRE infrastructure (Arnould et al., 2015). In contrast, a study by McConnell, Lonergan and Dietz (2012) demonstrated how satellite tagged harbour seals ignored wind turbine foundations, indicating that although there was no disturbance to the seals, neither was there a benefit through the provision of an attractive food resource.

Mikkelsen *et al.*, (2013) observed the re-emergence of a harbour porpoise population following the restoration of a rocky reef habitat in the shallow waters of the northern Kattegat, Denmark. Their results recorded increased harbour porpoise presence and foraging activity and suggested that this population took advantage of the newly available food source located at the reef. The potential artificial reef effect of OWF may lead the aggregation of fish and crustaceans and thus to similar foraging opportunities for other cetaceans, as noted in various review papers (see reviews by Bailey, Brookes and Thompson, (2014); Bergström *et al.*, (2014); Schuster, Bulling and Köppel, (2015); Hammar, Perry and Gullström, (2016)).

Harbour porpoise abundance at the Egmond aan Zee project post-construction activity surpassed that recorded during the pre-construction base-line survey effort (Scheidat et al., 2011). While no reason could be reliably determined, two possibilities were suggested; the increase in habitat complexity associated with the new artificial reef and greater foraging opportunities may have attracted more individuals to the area; alternatively, the reduction in cargo vessels and beam trawling activities and subsequent reduced soundscape within the exclusion zone surrounding the OWF may provide refuge to harbour porpoise from the chronic noise levels experience outside.

3.3.2 Fish and shellfish

Comparing the amount of available hard substrate from shipwrecks in the German North Sea with what may become available through planned OWFs suggests an increase in species limited to hard substrates by between 25 – 165% (Krone et al., 2013a). Commercially important shellfish and demersal fish species may benefit from a projected 430% increase of hard substrate over what is currently available from shipwrecks in that same area. Furthermore, wind turbine foundations provide stepping-stones for species range expansion, predicting an increase in the commercially important blue mussel (*Mytilus edulis*) (Krone et al., 2013b). This may result in ecological system changes, including increased secondary hard substrate, and higher filtration of sea water (Krone et al., 2013b). However, it must also be noted that while the introduced hard substrate may provide a steppingstone for the expansion of native reef forming species, it may also allow a similar pathway for invasive species to expand their range.

The structure of fish communities around an obsolete and thus well-established oil-platform in the North Sea were assessed, with Saithe (*Pollachus virens*), haddock (*Melanogrammus aeglefinus*) and Atlantic cod the most numerous species (Fujii, 2015). The steel jacketed structures are somewhat analogous to OWF turbine pylons, which suggest the importance of an artificial reef structure as a foraging habitat for commercially important fish species. Krone *et al.* (2017) compared mobile demersal communities among different OWF foundation types and found that monopile foundations with scour protection were colonized by reef-associated fauna including commercially important brown crab (*Cancerus pagurus*) at more than twice the number found at foundation types without scour protection, with additional evidence of development of nursery sites for at foundations with scour protection. Tripod and jacket foundation types had a much higher prevalence of soft-bottom species. Such results suggest that increasing the complexity of introduced hard substrate may give rise to greater diversity and abundance, particularly among crustacean species (Langhamer and Wilhelmsson, 2009; Wilhelmsson and Langhamer, 2014; Langhamer, 2012). However, a tagging study on the common shore crab (*Carcinus maenas*) found no effects, positive or negative, during and post installation of an offshore wind farm utilising gravity-based foundations with scour protection.

Patterns in fish community structure and distribution showed no long-term negative impacts on key species 7 years post-construction at the Horns Rev 1 OWF (Stenberg et al., 2015). There was a trend in reef-associated fish moving away from turbine foundations, and whiting (*Merlangus merlangus*) numbers were highest away from the turbines. Species diversity was highest close to the turbine foundations, and the authors suggest that the reef foundations are large enough to attract fish, but not so large that they adversely impacted the soft-bottom communities between the turbines. Furthermore, the exclusion of trawl fisheries allowed the soft-bottom communities to recover from historical fishing pressure. Similar results were found during a short-term study of an OWF in the Dutch coastal zone where Lindeboom *et al.* (2011) found no adverse effects on the fish communities two years post-construction. Hard-substrate communities were established with high densities of reef-associated fish observed at the monopiles and associated scour protection. Increased abundance of bivalves were attributed to fisheries exclusions rather than introduced hard-substrate. Certain commercially important demersal species appeared to find shelter within the OWF (Atlantic cod, common sole, whiting), but only minor changes occurred within the overall fish assemblages. Reubens, Degraer and Vincx (2011) observed elevated numbers of pouting around OWF foundations compared to local soft-bottom habitat, with stomach content analysis revealing a preference for prey items which have colonised the hard substrate of the turbine foundations. Bergström, Sundqvist and Bergström, (2013) found no large-scale differences in diversity and abundance between an established OWF and two reference sites, but smaller spatial scale changes were evident. There were increased densities of piscivorous fish (Atlantic cod, short-horn sculpin *Myoxocephalus scorpius*), European eel) close to turbine foundations, while there was also an increase in reef-associated fish such as goldsinny wrasse (*Ctenolabrus rupestris*) on the foundations. Lower numbers of black goby (*Gobius niger*), eelpout (*Zoarces viviparus*), and shore crab were observed which may indicate top-down predation by the piscivorous fish community. Reubens *et al.* (2013a) found enhanced biomass and densities of Atlantic cod and pouting at OWFs, with highest numbers occurring in summer and autumn, periods of intensive feeding for these species which suggests the OWFs can be important foraging habitats. Reubens *et al.* (2013c) also found the condition of pouting to be enhanced at OWFs. Other benefits to gadoids such as shelter from currents and predators have also been suggested (Reubens et al., 2014), with gadoids at the OWF showing high site fidelity, while specific age groups of cod and pouting were seasonally attracted to the OWFs (Reubens, Degraer and Vincx, 2014). However, there is not yet evidence for regional scale changes in production, although local production has been enhanced.

van Hal, Griffioen and van Keeken (2017) found local changes in fish assemblages in the Dutch part of the southern North Sea attributed to an increase in available hard substrate provided by the OWF and associated increase in habit complexity. This contrasts with Vandendriessche, Derweduwen and Hostens (2015), who focussed on the sandy bottom areas between turbines at Thornton and Bligh Banks in the Belgian Part of the North Sea, and found a temporary decrease in the abundance of dabs, dragonets, and ophiuroids, but some evidence for refugium effects, with larger sized plaice located within the OWFs than outside. Another study on the soft-bottom sandy habitat between turbine foundations found no artificial reef benefits for flatfish species (Wilber, Carey and Griffin, 2018). However, no negative impacts on flatfish species were detected from the construction and operation of the wind farm either. Coates *et al.* (2016) recorded an increase in the abundance of reef building species within the no fishery area three years post-establishment of an OWF in the Belgian Part of the North Sea. It was suggested that over time, this may lead to the establishment of a rich and sheltered habitat from a formerly less productive ecosystem, providing refuge, foraging, and breeding opportunities to higher trophic levels (Rabaut et al., 2010; de Juan, Thrush and Demestre, 2007; Bergman and Hup, 1992; Callaway, 2006). De Troch *et al.* (2013) suggest that the artificial reef created by the presence of OWF foundations provides sufficient energy to support local populations of of Atlantic cod and pouting *(Trisopterus luscus)*. Furthermore, the authors state their findings support the production over attraction hypothesis at OWFs.

In a series of papers, Raoux *et al.* (2017, 2019, 2018) modelled the ecological impacts from a planned offshore wind farm in Atlantic French waters. The first found positive impacts for higher predators including marine mammals, predatory fish, and seabirds due to greater foraging opportunities provided by dense aggregations of prey within the OWF. The second suggested an increase in benthic species abundance within the OWF despite any cumulative impacts from OWF, climate change, and fisheries pressures. The third study predicted a maturation of the ecosystem from the addition of hard substrate (turbine foundations) and fishing restrictions, with estimated fisheries closures not appearing to alter the ecosystem structure or functioning.

The restriction of fishing activities within the exclusion zone of an OWF will relieve fishing pressures on marine communities located therein (Wilhelmsson and Langhamer, 2014; Hammar, Perry and Gullström, 2016). However, it also important to note that the displacement of fishing effort from one area may result in an increase in fishing pressure in another and must be considered by managers (Vaughan, 2017). Roach et al. (2018) found increased size and abundance of European lobster during the temporary closure of the fishery during construction of an offshore windfarm in northeast UK. The fishery was able to recuperate some of the financial losses through increased catch rates and higher quality of lobsters immediately following the reopening of the fishery. The provision of artificial reefs which can act as fish aggregating devices, and fisheries exclusions, mean OWF have the potential to act as *de facto* Marine Protected Areas (MPAs) or marine reserves (Ashley, Mangi and Rodwell, 2014). It is unlikely that bottom trawling or dredging will be permitted around turbines, greatly benefitting benthic habitats and communities which rely on them for food and shelter (Langhamer, 2012). Such *de facto* MPAs may even be more effective than established networks of small MPAs, where regular fishing infringements occur, drastically reducing the benefit of a reserve, with little of the expected 'spill over'(McClanahan and Mangi, 2000) benefit occurring (Little *et al.* 2005). In some cases, static gears such as set nets and pots/traps may be permitted (Hooper and Austen, 2014; de Groot et al., 2014), and the same could be true for mid-water trawling or seines. However, evidence to date suggests that even where permitted, the fishing industry are reluctant to operate within OWF areas.

Ashley, Mangi and Rodwell (2014) reviewed the potential for OWFs to act as MPAs, and suggest that increased occurrence of species at artificial structures is species-specific, with the majority of no commercial value. Reefs designed to mimic habitat favoured by species of commercial value did see success, attracting schooling gadoids in particular. In contrast, soft sediment species such as flatfish and gobies showed either decreases, or no increase in abundance. However, this is contrary to studies by Wilber, Carey and Griffin (2018), Vandendriessche, Derweduwen and Hostens (2015) and Lindeboom *et al.* (2011), who all found either an increase, or no negative effect on species favouring soft-bottomed habitats including American plaice, European plaice, and dab. Furthermore, Methratta and Dardick (2019) conducted a meta-analysis of finfish abundance at offshore wind farms with results illustrating a clear overall increase in abundance of fish within OWF compared to reference sites (**Figure 3.2**). The review further illustrated an increased abundance around individual wind turbine foundations, and suggested that the number of turbines, density of turbines, overall OWF footprint area, and the edge-to-edge ratio could influence the absolute number of fishes within an established OWF.

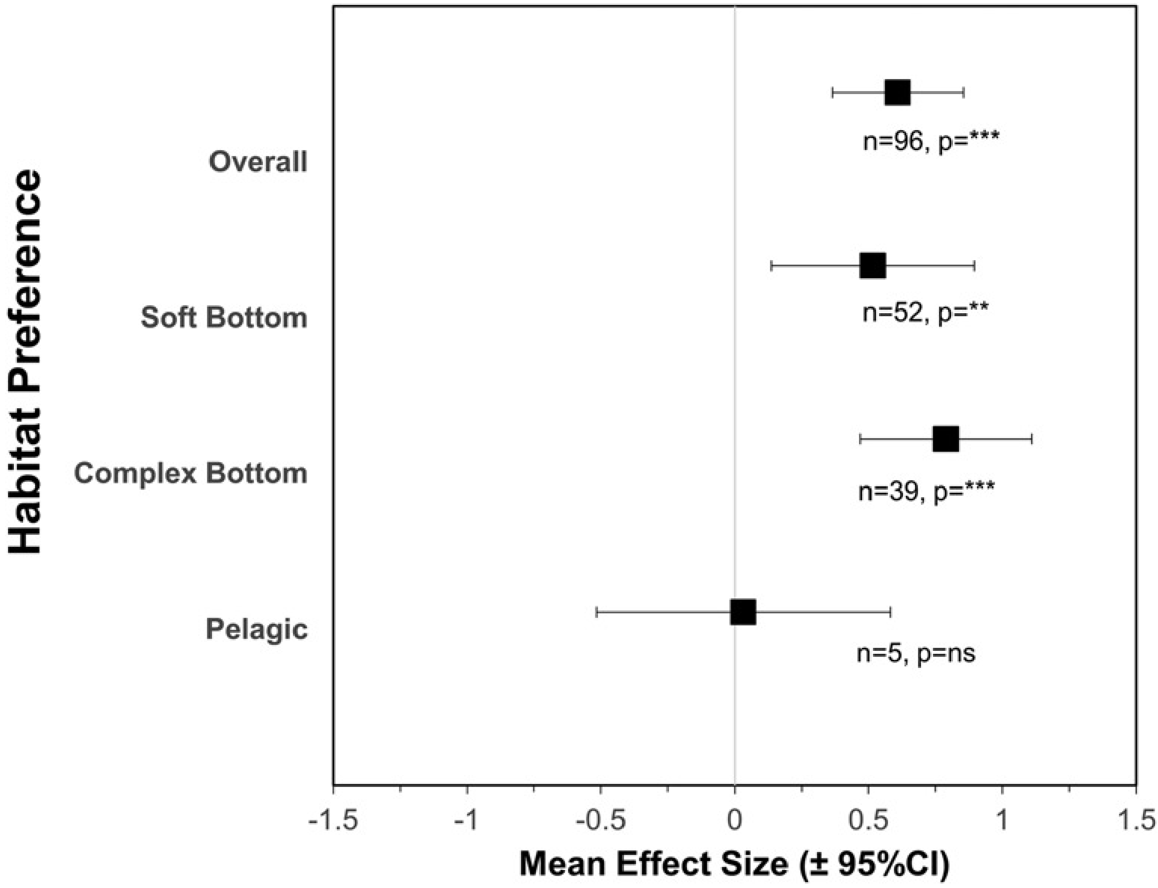

Figure 3.2: Overall effect size (mean ± 95% CI) and effect size for each general habitat group (soft-bottom, complex-bottom, pelagic). Significance in figures is denoted as p<0.05*, p<0.01**, p<0.001***. N = number of records of abundance per group. Adapted from Methratta and Dardick (2019).

3.4 Pollution

Pollution at an OWF may take the form of contaminated sediments resuspended during construction/decommissioning activities or leakage or spills of lubricants, hydraulic fluids, and fuels from vessels and turbines. Biocides may also be of concern. Carstensen *et al.* (2006) and Simmonds & Dolman (2008) mention the potential resuspension of contaminated sediments and increasing turbidity during the construction/decommissioning of offshore wind farms as a possible negative impact, but don't discuss how, or whether that impact is direct (e.g. does the resuspension disrupt foraging abilities or are toxins ingested?) or indirect (e.g. are prey species negatively impacted, or do they ingest toxins and thus act as a vector for the biomagnification of pollutants into marine mammals). Simmonds & Brown (2010) briefly mention the potential for increased chemical pollution to cause harm to marine mammals, possibly arising from leaks or spills of lubricants, hydraulic fluids or biocides to the environment. The risks of leaks or spills may be increased within an OWF as there is a greater potential for ship collision and possible leakage from operational turbines. However, further studies are required to improve our knowledge in this area.

4 Cables/Mooring lines

4.1 Summary

Power cables will become more prevalent within the marine environment as they are used to transport power from OWF to the national grid. Cables carrying electricity will generate both electric and magnetic fields (EMF) and heat. Laying cables causes suspension of sediment in soft-bottom habitats, with associated turbidity and the potential for suspension of toxins and heavy metals which had been stored in the sediment. The same process will produce noise, and the cables may also vibrate during energisation and transmission of power. Cables may be buried within the sediment, laid down on the sediment surface, or dynamic (as inter-array power cables) hanging in the water column (**Figure 4.1**). These latter may risk the entanglement of marine life.

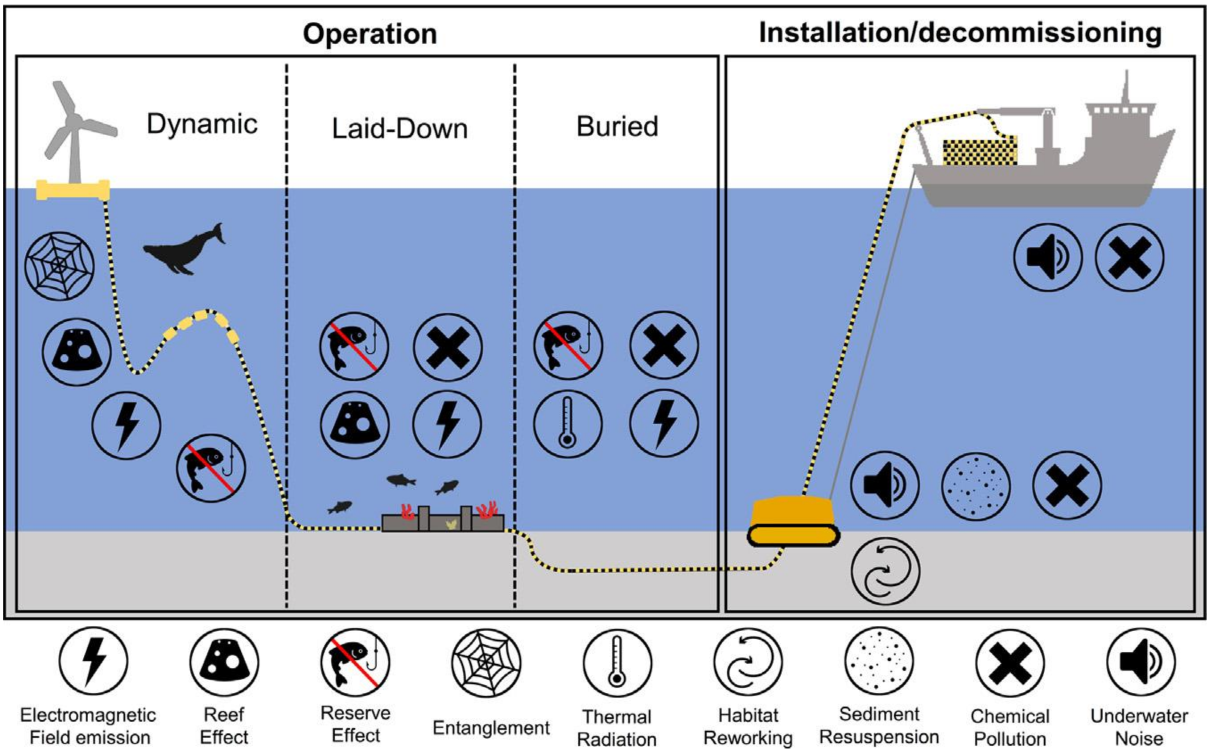

Figure 4.1: Diagram of potential impacts caused by different types of subsea power cables (from Taormina et al., 2018)

Little information on risks from cables and mooring lines to marine mammals currently exists. There are concerns that dynamic power cables, those which hang in the water column, and mooring lines for floating wind farm arrays could pose entanglement risks to large whales. However, the current view is, due to the large diameter and taut configurations of both mooring lines and subsea power cables, is that the direct risk of entanglement is negligible. It is possible that secondary entanglement could occur due to snagging of 'ghost' fishing gear on the cables, and marine life subsequently becoming trapped and drowned. Such snagging events are likely to be infrequent, and regular monitoring of both power cables and mooring lines should reduce this risk.

With regards to electro-magnetic fields (EMF) the risk to marine mammals is again believed to be low. While it has long been believed that marine mammals utilise the earths geomagnetic field for orientation and navigational purposes, the magnetic field emitted by subsea power cables is not thought to be strong to enough disorientate marine mammals, particularly given that its range will be limited to tens of metres at the very most, and likely less. Marine mammals are not electrosensitive and therefore the induced electrical fields pose no risk. There is a greater potential risk to fish, particularly among elasmobranchs, but it is not yet known whether these will impact at the population level. Such effects may cause impacts including disruption to predator/prey interactions, foraging and migration behaviours, the disturbance to one habitat and the possible creation of another. Positive impacts may include localised reef effects where cables are laid down in soft bottom habitats, and the creation of *de facto* marine reserves in areas where fishing is no longer permitted due to the danger of snagging on subsea cables.

However, there is a general paucity of information available. In their 2018 review on the potential risk to fish from subsea power cables, Taormina *et al.*, identified only 9 papers which concerned *in situ* effects of impacts from subsea power cables on the marine environment. In this report, these studies have been supplemented with information from the grey literature (technical reports and environmental impact assessments). As such, this is an area where further research is required.

4.2 Entanglement

4.2.1 Marine mammals

Entanglement has not been viewed as a threat from OWF using monopiles or gravity-based foundations (Schuster, Bulling and Köppel, 2015; Lovich and Ennen, 2013; Bergström et al., 2014; Simmonds and Brown, 2010). Yet the emergence of floating wind farm arrays, which are tethered to the seafloor via cables and also have power cables floating between turbines, necessitates an evaluation of the risk which they pose to marine mammals. Entanglement in ropes, mooring lines, or power cables may lead to death both directly (through drowning) and indirectly through infection of wounds suffered during entanglement. Loss of physical condition and emaciation may be caused by restricted foraging ability as a result of the placement of entangled gear on the bodies of marine mammals. Other considerations include the energetic costs of increased drag on an animal which is designed to be streamlined - particularly relevant for species with high metabolic rates such as the harbour porpoise. If entanglement became a widespread impact, then population level effects are likely in light of the low fecundity and reproduction rates for many marine mammals.

Given the relatively nascent development of floating wind turbine arrays, there is a general paucity of information on impacts. The few studies to date have agreed that the risk of entanglement is low. Copping & Grear (2018) modelled the movements of humpback whales through a floating wind farm array and demonstrated that the tethering cables should be easily avoided, whilst the less taut inter-array cables, which have a marginally greater risk of entanglement, could be floated at depths lower than the maximum foraging depth of the species most at risk in that region. Harnois *et al.* (2015) described the main parameters of interest for potential entanglement as: tension characteristics of cables; the swept volume ratio; and the mooring curvatures. Taut configurations had the lowest risk of entanglement, while those catenary moorings with chains and nylon ropes, or with accessory buoys had the highest risk. However, the absolute risk of entanglement was found to be quite low.

Species most at risk of entanglement in lines and cables in the water column are the large whales (baleen whales and sperm whales). Dolphins, by virtue of their smaller size, increased agility, and active echolocation, although vulnerable to bycatch in fishing nets, are much less likely to become entangled in lines/cables/chains. However, care must be taken with any loose or dangling lines, particularly with ropes, which have been known to entangle pinnipeds leading to potentially serious injuries and death.

Secondary entanglement in "ghost" fishing gear, which may become snagged on mooring lines or cables, is also a concern (Benjamins et al., 2014). Ghost nets or fishing gear are gears which have become lost and drift through the water column until they settle on the seabed. Often, these gears can be many metres in length. Should these become snagged on mooring lines they may prove a significant barrier for marine megafauna transiting through an MRE array. Active monitoring for such a threat should be a component of any regular maintenance work.

4.2.2 Fish and shellfish

The likelihood of direct entanglement of fish by cables in the marine is negligible (Taormina et al., 2018), even for the larger species of elasmobranch which exist in Irish waters such as basking shark, six-gill shark, Greenland shark, and porbeagle shark. As for marine mammals, it is possible that lost/discarded fishing gear could become snagged on dynamic power cables or mooring lines and thus entangle fish secondarily (Barreiros and Raykov, 2014; Moore et al., 2013; Macfadyen, Huntington and Cappell, 2009; Harnois et al., 2015; Benjamins et al., 2014; Stelfox, Hudgins and Sweet, 2016). This may become an issue for dense arrays of floating wind turbines. However, given the development of such arrays is still at a nascent stage it is currently not possible to quantify any such risk (Taormina et al., 2018).

4.3 Electro-Magnetic Fields

Electromagnetic fields are present naturally, either via emanations from the sun, or due to the rotation of the Earth. When considering the effects of EMF from electrical cables and MRE devices on marine animals, knowledge of the EMF as a source of potential effect is fundamental. This background can then be used to contextualize the electromagnetic (EM) environment in which receptor organisms are immersed in the marine environment. According to current industry specifications, the cables used inside tidal, wave, and wind energy arrays carry AC power, while those which transmit power back to shore, or a storage device utilise DC power. The field strength of the magnetic and electric fields generated depends on the amount of electrical current in the cable. Other factors that influence the strength of the fields are the magnetic shielding of the cable. **Figure 4.2** illustrates the extent of a magnetic field emitted by AC and DC cables buried in the sediment.

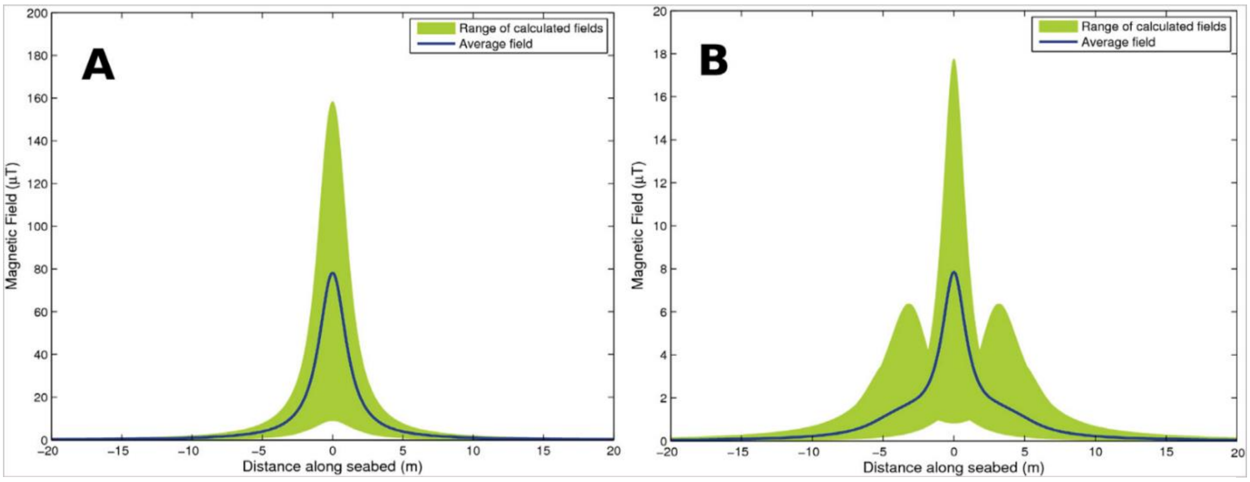

Figure 4.2: Modelled magnetic fields at the sediment-water interface originating from different types of buried submarine cables in operation. A: calculated data based on 9 DC cables. B: calculated data based on 10 AC cables. Source: Taormina et al., (2018)

4.3.1 Marine mammals

Although the use of geomagnetic navigation by marine mammals has long been hypothesised due to their ability to detect variation in the magnetic field (Gill et al., 2014), it has yet to be demonstrated experimentally. There have been very few studies investigating what impact, if any, anthropogenic induced EMF may have on marine mammals. What has been published suggests there is no evidence of any negative impacts on marine mammals from EMF (Gill et al., 2014). The EMF-field of an industry-standard cable has been shown to drop to back to levels of the ambient magnetic within 20 m (Frid et al., 2012), a range which is unlikely to cause major disturbance to any movement of marine mammals. It should be stressed, however, that while the apparent risk is low, we are still largely under informed due to the dearth of relevant studies in this field.

4.3.2 Fish and shellfish

Many fish species can detect magnetic fields; fewer are known to be electrosensitive yet many of these are widespread and numerous (the elasmobranchs). The use of magnetic fields is considered chiefly related to orientation and migration, while elasmobranchs (sharks and rays) are known to use electric impulses to locate prey. Therefore, EMF fields may cause negative impacts on fish species through disrupting predator/prey interactions, avoidance/attraction, effects on navigation and orientation capabilities, and physiological developmental effects.

Elasmobranches are known to detect electrical fields as low as 0.005 μV cm^{-1} (Normandeau et al., 2011), and magnetic fields from 20 - 75 μT (Bochert and Zettler, 2006). While strong electric fields are likely to repel most elasmobranchs, low fields are similar to those emitted by prey items and sharks have been observed to be attracted to them (Taormina et al., 2018). Skate have been also been observed to increase their foraging behaviour in the presence of an EMF >52 μT emitted by a high voltage DC subsea cable, which was above the ambient level for this region (ambient level measure at 51.3 μT at control site; max observed recorded eat exposure site was 65.5 μT) (Hutchison et al., 2018).

However, these reactions will vary depending on species, and, in some instances on individuals within species (Copping et al., 2016). The same study also observed a significant yet subtle effect on the behaviour of the American lobster (*Homarus americanus*). Lobster tended to stay closer to the seabed and were observed making larger turns, using more of the central space of the enclosure which overlaid the subsea power cable, which suggests a slight attraction to the EMF.

Diadromous fish species (those which migrate between freshwater and marine habitats during different stages of their life) are known to use geomagnetic fields during migration (Taormina et al., 2018; Gill, Bartlett and Thomsen, 2012). In a study of the movement of eels and members of the salmon family relative to noise and EMF emitted by MRED in the UK, it was found that the European eel deviates temporarily from their migration path when encountering cables which emitted a magnetic field, particularly in waters less than 20 m deep (Gill, Bartlett and Thomsen, 2012). In the Baltic Sea, European eels were observed to slow as they swam across power cables (Westerberg and Legenfelt, 2008). There has been no report of a population-scale impact from these effects, but data is limited, and further study is likely required.

Klimley, Wyman and Kavet (2017) observed that magentic anomalies emitted by bridges across the San Francisco Estuary, including the Bay and Golden Gate bridges, did not disturb the migration behaviour of juvenile Chinook salmon (*Oncorhynchus tshawytscha*) downstream, nor returning green sturgeon (*Acipenser medirostris*) migrating upstream. Magnetic anaomlaies were recorded as orders of magnitude greater than that which would be emitted by planned subsea DC power cables (Kavet, Wyman and Klimley, 2016), and therefore, it is unlikely that the EMF emitted by the cable would have a negative impact on the migration of these two magento-senstive species. Acoustic biotelemetry work found that the number of juvenile migrating Chinook salmon transiting across a DC power cable increased when energized, suggested a mild attraction behaviour (Wyman et al., 2018).

In terms of physiological effects, very little empirical evidence has been published. A recent study by Fey *et al.* (2019) assessed the threat posed by anthropogenic EMF on the success of early life stages of fishes. Rainbow trout (*Oncorhynchus mykiss*) are a salmonid closely related to Atlantic salmon and brown/sea trout (*Salmo trutta*) found in Irish waters. Eyed eggs were subjected to a static magnetic field of 10,000 µT and 50 Hz EMF of 1,000 µT for a period of 36 days, which equated to 25 and 26 days post-hatching respectively. No effects were found on embryonic or larval mortality, hatching time, larval growth, or time of larval swim up from the bottom in either group compared with the control. However, increased yolk-sac consumption was observed in EMF treated groups. The subsequent larvae were less efficient at feeding, and attained a lower weight at age, thus potentially impacting on their survival rate. The edible brown crab (*Cancer pagurus*), a commercially important species in Irish waters, was subjected to a simulated EMF field at 2,800 and 40,000 µT in laboratory conditions (Scott, Harsanyi and Lyndon, 2018). Exposure had no effect on physiological indicators of stress including haemocyanin concentrations, respiration rate, activity level or antennular flicking rate, but significantly disrupted haemolymph L-Lactate and D-Glucose natural circadian rhythms. In addition, crabs showed clear attraction to EMF, and significantly reduced the time spent roaming. It is possible that berried females (those carrying eggs) could be attracted to EMF emitted by subsea power cables and spend their 6 to 9 month hibernation period nearby. The consequences of this for the eggs and larval stages of crabs are unknown. Consequently, it is clear that there is some impact from EMF on crabs, and this must be considered when planning MRED.

4.4 Sediment suspension

Sediment suspension may occur during cable laying, and last from hours to a day for a given length of cable (Taormina et al., 2018). The overall duration will depend on the length of the cable laying process, the grain size of the sediment, and the local physical oceanography. Plumes of suspended sediment will increase turbidity and thus limit available light. This can decrease the ability of fish to use vision to detect prey. Mineral particles in the water column can also damage the gills of larval fish. However, these effects are likely to be short-term and temporary.

4.5 Noise/vibration

In-situ data concerning noise emitted during cable installation is very scarce (Taormina et al., 2018). Two studies have recorded sound pressure levels of 178 dB (0.7 – 50 kHz) (Nedwell and Howell, 2004) and 188.5 dB (11 kHz) (Bald et al., 2015) 1 m from source for installation in sandy gravel and sand, respectively. Modelling these data gives an estimated SPL of > 120 dB in an area of 400 km^2. The frequencies at which these noises are emitted above the hearing ranges of most fish (see **Figure 2.2**).

4.6 Heat

The transmission of power through a cable results in a loss of small amount of energy through heat. This leads to heating of the cable surface and the associated vicinity. AC cables have a greater loss of energy through heat transmission than DC at equal transmission rates. A constant flow of water dissipates the heat away, and it is restricted to the surface of the cable. In buried cables, however, the sediment may become heated, with possible effects on infaunal communities. This is particularly true of dense, highly compacted sediment types. Heat loss may be reduced via the physical characteristics and electrical tensions of the cables, the type of sediment present, and other physical characteristics of the environment.

Little published data on how heat loss may impact the environment exists (Taormina et al., 2018). This lack of knowledge prevents our ability to draw conclusions on ecological impacts. However, given the confined nature of the corridor along which heat loss may be emitted and the expected weakness of the thermal radiation, such impacts are not likely to be significant (Taormina et al., 2018).

4.7 Reef effect

Artificial structures such as cables and mooring lines are expected to have a limited reef effect in areas of pre-existing hard substrate. In soft sediments, however, unburied cables may display stronger reef effects. Please see Section 3.3 for detailed discussion of reef effects.

4.8 Reserve effect

The reserve effect from cables is likely to be similar to other built infrastructure. Due to the risk of snagging, bottom trawling cannot occur where subsea power cable routes exist. The same may be true for anchoring and dredging activities. Reserve effects are discussed in detail in Section 3.3.

5 Vessels

5.1 Marine mammals

The use of vessels during construction, maintenance and decommissioning of OWF is likely to have effects on marine mammals through attraction to or avoidance of vessels, or collision. In addition to potential behavioural responses to vessels themselves, vessel noise may result in masking of biological signals used for communication, foraging and navigation, and these are dealt with in Section 2.6.

A number of behavioural responses to vessels have been recorded. Culloch *et al.* (2016) found that construction related activities including vessel traffic reduced the presence of harbour porpoise, common dolphin, and Minke whale in northwest Ireland. Mikkelsen *et al.* (2019) tagged harbour and grey seals in the North Sea during 2015 – 2016 and showed that seals were exposed to audible vessel noise 2.2 – 20.5 % of their time in water. Furthermore, interruption of behaviours such as resting coincided with high vessel noise. Wisniewska *et al.* (2018) recorded changes in the behaviour of harbour porpoises when exposed to vessel noise at received levels greater than 96 dB re 1 µPa which impacted foraging behaviour. Similarly, harbour porpoise exposed to vessel traffic in southwest Wales showed a negative behavioural response, however, increased vessel speeds and vessel type were more significant factors, with 75 % of all negative response behaviours recorded in response to high-speed planing-hulled vessels (Oakley, Williams and Thomas, 2017). The study also noted a porpoise with a non-fatal propeller injury highlighting potential collision risk for this species. Similarly, Pirotta *et al.* (2015) described how the presence of boats disturbed bottlenose dolphin feeding behaviour, resulting in short-term reductions in foraging rates by up to 49 %. This effect increased with increasing number of vessels, and depended on vessel type, with motorboats causing the most disturbance. Sustained disturbance to foraging may significantly impair condition, particularly in species with a high metabolic rate such as harbour porpoise, with the potential for population level impacts in areas of high vessel activity. Seuront & Cribb (2011) demonstrated greater complexity in dive duration patterns of Indo-pacific bottlenose dolphins (*Tursiops aduncus*) in the presence of motorised vessels, suggesting an impact on natural behaviours. Similarly, Luís *et al.* (2014) found that mean call rates decreased significantly in the presence of operating motorised vessels. Furthermore, temporary shifts in whistle characteristics also occurred, possibly as a vocal response to the proximity of operating vessels in a busy estuarine environment.

OWFs are likely to exclude or significantly decrease vessel traffic within turbine arrays. In an environment with otherwise high levels of vessel traffic, this reduction will likely reduce ambient noise, and may be attractive to marine mammals for that reason (Scheidat *et al.* 2011). Consequently, marine mammals may be less exposed to traffic in shipping lanes and on fishing grounds, and thus reduce their risk of collision with vessels. However, due to the recorded behavioural response to vessel speed, it is important that OWF associated vessel traffic maintain appropriate speeds while transiting through OWF sites (Williams et al., 2019).

5.2 Fish and shellfish

The potential impact from vessel noise is discussed in Section 2.6. In their review, Whitfield and Becker (2014) summarised the literature on impacts on fish from recreational motorboats. Information on propeller strikes is sparse, but have been recorded, and at a high volume in a least one report.

Occasionally, fish have been observed jumping from the water in a predator escape response attributed to motorboat traffic. Gabel, Lorenz and Stoll (2017) highlighted the potential effects of large ships causing waves, noting increased stranding risk, impacts on foraging, impairment to growth rates and reproduction success, and a shift in community composition, but noted most potential impacts are of chief concern to coastal environments, and particularly to regions with infrequent storm activity. Therefore this is not likely to be an issue in the vicinity of OWF.

Pollution from leakage of fuel and lubricants, antifouling treatments, ballast water, human waste effluent (sewerage), and persistent organic pollutants may impact marine species/communities (Mann, 2006; Whitfield and Becker, 2014; Burgin and Hardiman, 2011; Abdulla and Linden, 2008). Operational day-to-day spills release significant amounts of oil and lubricants into the marine environment each year; within the EU, minor oil spills and leakages each year release 8 times as much oil to the marine environment as was released in the *Exxon-Valdez* disaster (Ng and Song, 2010). Diesel and other oil based-products are known to adversely affect the health fish (Whitfield and Becker, 2014) and thus it is vital that all observed spills and leakages that occur during construction and operation of an OWF are logged and reported to determine if any action is required.

Vessels may also be a vector for invasive species (Cole, Keller and Garbach, 2019; Miralles et al., 2018). Both recreational boating and commercial shipping have been shown to act as conduits for invasive species (Minchin, 2006; Minchin et al., 2006) through species adhering to the vessel hull, or being transported in ballast water (Cabrini et al., 2018). This can have significant although often mixed, effects on local populations of marine life, including fish (Bonanno and Orlando-Bonaca, 2019), benefitting some species, while having a negative impact on others. Negative impacts may include a loss of biodiversity and associated ecosystem resilience due to an invasive species outcompeting local fish populations (Azzurro et al., 2017; Costello et al., 2010), the transport of diseases between water bodies (Whitfield and Becker, 2014), the collapse of a food resource at the base of the food chain causing trophic cascades and potentially causing fish stocks to collapse (Bonanno and Orlando-Bonaca, 2019). In some cases, invasive species have even been suggested as the drivers of extinction events (Bellard, Cassey and Blackburn, 2016). As such, it is vital that vessels that operate within offshore wind farms at all phases of the project adhere to protocols regarding the transport of invasive species.

6 Impact mitigation

Gartman *et al.* (2016a, 2016b) conducted a thorough review of methods employed to mitigate impacts on wildlife from wind energy developments including data up to late 2015. The importance of planning and siting is emphasised, and the example of the German model is used. In it, a "noise prevention concept" was developed, specifically for harbour porpoise. Disturbance to harbour porpoise is prohibited from May – August (breeding season), and an adequate noise exposure buffer is required out of the breeding season. Furthermore, in consideration of cumulative impacts, only 10 % of the German EEZ should be exposed to construction noise at any one time. The value of having a few key demonstrator sites to study interactions with key receptor species has also been highlighted from experience of OWF construction in the Netherlands, Denmark, and Germany (Bailey, Brookes and Thompson, 2014). Mitigation measures to reduce the impact from offshore wind farms on marine life consist chiefly of siting and time-area restrictions, deterrence, and noise dampening measures (Dähne et al., 2017; Brandt et al., 2018, 2013a; Nehls et al., 2016; Gartman et al., 2016a).

6.1 Siting

In order to avoid harm to marine mammals during construction of an offshore wind farm, installations should be sited where such animals are not normally resident, or where they only maintain a seasonal presence. Figure 7 illustrates how the distribution of a species may be seasonally adjusted. Time-area restrictions may allow construction activity to occur outside of times of the day or year when animals are normally active in the vicinity of the sound source (Dähne et al., 2017). Understanding the temporal and spatial distribution of species at risk is therefore of vital importance, and where no baseline data exist, may require surveys incorporating both visual and/or acoustic detection methods to collect the required data (Wingfield et al., 2017).

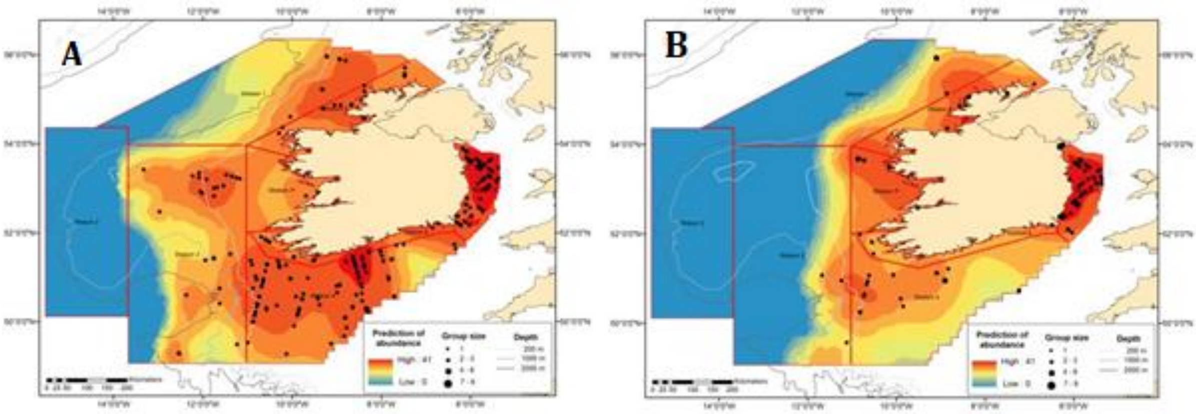

Figure 6.1: Seasonal distribution of the predicted abundance of harbour porpoise. A = summer, B = winter. Source: Observe-aerial/Rogan et al., (2018)

6.2 Deterrence

If time-area restriction methods are not applicable due to a steady presence of sensitive marine species, or where their presence is unpredictable, deterrence may be an option. The Irish guidelines on managing the risk to marine mammals from man-made noise (National Parks & Wildlife Service, 2014) suggest that deterrence methods combined with time-area restrictions should prove the most effective mitigation strategy. Acoustic Deterrent Devices (ADDs), or 'seal scarers' developed to deter seals from fish farms and fishing gears emit an acoustic pulse at a level loud enough to cause animals to vacate the immediate area. They have been shown to be quite effective, particularly in deterrence of harbour porpoise (Brandt et al., 2013a; Kastelein et al., 2017a). Their deployment may be used to disperse species at risk away from a zone where harmful levels of sound noise would be experienced due to construction activity, and should be deployed at least 30 – 40 min prior to the commencement of pile driving activity. In addition, they may be used in conjunction with a 'soft start' or ramp-up of piling activity. Here, the amount of hammer impact energy is gradually increased until the required energy level is achieved, and is often required at the commencement of piling for purely technical reasons. Deterrence methods may be combined with visual and or/acoustic pre-watches conducted by marine mammal observers and/or passive acoustic monitoring, although the effective range of these monitoring techniques is quite limited, and poor weather will often make them redundant (Tougaard and Mikaelsen, 2018).

The use of ADDs should be tightly controlled as they themselves have the potential to cause unnecessary disturbance to marine life (Brandt et al., 2013b). The amount of noise they emit should be set to the minimum required to encourage animals to vacate the zone of concern, and activated for the minimum duration possible.

6.3 Noise mitigation

Deterrence will only mitigate for direct physical harm, i.e. hearing loss, and is not intended to mitigate for loss of habitat due to displacement (Dähne et al., 2017). Noise mitigation systems (NMS) have been developed to reduce underwater noise emissions from construction activities. This type of mitigation works to reduce the received level of noise experienced by the animal, and to reduce the range over which sound levels propagate and can be heard, and thus potentially reduce the level of habitat displacement as well as the risk of hearing loss. These systems take two main forms: bubble and non-bubble curtains. Bellman (2014) supplies an overview of existing noise mitigation systems to reduce pile driving noise. These include the German regulations governing noise exposure levels which must be adhered to within 750 m of the sound source, i.e., sound thresholds of 160 dB SEL and 190 dB SPL. The study notes that the level of sound produced is a product of the pile diameter and energy required to hammer it in to place: an increased diameter is associated with a subsequent increase in noise exposure. **Table 6.1** provides an overview of the different NMSs that have been tested, the amount of reduction in noise, and the number of pilings on which they have been tested.

Table 6.1: Overview of noise mitigation techniques and measured mitigation (in dB), and the number of piles that these techniques have been used for. Table modified after Bellmann *et al.* (2015).

Noise Mitigation technique	ΔSEL [dB]	Number of test (pilings)
Big bubble curtain (>0.3 $m^3/(min*m)$, ballast chain inside, water depth <30 m	10 < 13 < 15	>150 (>300)
Double big bubble curtain (>0.3 $m^3/(min*m)$, ballast chain inside, water depth <30 m, distance between hoses > water depth	14 < 17 < 18	>150 (>300)
Small bubble curtain, use air volume, hole configuration	(5 <) 10 < 14	2
Hydro Sound Dampers (Number and size of HSD elements)	8 < 10 < 13	>50
Noise mitigation screen (IHC-NMS)	10 < 13 < 15	>140
Cofferdam (Function of sealing gasket)	problem < 10, no problem ≥ 20	>10 (>10)
Combination of two big bubble curtain systems (double and single)	15 < 16 < 19	>30 (>70)
Combination of IHC-NMS + big bubble curtain	17 < 19 < 23	>90
Big bubble curtain + hydro sound damper	15 < 16 < 20	>10
Double big bubble curtain (Weyres) + hydro sound damper	14 < 16 < 22	2

*Units: SEL = dB re 1 $\mu Pa^2 s^{-1}$

Big Bubble Curtains (BBC) generally consist of a length of hose/piping laid down on the seafloor around the construction activity. Air is fed through the pipe and emerges through regularly spaced perforations or nozzles along its length. The air expands as it rises, and forms a barrier which scatters, reflects, and absorbs noise as it passes through it (Brandt et al., 2018; BSH and BMU, 2014; Dähne et al., 2017). Bubble curtain configuration is not standardised and has differed from site to site across different projects, incorporating differences in the number of nozzles per hose, the volume of compressed air, the distance between the hose and the piling activity, and the number and size of nozzles (BSH and BMU, 2014). Double Big Bubble Curtains (DBBC) work on the same principle but consist of a second curtain of bubbles from another hose encircling the first hose. The distance between curtains has varied between deployments of this measure.

Little ("Small" in some publications) Bubble Curtains (LBC) work on a similar principle to BBC, but with the bubble hose surrounding the pile in a close fit (Koschinski and Ludeman, 2013). There are variations of this system including:

- layered – a series of horizontal hoses encircling the pile from top to bottom,
- confined – with a casing around the area of rising air bubbles,
- small bubble curtain –a series of vertically orientated pipes surrounding the pile.

Isolation casings are another method employed to dampen noise from construction activity. Generally, these are pile sleeves of different materials or hollow steel tubes of greater diameter than the pile. Examples include the IHC- Noise Mitigation System and the BEKA Shell. They may be used with a confined bubble curtain within the casing to increase their efficiency. Noise reductions of 16 – 18 dB SEL and 13 – 21 dB SPL have been reported (BSH and BMU, 2014). Hydro sound dampers (HSD) (encapsulated bubble system) comprise a series of fixed air-filled balloons or foam elements of different sizes (Nehls et al., 2016; Bellman, 2014; Bellmann et al., 2017) affixed to a net which surrounds the pile and is kept in place using a ballast weight. The advantage over a bubble curtain is the independence from compressed air (which requires significant deck space during operation). Broadband noise reductions of 7 – 13 dB SEL and 7 – 15 dB SPL have been achieved during tests carried out during the installation of a monopile. Cofferdams operate on the principle of reduced sound propagation through air compared with water (Nehls et al., 2016; Bellman, 2014; Bellmann et al., 2017). These are essentially isolation casings with the water removed from the space between the pile and the inner wall of the casing. They have a mitigation potential of approximately 20 dB SEL.

Brandt *et al.* (2018) reported on the first seven wind farms to be constructed in German waters. Of these, 6 had NMS with 5 of those consisting of big bubble curtains and 1 casing type (IHC-NMS). The mean reduction in SEL exceeded during 5% of piling time at 750 m was 7 dB with a range of 2-11 dB (**Table 6.2**).

Table 6.2: Noise levels (in dB) are given as the mean sound exposure level exceeded during 5% of piling time (SEL_{05}) at 750 m separately for unmitigated and mitigated piling events (i.e. without and with noise mitigation systems, NMS). NA: not applicable, BBC: big bubble curtain, IHC-NMS: IHC-noise mitigation system. Adapted from Brandt *et al.* (2018)

Wind farm	SEL_{05} without NMS (sample size)	SEL_{05} with NMS (sample size)	Reduction (dB)	Foundation type	NMS type	Water depth (m)
BARD	179 (2)	NA	NA	Tripod	None	39 – 41
BWII	173 (10)	163 (28)	10	Tripod	BBC	28 – 33
DT	178 (2)	169 (78)	9	Monopile	BBC	21 – 29
GTI	176 (2)	169 (78)	7	Tripod	BBC	38 – 41
MSO	180 (2)	169 (76)	11	Monopile	BBC	24 – 27
NSO	168 (1)	166 (48)	2	Jacket	BBC	22 – 25
RG	NA	163 (8)	NA	Monopile	IHC - NMS	18 – 23

*Units: SEL = dB re 1 $\mu Pa^2 s^{-1}$

6.4 Sediments

The suspension of benthic materials including fine sediments can occur during OWF construction. Some of these sediments may contain toxins or heavy metals in higher concentrations than the surrounding water column. Vaissière *et al.* (2014) note that 10 publications recommended the use of an underwater plough to install cables rather than water jetting. The plough method has been shown to affect a smaller surface area, and thus reduce the amount of suspended sediment and associated increased water turbidity.

7 Conclusion and recommendations

The objective of this report was to provide a current overview on the status of knowledge concerning impacts from the offshore wind industry on marine mammals and fish. The available literature suggests that the most significant stressor to marine life from OWFs is the noise component, specifically that which occurs during pile-driving activities. However, there is a growing concern regarding the impact from chronic, low-level noise such as that from vessel engines and operational wind turbines. There is also evidence that established OWFs provide benefits to marine life through the provision of increased foraging opportunities and shelter via the reduction in fishing effort and artificial reef effect of built structures. Scour protection appears to enhance such effects, with increased biodiversity recorded at foundation bases with scour protection when compared to those which lack it.

Arising from this literature review, a number of recommendations can be made:

1. Data on the distribution and abundance of sensitive species should be used to inform appropriate site selection. The recent 'ObSERVE' programme, incorporating visual and acoustic surveys as well as data held by the National Biodiversity Data Centre (NBDC) are excellent sources of data on the broad distribution of marine mammals and seabirds in Irish waters. However, it may still be necessary to survey potential wind farm sites to ascertain the community assemblages present, and inform appropriate mitigation measures.
2. The use of robust sound propagation models developed for both the hearing abilities of the marine life present and the physical characteristics of that location will be a valuable tool in establishing what mitigation measures may be most appropriate for development areas.
3. Time-area restrictions may allow construction activity to occur outside of times of the day or year when animals are normally active in a proposed OWF site. Understanding the temporal and spatial distribution of species at risk is therefore of vital importance, and where no baseline data exist may require surveys incorporating visual and/or acoustic detection methods to collect the required data.
4. The current NPWS guidelines on permitted noise are based on values described in Southall *et al.* (2007). However, Southall et al. (2019) recently revised these threshold levels. It may be appropriate to revise the Irish guidelines incorporating more recent knowledge and following international best practice.
5. There are no current Irish guidelines concerning the impacts on fish from anthropogenic noise. Those set out by Popper and Hawkins (2019) (see **Table 2.5**) may be adopted as a starting point.

6. Future research on the impacts of noise on fish species should place greater emphasis on field studies and give greater consideration to the particle motion component of sound.
7. Acoustic deterrent devices may be used in conjunction with a 'soft-start' procedure to encourage marine life to vacate the vicinity of imminent construction activity. Their controlled deployment may be used to disperse species at risk beyond the zone where harmful levels of sound noise would be experienced. They should be deployed at least 30 – 40 min prior to the commencement of pile driving activity.
8. Noise mitigation systems should be employed during pile driving. These have been proven to reduce the level of noise exposure and decrease the size of the area affected.
9. Where practical, 'ploughing' should be employed when trenching cables over the use of water jetting.
10. The surface complexity of the hard substrate being deployed should be considered. Increased heterogeneity of the substrate has been shown to increase biodiversity, leading to a more productive and resilient community. It is most applicable to the design of scour protection and anchors of floating OWT arrays. This may help towards offsetting negative consequences arising during the installation and operation of an OWF.
11. Where practical, fishing and shipping activity should be reduced/restricted within OWFs. Some authors recommend that no fisheries be allowed to occur within an OWF, static or otherwise. This leads to OWFs acting as *de facto* MPAs, with well-established benefits in terms of ecosystem recovery and spillover effects for fisheries.
12. Secondary entanglement in "ghost" fishing gear, which may become snagged on mooring lines or cables, is a concern. Active monitoring for such a threat should be a component of regular maintenance work.
13. There is a general lack of scientific studies conducted *in-situ* on impacts to fish from cables, whether via EMF, the addition of hard substrate, or as vectors for invasive species. As such, this is an area where further research is required.
14. Vessels should maintain appropriate speeds when transiting through an OWF. OWFs may provide a quiet space for marine life, and marine mammals in particular. It is important that positive outcome is not undone unconsciously through the lack of relevant protocols.

References

Abdulla, A. and Linden, O. eds., 2008. *Maritime traffic effects on biodiversity in the Mediterranean Sea. Review of impacts, priority areas and mitigation measures*. [online] Malaga, Spain: IUCN Centre for Mediterranean Cooperation. Available at: <https://books.google.ie/books?hl=en&lr=&id=CCa708lYvdIC&oi=fnd&pg=PA5&dq=Maritime+traffic+effects+on+biodiversity+in+the+Mediterranean+Sea&ots=O0EZcNCaFZ&sig=Kk3HI-KAyTLN0JHkaJ1mDs6iZxI&redir_esc=y#v=onepage&q=Maritime traffic effects on biodivers> [Accessed 14 Jun. 2019].

Andersson, M.H., 2011. *Offshore wind farms - ecological effects of noise and habitat alteration on fish*. Department of Zoology, Stockholm University.

Andersson, M.H., Sigray, P. and Persson, L.K.G., 2011. Operational wind farm noise and shipping noise compared with estimated zones of audibility for four species of fish. *The Journal of the Acoustical Society of America*, [online] 129(4), pp.2498–2498. Available at: <http://asa.scitation.org/doi/10.1121/1.3588249>.

Andriguetto-Filho, J.M., Ostrensky, A., Pie, M.R., Silva, U.A. and Boeger, W.A., 2005. Evaluating the impact of seismic prospecting on artisanal shrimp fisheries. *Continental Shelf Research*, [online] 25(14), pp.1720–1727. Available at: <https://www.sciencedirect.com/science/article/pii/S0278434305001007> [Accessed 22 Jun. 2019].

Arnould, J.P.Y., Monk, J., Ierodiaconou, D., Hindell, M.A., Semmens, J., Hoskins, A.J., Costa, D.P., Abernathy, K. and Marshall, G.J., 2015. Use of anthropogenic sea floor structures by Australian fur seals: Potential positive ecological impacts of marine industrial development? *PLoS ONE*, [online] 10(7), pp.1–13. Available at: <http://dx.doi.org/10.1371/journal.pone.0130581>.

Arrambide, I., Zubia, I. and Madariaga, A., 2019. Critical review of offshore wind turbine energy production and site potential assessment. *Electric Power Systems Research*, [online] 167, pp.39–47. Available at: <https://www.sciencedirect.com/science/article/pii/S0378779618303341> [Accessed 23 Jun. 2019].

Ashley, M.C., Mangi, S.C. and Rodwell, L.D., 2014. The potential of offshore windfarms to act as marine protected areas – A systematic review of current evidence. *Marine Policy*, [online] 45, pp.301–309. Available at: <http://dx.doi.org/10.1016/j.marpol.2013.09.002>.

Azzurro, E., Stancanelli, B., Di Martino, V. and Bariche, M., 2017. Range expansion of the common lionfish Pterois miles (Bennett, 1828) in the Mediterranean Sea: an unwanted new guest for Italian waters. *BioInvasions Records*, [online] 6(2), pp.95–98. Available at: <http://www.reabic.net/journals/bir/2017/Issue2.aspx> [Accessed 14 Jun. 2019].

Bailey, H., Brookes, K.L. and Thompson, P.M., 2014. Assessing environmental impacts of offshore wind farms: lessons learned and recommendations for the future. *Aquatic Biosystems*, [online] 10(1), p.8. Available at: <http://aquaticbiosystems.biomedcentral.com/articles/10.1186/2046-9063-10-8> [Accessed 20 Feb. 2019].

Bailey, H., Senior, B., Simmons, D., Rusin, J., Picken, G. and Thompson, P.M., 2010. Assessing underwater noise levels during pile-driving at an offshore windfarm and its potential effects on marine mammals. *Marine Pollution Bulletin*, [online] 60(6), pp.888–897. Available at: <https://www.sciencedirect.com/science/article/pii/S0025326X10000044> [Accessed 21 Feb. 2019].

Bald, J., Hernández, C., Uriarte, A., Castillo, J.A., Ruiz, P., Ortega, N., Torre Enciso, Y. and Marina, D., 2015. Acoustic characterization of submarine cable installation in the Biscay Marine Energy Platform (bimep). In: *Bilbao Energy Week 2015*. [online] Available at:

<https://www.researchgate.net/profile/Juan_Bald/publication/285512370_Acoustic_characterization_of_submarine_cable_installation_in_the_Biscay_Marine_Energy_Platform_bimep/links/565eb87608aefe619b272f16.pdf> [Accessed 12 Jun. 2019].

Barreiros, J.P. and Raykov, V.S., 2014. Lethal lesions and amputation caused by plastic debris and fishing gear on the loggerhead turtle Caretta caretta (Linnaeus, 1758). Three case reports from Terceira Island, Azores (NE Atlantic). *Marine Pollution Bulletin*, 86(1–2), pp.518–522.

Bass, A.H. and Ladich, F., 2008. Vocal–Acoustic Communication: From Neurons to Behavior. In: *Fish Bioacoustics*. [online] New York, NY: Springer New York, pp.253–278. Available at: <http://link.springer.com/10.1007/978-0-387-73029-5_8> [Accessed 28 May 2019].

Bellard, C., Cassey, P. and Blackburn, T.M., 2016. Alien species as a driver of recent extinctions. *Biology Letters*, [online] 12(2), p.20150623. Available at: <http://rsbl.royalsocietypublishing.org/lookup/doi/10.1098/rsbl.2015.0623> [Accessed 14 Jun. 2019].

Bellman, M.A., 2014. Overview of existing Noise Mitigation Systems for reducing Pile-Driving Noise. In: *Inter-noise*. [online] Melbourne, p.11. Available at: <http://link.springer.com/10.1007/s00181-005-0010-0>.

Bellmann, M.A., Schuckenbrock, J., Gündert, S., Müller, M., Holst, H. and Remmers, P., 2017. Is There a State-of-the-Art to Reduce Pile-Driving Noise? In: *Wind Energy and Wildlife Interactions*. [online] Cham: Springer International Publishing, pp.161–172. Available at: <http://link.springer.com/10.1007/978-3-319-51272-3_9> [Accessed 28 Mar. 2019].

Benjamins, S., Hamois, V., Smith, H.C.M., Johanning, L., Greenhill, L., Carter, C. and Wilson, B., 2014. *Understanding the potential for marine megafauna entanglement risk from marine renewable energy developments*. [online] *Scottish Natural Heritage Commissioned Report No. 791*, Available at: <http://www.snh.gov.uk/publications-data-and-research/publications/search-the-catalogue/publication-detail/?id=2174>.

Bergman, M.J.N. and Hup, M., 1992. Direct effects of beamtrawling on macrofauna in a sandy sediment in the southern North Sea. *ICES Journal of Marine Science*, [online] 49(1), pp.5–11. Available at: <https://academic.oup.com/icesjms/article-lookup/doi/10.1093/icesjms/49.1.5> [Accessed 13 Jun. 2019].

Bergström, L., Kautsky, L., Malm, T., Rosenberg, R., Wahlberg, M., Åstrand Capetillo, N. and Wilhelmsson, D., 2014. Effects of offshore wind farms on marine wildlife - A generalized impact assessment. *Environmental Research Letters*.

Bergström, L., Sundqvist, F. and Bergström, U., 2013. Effects of an offshore wind farm on temporal and spatial patterns in the demersal fish community. *Marine Ecology Progress Series*, [online] 485, pp.199–210. Available at: <http://www.int-res.com/abstracts/meps/v485/p199-210/> [Accessed 3 Apr. 2019].

Bochert, R. and Zettler, M.L., 2006. Effect of Electromagnetic Fields on Marine Organisms. In: *Offshore Wind Energy*. [online] Springer Berlin Heidelberg, pp.223–234. Available at: <http://link.springer.com/10.1007/978-3-540-34677-7_14> [Accessed 11 Jun. 2019].

Boehlert, G.W. and Gill, A.B., 2010. Environmental and Ecological Effects of Ocean Renewable Energy Development: A Current Synthesis. *Oceanography*, [online] 23(2), pp.68–81. Available at: <http://ir.library.oregonstate.edu/xmlui/handle/1957/16152> [Accessed 24 Apr. 2017].

Bolle, L.J., de Jong, C.A.F., Bierman, S.M., van Beek, P.J.G., van Keeken, O.A., Wessels, P.W., van Damme, C.J.G., Winter, H. V., de Haan, D. and Dekeling, R.P.A., 2012. Common Sole Larvae Survive

High Levels of Pile-Driving Sound in Controlled Exposure Experiments. *PLoS ONE*, [online] 7(3), p.e33052. Available at: <https://dx.plos.org/10.1371/journal.pone.0033052> [Accessed 28 May 2019].

Bonanno, G. and Orlando-Bonaca, M., 2019. Non-indigenous marine species in the Mediterranean Sea—Myth and reality. *Environmental Science & Policy*, [online] 96, pp.123–131. Available at: <https://www.sciencedirect.com/science/article/pii/S1462901119300462#bib0490> [Accessed 14 Jun. 2019].

Brandt, M., Diederichs, A., Betke, K. and Nehls, G., 2011. Responses of harbour porpoises to pile driving at the Horns Rev II offshore wind farm in the Danish North Sea. *Marine Ecology Progress Series*, [online] 421, pp.205–216. Available at: <http://www.int-res.com/abstracts/meps/v421/p205-216/> [Accessed 21 Feb. 2019].

Brandt, M., Dragon, A., Diederichs, A., Bellmann, M., Wahl, V., Piper, W., Nabe-Nielsen, J. and Nehls, G., 2018. Disturbance of harbour porpoises during construction of the first seven offshore wind farms in Germany. *Marine Ecology Progress Series*, [online] 596, pp.213–232. Available at: <http://www.int-res.com/abstracts/meps/v596/p213-232/> [Accessed 20 Feb. 2019].

Brandt, M.J., Höschle, C., Diederichs, A., Betke, K., Matuschek, R. and Nehls, G., 2013a. Seal scarers as a tool to deter harbour porpoises from offshore construction sites. *Marine Ecology Progress Series*, [online] 475, pp.291–302. Available at: <http://www.int-res.com/abstracts/meps/v475/p291-302/> [Accessed 21 Feb. 2019].

Brandt, M.J., Höschle, C., Diederichs, A., Betke, K., Matuschek, R., Witte, S. and Nehls, G., 2013b. Far-reaching effects of a seal scarer on harbour porpoises, Phocoena phocoena. *Aquatic Conservation: Marine and Freshwater Ecosystems*, [online] 23(2), pp.222–232. Available at: <http://doi.wiley.com/10.1002/aqc.2311> [Accessed 21 Feb. 2019].

Breithaupt, T., 2002. Sound Perception in Aquatic Crustaceans. In: *The Crustacean Nervous System*. [online] Berlin, Heidelberg: Springer Berlin Heidelberg, pp.548–558. Available at: <http://link.springer.com/10.1007/978-3-662-04843-6_41> [Accessed 22 Jun. 2019].

Breithaupt, T. and Tautz, J., 1988. Vibration sensitivity of the crayfish statocyst. *Naturwissenschaften*, [online] 75(6), pp.310–312. Available at: <http://link.springer.com/10.1007/BF00367325> [Accessed 22 Jun. 2019].

Breithaupt, T. and Tautz, J., 1990. The Sensitivity of Crayfish Mechanoreceptors to Hydrodynamic and Acoustic Stimuli. In: *Frontiers in Crustacean Neurobiology*. [online] Basel: Birkhäuser Basel, pp.114–120. Available at: <http://link.springer.com/10.1007/978-3-0348-5689-8_12> [Accessed 22 Jun. 2019].

Bruintjes, R., Purser, J., Everley, K.A., Mangan, S., Simpson, S.D. and Radford, A.N., 2016a. Rapid recovery following short-term acoustic disturbance in two fish species. *Royal Society Open Science*, [online] 3(1), p.150686. Available at: <http://rsos.royalsocietypublishing.org/lookup/doi/10.1098/rsos.150686> [Accessed 4 Jun. 2019].

Bruintjes, R., Simpson, S.D., Harding, H., Bunce, T., Benson, T., Rossington, K. and Jones, D., 2016b. The impact of experimental impact pile driving on oxygen uptake in black seabream and plaice. In: *Proceedings of Meetings on Acoustics*. [online] Acoustical Society of America, p.010042. Available at: <http://asa.scitation.org/doi/abs/10.1121/2.0000422> [Accessed 4 Jun. 2019].

BSH and BMU, 2014. *Ecological Research at the Offshore Windfarm alpha ventus – Challenges, Results and Perspectives. Federal Maritime and Hydrographic Agency (BSH), Federal Ministry for the Environment, Nature Conservation and Nuclear Safety (BMU).* Springer Spektrum.

Burgin, S. and Hardiman, N., 2011. The direct physical, chemical and biotic impacts on Australian coastal waters due to recreational boating. *Biodiversity and Conservation*, [online] 20(4), pp.683–701. Available at: <http://link.springer.com/10.1007/s10531-011-0003-6> [Accessed 14 Jun. 2019].

Buscaino, G., Filiciotto, F., Buffa, G., Bellante, A., Stefano, V. Di, Assenza, A., Fazio, F., Caola, G. and Mazzola, S., 2010. Impact of an acoustic stimulus on the motility and blood parameters of European sea bass (Dicentrarchus labrax L.) and gilthead sea bream (Sparus aurata L.). *Marine Environmental Research*, [online] 69(3), pp.136–142. Available at: <https://www.sciencedirect.com/science/article/pii/S0141113609001214> [Accessed 3 Jun. 2019].

Cabrini, M., Cerino, F., de Olazabal, A., Di Poi, E., Fabbro, C., Fornasaro, D., Goruppi, A., Flander-Putrle, V., Francé, J., Gollasch, S., Hure, M., Lipej, L., Lučić, D., Magaletti, E., Mozetič, P., Tinta, T., Tornambè, A., Turk, V., Uhan, J. and David, M., 2018. Potential transfer of aquatic organisms via ballast water with a particular focus on harmful and non-indigenous species: A survey from Adriatic ports. *Marine Pollution Bulletin*. [online] Available at: <https://linkinghub.elsevier.com/retrieve/pii/S0025326X18300857> [Accessed 14 Jun. 2019].

California Department of Transportation, 2001. *Pile installation demonstration project, fisheries impact assessment*. San Francisco: San Francisco - Oakland Bay Bridge East Span Seismic Safety Project.

Callaway, R., 2006. Tube worms promote community change. *Marine Ecology Progress Series*, [online] 308, pp.49–60. Available at: <http://www.int-res.com/abstracts/meps/v308/p49-60/> [Accessed 13 Jun. 2019].

Carstensen, J., Henriksen, O. and Teilmann, J., 2006. Impacts of offshore wind farm construction on harbour porpoises: acoustic monitoring of echolocation activity using porpoise detectors (T-PODs). *Marine Ecology Progress Series*, [online] 321, pp.295–308. Available at: <http://www.int-res.com/abstracts/meps/v321/p295-308/> [Accessed 25 Feb. 2019].

Casper, B.M., Halvorsen, M.B., Carlson, T.J. and Popper, A.N., 2017. Onset of barotrauma injuries related to number of pile driving strike exposures in hybrid striped bass. *The Journal of the Acoustical Society of America*, [online] 141(6), pp.4380–4387. Available at: <http://asa.scitation.org/doi/10.1121/1.4984976>.

Casper, B.M., Halvorsen, M.B., Matthews, F., Carlson, T.J. and Popper, A.N., 2013a. Recovery of Barotrauma Injuries Resulting from Exposure to Pile Driving Sound in Two Sizes of Hybrid Striped Bass. *PLoS ONE*, [online] 8(9), p.e73844. Available at: <https://dx.plos.org/10.1371/journal.pone.0073844> [Accessed 28 May 2019].

Casper, B.M., Popper, A.N., Matthews, F., Carlson, T.J. and Halvorsen, M.B., 2012. Recovery of Barotrauma Injuries in Chinook Salmon, Oncorhynchus tshawytscha from Exposure to Pile Driving Sound. *PLoS ONE*, [online] 7(6), p.e39593. Available at: <https://dx.plos.org/10.1371/journal.pone.0039593> [Accessed 28 May 2019].

Casper, B.M., Smith, M.E., Halvorsen, M.B., Sun, H., Carlson, T.J. and Popper, A.N., 2013b. Effects of exposure to pile driving sounds on fish inner ear tissues. *Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology*, 166(2), pp.352–360.

Castro-Santos, L. and Diaz-Casas, V., 2015. Sensitivity analysis of floating offshore wind farms. *Energy Conversion and Management*, [online] 101, pp.271–277. Available at: <https://www.sciencedirect.com/science/article/pii/S0196890415004793> [Accessed 22 Jun. 2019].

Cato, D.H., Dunlop, R.A., Noad, M.J., McCauley, R.D., Kniest, E., Paton, D. and Kavanagh, A.S., 2016. Addressing Challenges in Studies of Behavioral Responses of Whales to Noise. In: A. Popper and Hawkins A., eds., *The Effects of Noise on Aquatic Life II*. [online] New York, NY: Springer, New York, NY, pp.145–152. Available at: <http://link.springer.com/10.1007/978-1-4939-2981-8_17> [Accessed 22 Jun. 2019].

Cheung, W.W.L., Pinnegar, J., Merino, G., Jones, M.C. and Barange, M., 2012. Review of climate change impacts on marine fisheries in the UK and Ireland. *Aquatic Conservation: Marine and Freshwater Ecosystems*, [online] 22(3), pp.368–388. Available at: <http://doi.wiley.com/10.1002/aqc.2248> [Accessed 21 Jun. 2019].

Cholewiak, D., Clark, C.W., Ponirakis, D., Frankel, A., Hatch, L.T., Risch, D., Stanistreet, J.E., Thompson, M., Vu, E. and Van Parijs, S.M., 2018. Communicating amidst the noise: Modeling the aggregate influence of ambient and vessel noise on baleen whale communication space in a national marine sanctuary. *Endangered Species Research*, 36.

Coates, D.A., van Hoey, G., Colson, L., Vincx, M. and Vanaverbeke, J., 2015. Rapid macrobenthic recovery after dredging activities in an offshore wind farm in the Belgian part of the North Sea. *Hydrobiologia*, [online] 756(1), pp.3–18. Available at: <http://link.springer.com/10.1007/s10750-014-2103-2> [Accessed 13 Jun. 2019].

Coates, D.A., Kapasakali, D.-A.A., Vincx, M. and Vanaverbeke, J., 2016. Short-term effects of fishery exclusion in offshore wind farms on macrofaunal communities in the Belgian part of the North Sea. *Fisheries Research*, [online] 179, pp.131–138. Available at: <https://www.sciencedirect.com/science/article/pii/S0165783616300492> [Accessed 3 Apr. 2019].

Cole, E., Keller, R.P. and Garbach, K., 2019. Risk of invasive species spread by recreational boaters remains high despite widespread adoption of conservation behaviors. *Journal of Environmental Management*, [online] 229, pp.112–119. Available at: <https://www.sciencedirect.com/science/article/pii/S0301479718307308> [Accessed 14 Jun. 2019].

Copping, A. and Grear, M., 2018. *Humpback Whale Encounter with Offshore Wind Mooring Lines and Inter-Array Cables Final Report*.

Copping, A., Sather, N., Hannah, L., Whiting, J., Zydlewski, G., Staines, G., Gill, A., Hutchinson, I., A, O., Simas, T., Bald, J., Sparling, C., Wood, J. and Masden, E., 2016. *Annex IV 2016 State of the Science Report: Environmental Effects of Marine Renewable Energy Development Around the World*.

Costello, M.J., Coll, M., Danovaro, R., Halpin, P., Ojaveer, H. and Miloslavich, P., 2010. A Census of Marine Biodiversity Knowledge, Resources, and Future Challenges. *PLoS ONE*, [online] 5(8), p.e12110. Available at: <https://dx.plos.org/10.1371/journal.pone.0012110> [Accessed 14 Jun. 2019].

Cox, K., Brennan, L.P., Gerwing, T.G., Dudas, S.E. and Juanes, F., 2018. Sound the alarm: A meta-analysis on the effect of aquatic noise on fish behavior and physiology. *Global Change Biology*, [online] 24(7), pp.3105–3116. Available at: <http://doi.wiley.com/10.1111/gcb.14106>.

Cox, T., Ragen, T., Read, A., Vos, E., Baird, R.W., Balcomb, K., Barlow, J., Caldwell, J., Cranford, T., Crum, L., D'Amico, A., D'Spain, G., Fernandez, a, Finneran, J.J., Gentry, R.L., Gerth, W., Gulland, F., Hildebrand, J. a, Houser, D.S., Hullar, T., Jepson, P., Ketten, D., MacLeod, C.D., Miller, P., Moore, S.E., Mountain, D., Palka, D., Ponganis, P., Rommel, S., Rowles, T., Taylor, B., Tyack, P.L., Wartzok, D., Gisiner, R.C., Mead, J. and Benner, L., 2006. Understanding the impacts of anthropogenic sound on beaked whales. *Journal of Cetacean Research and Management*, [online] 7(3), pp.177–187. Available at: <http://oai.dtic.mil/oai/oai?verb=getRecord&metadataPrefix=html&identifier=ADA593622%5Cnhttp://www.nmfs.noaa.gov/pr/health/noise/docs/cox_sound.pdf>.

Culloch, R., Anderwald, P., Brandecker, A., Haberlin, D., McGovern, B., Pinfield, R., Visser, F., Jessopp, M. and Cronin, M., 2016. Effect of construction-related activities and vessel traffic on marine mammals. *Marine Ecology Progress Series*, [online] 549, pp.231–242. Available at: <http://www.int-res.com/abstracts/meps/v549/p231-242/> [Accessed 24 Apr. 2017].

Cummins, S., Lewis, L.J. and Egan, S., 2016. *Life on the Edge - Seabirds and Fisheries in Irish Waters.*

Dähne, M., Gilles, A., Lucke, K., Peschko, V., Adler, S., Krügel, K., Sundermeyer, J. and Siebert, U., 2013. Effects of pile-driving on harbour porpoises (Phocoena phocoena) at the first offshore wind farm in Germany. *Environmental Research Letters*, [online] 8(2), p.025002. Available at: <http://stacks.iop.org/1748-9326/8/i=2/a=025002?key=crossref.eb20ae916e74bd57c0cfe52c046dfde5>.

Dähne, M., Peschko, V., Gilles, A., Lucke, K., Adler, S., Ronnenberg, K. and Siebert, U., 2014. Marine mammals and windfarms: Effects of alpha ventus on harbour porpoises. In: *Ecological Research at the Offshore Windfarm alpha ventus*. [online] Wiesbaden: Springer Fachmedien Wiesbaden, pp.133–149. Available at: <http://link.springer.com/10.1007/978-3-658-02462-8_13> [Accessed 26 Feb. 2019].

Dähne, M., Tougaard, J., Carstensen, J., Rose, A. and Nabe-Nielsen, J., 2017. Bubble curtains attenuate noise from offshore wind farm construction and reduce temporary habitat loss for harbour porpoises. *Marine Ecology Progress Series*, [online] 580, pp.221–237. Available at: <http://www.int-res.com/abstracts/meps/v580/p221-237/>.

David, J.A., 2006. Likely sensitivity of bottlenose dolphins to pile-driving noise. *Water and Environment Journal*, [online] 20(1), pp.48–54. Available at: <http://doi.wiley.com/10.1111/j.1747-6593.2005.00023.x>.

Debusschere, E., De Coensel, B., Bajek, A., Botteldooren, D., Hostens, K., Vanaverbeke, J., Vandendriessche, S., Van Ginderdeuren, K., Vincx, M. and Degraer, S., 2014. In situ mortality experiments with juvenile sea bass (dicentrarchus labrax) in relation to impulsive sound levels caused by pile driving of windmill foundations. *PLoS ONE*, [online] 9(10), pp.1–9. Available at: <http://dx.doi.org/10.1371/journal.pone.0109280>.

Debusschere, E., Hostens, K., Adriaens, D., Ampe, B., Botteldooren, D., De Boeck, G., De Muynck, A., Sinha, A.K., Vandendriessche, S., Van Hoorebeke, L., Vincx, M. and Degraer, S., 2016. Acoustic stress responses in juvenile sea bass Dicentrarchus labrax induced by offshore pile driving. *Environmental Pollution*, [online] 208, pp.747–757. Available at: <https://linkinghub.elsevier.com/retrieve/pii/S026974911530155X>.

Department of Communications Climate Action & Environment, 2019. *Climate action plan*.

Department of Communications Energy and Natural Resources, 2014. *Offshore Renewable Energy Development Plan: A Framework for the Sustainable Development of Ireland's Offshore Renewable Energy Resource*. [online] Available at: <https://www.dccae.gov.ie/documents/20140204 DCENR - Offshore Renewable Energy Development Plan.pdf>.

DeRuiter, S.L., Southall, B.L., Calambokidis, J., Zimmer, W.M.X., Sadykova, D., Falcone, E.A., Friedlaender, A.S., Joseph, J.E., Moretti, D.J., Schorr, G.S., Thomas, L. and Tyack, P.L., 2013. First direct measurements of behavioural responses by Cuvier's beaked whales to mid-frequency active sonar. *Biology Letters*, [online] 9, p.20130223. Available at: <http://www.pubmedcentral.nih.gov/articlerender.fcgi?artid=3730631&tool=pmcentrez&rendertype=abstract>.

Dolman, S.J., Evans, P.G.H., Notarbartolo-di-Sciara, G. and Frisch, H., 2011. Active sonar, beaked whales and European regional policy. *Marine Pollution Bulletin*, [online] 63(1–4), pp.27–34. Available at: <http://dx.doi.org/10.1016/j.marpolbul.2010.03.034>.

Dooling, R.J., Leek, M.R. and Popper, A.N., 2015. Effects of noise on fishes: What we can learn from humans and birds. *Integrative Zoology*, [online] 10(1), pp.29–37. Available at: <http://doi.wiley.com/10.1111/1749-4877.12094> [Accessed 14 May 2019].

Dorschel, B., Wheeler, A.J., Monteys, X. and Verbruggen, K., 2010. *Atlas of the Deep-Water Seabed*. [online] Dordrecht: Springer Netherlands. Available at: <http://link.springer.com/10.1007/978-90-481-9376-9>.

Duarte, C.M., Pitt, K.A., Lucas, C.H., Purcell, J.E., Uye, S., Robinson, K., Brotz, L., Decker, M.B., Sutherland, K.R., Malej, A., Madin, L., Mianzan, H., Gili, J.-M., Fuentes, V., Atienza, D., Pagés, F., Breitburg, D., Malek, J., Graham, W.M. and Condon, R.H., 2013. Is global ocean sprawl a cause of jellyfish blooms? *Frontiers in Ecology and the Environment*, [online] 11(2), pp.91–97. Available at: <http://doi.wiley.com/10.1890/110246>.

Edmonds, N.J., Firmin, C.J., Goldsmith, D., Faulkner, R.C. and Wood, D.T., 2016. A review of crustacean sensitivity to high amplitude underwater noise: Data needs for effective risk assessment in relation to UK commercial species. *Marine Pollution Bulletin*, [online] 108(1–2), pp.5–11. Available at: <https://www.sciencedirect.com/science/article/pii/S0025326X16302892> [Accessed 28 May 2019].

EPA, 2019. *Ireland's Greenhouse Gas Emissions Projections 2018 - 2040*.

Erbe, C., Reichmuth, C., Cunningham, K., Lucke, K. and Dooling, R., 2016. Communication masking in marine mammals: A review and research strategy. *Marine Pollution Bulletin*, [online] 103(1–2), pp.15–38. Available at: <http://dx.doi.org/10.1016/j.marpolbul.2015.12.007>.

European Commission, 1992. Council Directive 92/43/EEC On the Conservation of Natural Habitats and of Wild Fauna and Flora. *Official Journal of the European Union*, 94(1259), pp.40–52.

European Commission, 2008. Directive 2008/56/EC of the European Parliament and of the Council, of 17 June 2008, establising a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). *Official Journal of the European Union*, pp.19–40.

European Commission, 2009. To reduce their greenhouse gas emissions to meet the Community's greenhouse gas emission reduction commitments up to 2020. *Official Journal of the European Union*, [online] 175(1), pp.136–148. Available at: <https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32009D0406&from=EN>.

European Commission, 2010. Directive 2009/147/EC of the European Parliament and of the Council on the conservation of wild birds. *Official Journal of the European Union*, [online] L 20, pp.7–25. Available at: <http://www.ecolex.org/ecolex/ledge/view/RecordDetails;jsessionid=90B0F1A51EC0A3435141652323F905BE?id=LEX-FAOC092236&index=documents>.

European Commission, 2011. Directive 2011/92/EU of the European Parliament and of the Council of 13 December 2011 on the assessment of the effects of certain public and private projects on the environment. *Official Journal of the European Union*, [online] 26, pp.1–21. Available at: <http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2012:026:0001:0021:EN:PDF>.

EWEA, 2011. *Wind in our Sails*. [online] *Wind Energy*. Available at: <http://www.ewea.org/fileadmin/ewea_documents/documents/publications/reports/23420_Offshore_report_web.pdf>.

FAO, 2018. *The State of World Fisheries and Aquaculture 2018 - Meeting the sustainable development goals*. [online] Available at: <http://www.fao.org/documents/card/en/c/I9540EN>.

Fay, R.R. and Popper, A.N., 2000. Evolution of hearing in vertebrates: the inner ears and processing.

Hearing Research, [online] 149(1–2), pp.1–10. Available at: <https://www.sciencedirect.com/science/article/pii/S0378595500001684> [Accessed 28 May 2019].

Fey, D.P., Jakubowska, M., Greszkiewicz, M., Andrulewicz, E., Otremba, Z. and Urban-Malinga, B., 2019. Are magnetic and electromagnetic fields of anthropogenic origin potential threats to early life stages of fish? *Aquatic Toxicology*, [online] 209(December 2018), pp.150–158. Available at: <https://doi.org/10.1016/j.aquatox.2019.01.023>.

Finneran, J.J., 2015. Noise-induced hearing loss in marine mammals: A review of temporary threshold shift studies from 1996 to 2015. *The Journal of the Acoustical Society of America*, 138(3).

Firth, L.B., Knights, A.M., Bridger, D., Evans, A.J., Mieszkowska, N., Moore, P.J., O'Connor, N.E., Sheehan, E. V., Thompson, R.C. and Hawkins, S.J., 2016. Ocean Sprawl: Challenges and Opportunities for Biodiversity Management in a Changing World. In: R.N. Hughes, D.J. Hughes, I.P. Smith and A.C. Dale, eds., 1st ed. [online] CRC Press, p.78. Available at: <http://www.crcnetbase.com/doi/10.1201/9781315368597-9> [Accessed 9 Jun. 2019].

Floeter, J., van Beusekom, J.E.E., Auch, D., Callies, U., Carpenter, J., Dudeck, T., Eberle, S., Eckhardt, A., Gloe, D., Hänselmann, K., Hufnagl, M., Janßen, S., Lenhart, H., Möller, K.O., North, R.P., Pohlmann, T., Riethmüller, R., Schulz, S., Spreizenbarth, S., Temming, A., Walter, B., Zielinski, O. and Möllmann, C., 2017. Pelagic effects of offshore wind farm foundations in the stratified North Sea. *Progress in Oceanography*, [online] 156, pp.154–173. Available at: <https://www.sciencedirect.com/science/article/pii/S0079661117300381> [Accessed 3 Apr. 2019].

Frid, C., Andonegi, E., Depestele, J., Judd, A., Rihan, D., Rogers, S.I. and Kenchington, E., 2012. The environmental interactions of tidal and wave energy generation devices. *Environmental Impact Assessment Review*, [online] 32(1), pp.133–139. Available at: <http://dx.doi.org/10.1016/j.eiar.2011.06.002>.

Fujii, T., 2015. Temporal variation in environmental conditions and the structure of fish assemblages around an offshore oil platform in the North Sea. *Marine Environmental Research*, [online] 108, pp.69–82. Available at: <http://dx.doi.org/10.1016/j.marenvres.2015.03.013>.

Gabel, F., Lorenz, S. and Stoll, S., 2017. Effects of ship-induced waves on aquatic ecosystems. *Science of The Total Environment*, [online] 601–602, pp.926–939. Available at: <https://www.sciencedirect.com/science/article/pii/S004896971731313X> [Accessed 13 Jun. 2019].

Gartman, V., Bulling, L., Dahmen, M., Geißler, G. and Köppel, J., 2016a. Mitigation Measures for Wildlife in Wind Energy Development, Consolidating the State of Knowledge — Part 1: Planning and Siting, Construction. *Journal of Environmental Assessment Policy and Management*, [online] 18(03), p.1650013. Available at: <http://www.worldscientific.com/doi/abs/10.1142/S1464333216500137> [Accessed 1 Apr. 2019].

Gartman, V., Bulling, L., Dahmen, M., Geißler, G. and Köppel, J., 2016b. Mitigation Measures for Wildlife in Wind Energy Development, Consolidating the State of Knowledge — Part 2: Operation, Decommissioning. *Journal of Environmental Assessment Policy and Management*, [online] 18(03), p.1650014. Available at: <http://www.worldscientific.com/doi/abs/10.1142/S1464333216500149> [Accessed 1 Apr. 2019].

German Federal Ministry for the Environment and Nuclear Safety, 2013. *Konzept für den Schutz der Schweinswale vor Schallbelastungen bei der Errichtung von Offshore-Windparks in der deutschen Nordsee (Schallschutzkonzept)*. [online] Available at: <https://www.bfn.de/fileadmin/BfN/awz/Dokumente/schallschutzkonzept_BMU.pdf>.

Gill, A.B., Bartlett, M. and Thomsen, F., 2012. Potential interactions between diadromous fishes of U.K. conservation importance and the electromagnetic fields and subsea noise from marine renewable energy developments. *Journal of Fish Biology*, [online] 81(2), pp.664–695. Available at: <http://doi.wiley.com/10.1111/j.1095-8649.2012.03374.x> [Accessed 8 Jun. 2019].

Gill, A.B., Gloyne-Philips, I., Kimber, J. and Sigray, P., 2014. Marine Renewable Energy, Electromagnetic (EM) Fields and EM-Sensitive Animals. [online] Springer, Dordrecht, pp.61–79. Available at: <http://link.springer.com/10.1007/978-94-017-8002-5_6> [Accessed 15 Mar. 2019].

Goodall, C.A., 1988. *The sensory detection of water borne vibrational stimuli and their motor effects in the Norway lobster, Nephrops norvegicus (L.)*. [online] University of Glasgow. Available at: <http://theses.gla.ac.uk/3246/> [Accessed 22 Jun. 2019].

Graham, I.M., Pirotta, E., Merchant, N.D., Farcas, A., Barton, T.R., Cheney, B., Hastie, G.D. and Thompson, P.M., 2017. Responses of bottlenose dolphins and harbor porpoises to impact and vibration piling noise during harbor construction. *Ecosphere*, [online] 8(5), p.e01793. Available at: <http://doi.wiley.com/10.1002/ecs2.1793> [Accessed 20 Feb. 2019].

de Groot, J., Campbell, M., Ashley, M. and Rodwell, L., 2014. Investigating the co-existence of fisheries and offshore renewable energy in the UK: Identification of a mitigation agenda for fishing effort displacement. *Ocean and Coastal Management*, [online] 102(PA), pp.7–18. Available at: <http://dx.doi.org/10.1016/j.ocecoaman.2014.08.013>.

van Hal, R., Griffioen, A.B. and van Keeken, O.A., 2017. Changes in fish communities on a small spatial scale, an effect of increased habitat complexity by an offshore wind farm. *Marine Environmental Research*, 126, pp.26–36.

Halvorsen, M.B., Casper, B.M., Carlson, T.J., Woodley, C.M. and Popper, A.N., 2012a. Assessment of Barotrauma Injury and Cumulative Sound Exposure Level in Salmon After Exposure to Impulsive Sound. [online] Springer, New York, NY, pp.235–237. Available at: <http://link.springer.com/10.1007/978-1-4419-7311-5_52> [Accessed 24 May 2019].

Halvorsen, M.B., Casper, B.M., Matthews, F., Carlson, T.J. and Popper, A.N., 2012b. Effects of exposure to pile-driving sounds on the lake sturgeon, Nile tilapia and hogchoker. *Proceedings of the Royal Society B: Biological Sciences*, [online] 279(1748), pp.4705–4714. Available at: <http://rspb.royalsocietypublishing.org/cgi/doi/10.1098/rspb.2012.1544> [Accessed 24 May 2019].

Halvorsen, M.B., Casper, B.M., Woodley, C.M., Carlson, T.J. and Popper, A.N., 2012c. Threshold for Onset of Injury in Chinook Salmon from Exposure to Impulsive Pile Driving Sounds. *PLoS ONE*, [online] 7(6), p.e38968. Available at: <https://dx.plos.org/10.1371/journal.pone.0038968> [Accessed 24 May 2019].

Hammar, L., 2015. Will Ocean Energy Harm Marine Ecosystems? In: B. Sandén, ed., *Systems Perspectives on Renewable Power*. [online] Göteborg, Sweden: Chalmers University of Technology, pp.128–136. Available at: <http://publications.lib.chalmers.se/records/fulltext/210512/local_210512.pdf>.

Hammar, L., Gullström, M., Dahlgren, T.G., Asplund, M.E., Goncalves, I.B. and Molander, S., 2017. Introducing ocean energy industries to a busy marine environment. *Renewable and Sustainable Energy Reviews*, [online] 74, pp.178–185. Available at: <https://linkinghub.elsevier.com/retrieve/pii/S1364032117301090> [Accessed 27 May 2019].

Hammar, L., Perry, D. and Gullström, M., 2016. Offshore Wind Power for Marine Conservation. *Open Journal of Marine Science*.

Hammar, L., Wikström, A. and Molander, S., 2014. Assessing ecological risks of offshore wind power on Kattegat cod. *Renewable Energy*, [online] 66, pp.414–424. Available at: <http://dx.doi.org/10.1016/j.renene.2013.12.024>.

Harnois, V., Smith, H.C.M., Benjamins, S. and Johanning, L., 2015. Assessment of entanglement risk to marine megafauna due to offshore renewable energy mooring systems. *International Journal of Marine Energy*, [online] 11, pp.27–49. Available at: <http://dx.doi.org/10.1016/j.ijome.2015.04.001>.

Hastie, G.D., Russell, D.J.F., Mcconnell, B., Moss, S., Thompson, D. and Janik, V.M., 2015. Sound exposure in harbour seals during the installation of an offshore wind farm: Predictions of auditory damage. *Journal of Applied Ecology*, 52(3), pp.631–640.

Hawkins, A.D. and Popper, A.N., 2017. A sound approach to assessing the impact of underwater noise on marine fishes and invertebrates. *ICES Journal of Marine Science*, 74(3), pp.635–651.

Hawkins, A.D. and Popper, A.N., 2018. Effects of Man-Made Sound on Fishes. [online] pp.145–177. Available at: <http://link.springer.com/10.1007/978-1-4939-8574-6_6> [Accessed 28 May 2019].

Hawkins, A.D., Roberts, L. and Cheesman, S., 2014. Responses of free-living coastal pelagic fish to impulsive sounds. *The Journal of the Acoustical Society of America*, [online] 135(5), pp.3101–3116. Available at: <http://asa.scitation.org/doi/10.1121/1.4870697>.

Heinisch, P. and Wiese, K., 1987. Sensitivity to Movement and Vibration of Water in the North Sea Shrimp Crangon Crangon L. *Journal of Crustacean Biology*, [online] 7(3), pp.401–413. Available at: <https://academic.oup.com/jcb/article-lookup/doi/10.2307/1548290> [Accessed 22 Jun. 2019].

Helfman, G.S., Collette, B.B., Facey, D.E. and Bowen, B.W., 2009. *The diversity of fishes: biology, evolution, and ecology*. 2nd ed. [online] Wiley-Blackwell. Available at: <https://www.wiley.com/en-ie/The+Diversity+of+Fishes:+Biology,+Evolution,+and+Ecology,+2nd+Edition-p-9781405124942> [Accessed 16 May 2019].

Higgins, P. and Foley, A., 2014. The evolution of offshore wind power in the United Kingdom. *Renewable and Sustainable Energy Reviews*, [online] 37, pp.599–612. Available at: <https://linkinghub.elsevier.com/retrieve/pii/S1364032114003839> [Accessed 13 Jun. 2019].

Hooper, T. and Austen, M., 2014. The co-location of offshore windfarms and decapod fisheries in the UK: Constraints and opportunities. *Marine Policy*, [online] 43, pp.295–300. Available at: <http://dx.doi.org/10.1016/j.marpol.2013.06.011>.

Hooper, T., Beaumont, N. and Hattam, C., 2017. The implications of energy systems for ecosystem services: A detailed case study of offshore wind. *Renewable and Sustainable Energy Reviews*, [online] 70, pp.230–241. Available at: <https://linkinghub.elsevier.com/retrieve/pii/S1364032116310206> [Accessed 27 Jan. 2019].

Houser, D.S., Champagne, C.D., Crocker, D.E., Kellar, N.M., Cockrem, J., Romano, T., Booth, R.K. and Wasser, S.K., 2016. Natural Variation in Stress Hormones, Comparisons Across Matrices, and Impacts Resulting from Induced Stress in the Bottlenose Dolphin. In: A. Popper and A. Hawkins, eds., *The Effects of Noise on Aquatic Life II*. [online] Springer, New York, NY, pp.467–471. Available at: <http://link.springer.com/10.1007/978-1-4939-2981-8_56> [Accessed 22 Jun. 2019].

Hutchison, Z., Sigray, P., He, H., Gill, A., King, J. and Gibson, C., 2018. Electromagnetic Field (EMF) Impacts on Elasmobranch (shark, rays, and skates) and American Lobster Movement and Migration from Direct Current Cables. p.254.

Iafrate, J.D., Watwood, S.L., Reyier, E.A., Scheidt, D.M., Dossot, G.A. and Crocker, S.E., 2016. Effects of pile driving on the residency and movement of tagged reef fish. *PLoS ONE*, 11(11), pp.1–17.

Igwemezie, V., Mehmanparast, A. and Kolios, A., 2019. Current trend in offshore wind energy sector and material requirements for fatigue resistance improvement in large wind turbine support structures – A review. *Renewable and Sustainable Energy Reviews*, [online] 101, pp.181–196.

Available at: <https://www.sciencedirect.com/science/article/pii/S1364032118307342#f0025> [Accessed 13 Jun. 2019].

International Energy Agency, 2018. *Renewables 2018. Market analysis and forecast from 2018 to 2023*. [online] Paris, France. Available at: <https://www.iea.org/renewables2018/> [Accessed 24 Jun. 2019].

IPCC, 2014. *Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. *Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Geneva, Switzerland.

James, V., 2013. *Marine renewable energy: A global review of the extent of marine renewable energy developments, the developing technologies and possible conservation implications for cetaceans*. 1st ed. Chippenham, United Kingdom: Whale and Dolphin Conservation.

de Jong, K., Amorim, M.C.P., Fonseca, P.J., Fox, C.J. and Heubel, K.U., 2018. Noise can affect acoustic communication and subsequent spawning success in fish. *Environmental Pollution*, [online] 237, pp.814–823. Available at: <https://www.sciencedirect.com/science/article/pii/S0269749117326064> [Accessed 21 Feb. 2019].

de Juan, D., Thrush, S. and Demestre, M., 2007. Functional changes as indicators of trawling disturbance on a benthic community located in a fishing ground (NW Mediterranean Sea). *Marine Ecology Progress Series*, [online] 334, pp.117–129. Available at: <http://www.int-res.com/abstracts/meps/v334/p117-129/> [Accessed 13 Jun. 2019].

Kastelein, R.A., 2008. Effects of vibrations on the behaviour of cockles (bivalve molluscs). *Bioacoustics*, [online] 17(1–3), pp.74–75. Available at: <http://www.tandfonline.com/doi/abs/10.1080/09524622.2008.9753770> [Accessed 22 Jun. 2019].

Kastelein, R.A., Helder-Hoek, L., Kommeren, A., Covi, J. and Gransier, R., 2018. Effect of pile-driving sounds on harbor seal (Phoca vitulina) hearing. *The Journal of the Acoustical Society of America*, [online] 143(6), pp.3583–3594. Available at: <http://asa.scitation.org/doi/10.1121/1.5040493> [Accessed 20 Feb. 2019].

Kastelein, R.A., Huybrechts, J., Covi, J. and Helder-Hoek, L., 2017a. Behavioral Responses of a Harbor Porpoise (Phocoena phocoena) to Sounds from an Acoustic Porpoise Deterrent. *Aquatic Mammals*, [online] 43(3), pp.233–244. Available at: <http://www.aquaticmammalsjournal.org/index.php?option=com_content&view=article&id=1667:behavioral-responses-of-a-harbor-porpoise-phocoena-phocoena-to-sounds-from-an-acoustic-porpoise-deterrent&catid=160&Itemid=326> [Accessed 20 Feb. 2019].

Kastelein, R.A., Jennings, N., Kommeren, A., Helder-Hoek, L. and Schop, J., 2017b. Acoustic dose-behavioral response relationship in sea bass (Dicentrarchus labrax) exposed to playbacks of pile driving sounds. *Marine Environmental Research*, [online] 130, pp.315–324. Available at: <https://www.sciencedirect.com/science/article/pii/S0141113617303732> [Accessed 4 Jun. 2019].

Kastelein, R.A., Schop, J., Hoek, L. and Covi, J., 2015. Hearing thresholds of a harbor porpoise (*Phocoena phocoena*) for narrow-band sweeps. *The Journal of the Acoustical Society of America*, [online] 138(4), pp.2508–2512. Available at: <http://asa.scitation.org/doi/10.1121/1.4932024> [Accessed 29 Mar. 2019].

Kastelein, R.A., Van de Voorde, S. and Jennings, N., 2018. Swimming Speed of a Harbor Porpoise (Phocoena phocoena) During Playbacks of Offshore Pile Driving Sounds. *Aquatic Mammals*, [online] 44(1), pp.92–99. Available at: <http://www.aquaticmammalsjournal.org/index.php?option=com_content&view=article&id=1734:s

wimming-speed-of-a-harbor-porpoise-phocoena-phocoena-during-playbacks-of-offshore-pile-driving-sounds&catid=167&Itemid=326> [Accessed 20 Feb. 2019].

Kasumyan, A.O., 2008. Sounds and sound production in fishes. *Journal of Ichthyology*, [online] 48(11), pp.981–1030. Available at: <http://link.springer.com/10.1134/S0032945208110039> [Accessed 27 May 2019].

Kavet, R., Wyman, M.T. and Klimley, A.P., 2016. Modeling Magnetic Fields from a DC Power Cable Buried Beneath San Francisco Bay Based on Empirical Measurements. *PLOS ONE*, [online] 11(2), p.e0148543. Available at: <https://dx.plos.org/10.1371/journal.pone.0148543> [Accessed 11 Jun. 2019].

Kikuchi, R., 2010. Risk formulation for the sonic effects of offshore wind farms on fish in the EU region. *Marine Pollution Bulletin*, [online] 60(2), pp.172–177. Available at: <http://dx.doi.org/10.1016/j.marpolbul.2009.09.023> [Accessed 5 Jun. 2019].

Klimley, A.P., Wyman, M.T. and Kavet, R., 2017. Chinook salmon and green sturgeon migrate through San Francisco Estuary despite large distortions in the local magnetic field produced by bridges. *PLOS ONE*, [online] 12(6), p.e0169031. Available at: <https://dx.plos.org/10.1371/journal.pone.0169031> [Accessed 11 Jun. 2019].

Koschinski, S. and Ludeman, K., 2013. *Development of Noise Mitigation Measures in Offshore Wind Farm Construction*. [online] Nehmten and Hamburg, Germany. Available at: <https://www.researchgate.net/publication/308110557_Development_of_Noise_Mitigation_Measures_in_Offshore_Wind_Farm_Construction>.

Krägefsky, S., 2014. Effects of the alpha ventus offshore test site on pelagic fish. In: *Ecological Research at the Offshore Windfarm alpha ventus*. [online] Wiesbaden: Springer Fachmedien Wiesbaden, pp.83–94. Available at: <http://link.springer.com/10.1007/978-3-658-02462-8_10> [Accessed 6 Jun. 2019].

Krone, R., Dederer, G., Kanstinger, P., Krämer, P., Schneider, C. and Schmalenbach, I., 2017. Mobile demersal megafauna at common offshore wind turbine foundations in the German Bight (North Sea) two years after deployment - increased production rate of Cancer pagurus. *Marine Environmental Research*, [online] 123, pp.53–61. Available at: <https://www.sciencedirect.com/science/article/pii/S0141113616303014> [Accessed 8 Mar. 2019].

Krone, R., Gutow, L., Brey, T., Dannheim, J. and Schröder, A., 2013a. Mobile demersal megafauna at artificial structures in the German Bight - Likely effects of offshore wind farm development. *Estuarine, Coastal and Shelf Science*, [online] 125, pp.1–9. Available at: <http://dx.doi.org/10.1016/j.ecss.2013.03.012>.

Krone, R., Gutow, L., Joschko, T.J. and Schröder, A., 2013b. Epifauna dynamics at an offshore foundation – Implications of future wind power farming in the North Sea. *Marine Environmental Research*, [online] 85, pp.1–12. Available at: <https://www.sciencedirect.com/science/article/pii/S014111361200222X> [Accessed 14 May 2019].

Langhamer, O., 2012. Artificial Reef Effect in relation to Offshore Renewable Energy Conversion: State of the Art. *The Scientific World Journal*, [online] 2012, pp.1–8. Available at: <http://www.hindawi.com/journals/tswj/2012/386713/>.

Langhamer, O., Dahlgren, T.G. and Rosenqvist, G., 2018. Effect of an offshore wind farm on the viviparous eelpout: Biometrics, brood development and population studies in Lillgrund, Sweden. *Ecological Indicators*, [online] 84, pp.1–6. Available at: <https://www.sciencedirect.com/science/article/pii/S1470160X17305253> [Accessed 3 Apr. 2019].

Langhamer, O. and Wilhelmsson, D., 2009. Colonisation of fish and crabs of wave energy foundations and the effects of manufactured holes - A field experiment. *Marine Environmental Research*, [online] 68(4), pp.151–157. Available at: <http://dx.doi.org/10.1016/j.marenvres.2009.06.003>.

Langhamer, O., Wilhelmsson, D. and Engström, J., 2009. Artificial reef effect and fouling impacts on offshore wave power foundations and buoys – a pilot study. *Estuarine, Coastal and Shelf Science*, [online] 82(3), pp.426–432. Available at: <https://www.sciencedirect.com/science/article/pii/S0272771409000626> [Accessed 7 Jun. 2019].

Lindeboom, H.J., Kouwenhoven, H.J., Bergman, M.J.N., Bouma, S., Brasseur, S., Daan, R., Fijn, R.C., de Haan, D., Dirksen, S., van Hal, R., Hille Ris Lambers, R., ter Hofstede, R., Krijgsveld, K.L., Leopold, M. and Scheidat, M., 2011. Short-term ecological effects of an offshore wind farm in the Dutch coastal zone; a compilation. *Environmental Research Letters*, [online] 6(035101), p.13. Available at: <http://stacks.iop.org/1748-9326/6/i=3/a=035101?key=crossref.94f6f6176b9f45ddab616c8da9b7b2a0> [Accessed 3 Apr. 2019].

Little, L.R., Smith, A.D.M., McDonald, A.D., Punt, A.E., Mapstone, B.D., Pantus, F. and Davies, C.R., 2005. Effects of size and fragmentation of marine reserves and fisher infringement on the catch and biomass of coral trout, Plectropomus leopardus, on the Great Barrier Reef, Australia. *Fisheries Management and Ecology*, [online] 12(3), pp.177–188. Available at: <http://doi.wiley.com/10.1111/j.1365-2400.2005.00440.x> [Accessed 8 Jun. 2019].

Lovich, J.E. and Ennen, J.R., 2013. Assessing the state of knowledge of utility-scale wind energy development and operation on non-volant terrestrial and marine wildlife. *Applied Energy*, [online] 103(April 2012), pp.52–60. Available at: <http://dx.doi.org/10.1016/j.apenergy.2012.10.001>.

Lucke, K., Siebert, U., Lepper, P.A. and Blanchet, M.-A., 2009. Temporary shift in masked hearing thresholds in a harbor porpoise (Phocoena phocoena) after exposure to seismic airgun stimuli. *The Journal of the Acoustical Society of America*, [online] 125(6), pp.4060–4070. Available at: <http://scitation.aip.org/content/asa/journal/jasa/125/6/10.1121/1.3117443>.

Luís, A.R., Couchinho, M.N. and dos Santos, M.E., 2014. Changes in the acoustic behavior of resident bottlenose dolphins near operating vessels. *Marine Mammal Science*, [online] 30(4), pp.1417–1426. Available at: <http://doi.wiley.com/10.1111/mms.12125> [Accessed 1 Mar. 2019].

Macfadyen, G., Huntington, T. and Cappell, R., 2009. *Abandoned, lost or otherwise discarded fishing gear*. [online] *FAO Fisheries and Aquaculture Technical Paper 523*, Available at: <http://www.unep.org/regionalseas/marinelitter/publications/default.asp>.

Madsen, P., Wahlberg, M., Tougaard, J., Lucke, K. and Tyack, P., 2006. Wind turbine underwater noise and marine mammals: implications of current knowledge and data needs. *Marine Ecology Progress Series*, [online] 309, pp.279–295. Available at: <http://www.int-res.com/abstracts/meps/v309/p279-295/> [Accessed 29 Mar. 2019].

Magnhagen, C., Johansson, K. and Sigray, P., 2017. Effects of motorboat noise on foraging behaviour in Eurasian perch and roach: a field experiment. *Marine Ecology Progress Series*, [online] 564, pp.115–125. Available at: <http://www.int-res.com/abstracts/meps/v564/p115-125/> [Accessed 2 Apr. 2019].

Mann, R., 2006. The local costs to ecological services associated with high seas global transport. In: J. Davenport and J.L. Davenport, eds., *The Ecology of Transportation: Managing Mobility for the Environment. Environmental Pollution, vol 10*. [online] Springer, Dordrecht, pp.25–38. Available at: <http://link.springer.com/10.1007/1-4020-4504-2_2> [Accessed 14 Jun. 2019].

Marine Institute, 2018. *The Stock Book. Annual Review of Fish Stocks in 2018 with Management Advice for 2019.* Galway Ireland.

Marine Institute and Board Iascaigh Mhara, 2017. *Shellfish Stocks and Fisheries. Review 2016-2017. An assessment of seleted stocks.* Galway, Ireland.

Marino, L., Rose, N.A., Visser, I.N., Rally, H., Ferdowsian, H. and Slootsky, V., 2019. The Harmful Effects of Captivity and Chronic Stress on the Well-being of Orcas (Orcinus orca). *Journal of Veterinary Behavior*. [online] Available at: <https://www.sciencedirect.com/science/article/pii/S1558787819300164> [Accessed 22 Jun. 2019].

McClanahan, T.R. and Mangi, S., 2000. Spillover of Exploitable Fishes from a Marine Park and Its Effect on the Adjacent Fishery. *Ecological Applications*, [online] 10(6), p.1792. Available at: <http://ze5mw2yz8y.search.serialssolutions.com//?url_ver=Z39.88-2003&ctx_ver=Z39.88-2003&ctx_enc=info:ofi/enc:UTF-8&rft_id=info:doi/10.1890%2F1051-0761%282000%29010%5B1792%3ASOEFFA%5D2.0.CO%3B2&rft_val_fmt=info:ofi/fmt:kev:mtx:journal&rft.genre=article&rft> [Accessed 13 Jun. 2019].

McConnell, B., Lonergan, M. and Dietz, R., 2012. *Interactions between seals and offshore wind farms. Marine Estate research report, The Crown Estate, London*.

Merchant, N.D., Pirotta, E., Barton, T.R. and Thompson, P.M., 2014. Monitoring ship noise to assess the impact of coastal developments on marine mammals. *Marine Pollution Bulletin*, [online] 78(1–2), pp.85–95. Available at: <https://www.sciencedirect.com/science/article/pii/S0025326X13006802> [Accessed 21 Feb. 2019].

Methratta, E.T. and Dardick, W.R., 2019. Meta-Analysis of Finfish Abundance at Offshore Wind Farms. *Reviews in Fisheries Science and Aquaculture*, [online] 27(2), pp.242–260. Available at: <https://doi.org/10.1080/23308249.2019.1584601>.

Mikkelsen, L., Johnson, M., Wisniewska, D.M., van Neer, A., Siebert, U., Madsen, P.T. and Teilmann, J., 2019. Long-term sound and movement recording tags to study natural behavior and reaction to ship noise of seals. *Ecology and Evolution*, [online] (October 2018), p.ece3.4923. Available at: <https://onlinelibrary.wiley.com/doi/abs/10.1002/ece3.4923>.

Mikkelsen, L., Mouritsen, K.N., Dahl, K., Teilmann, J. and Tougaard, J., 2013. Re-established stony reef attracts harbour porpoises Phocoena phocoena. *Marine Ecology Progress Series*, [online] 481, pp.239–248. Available at: <http://www.int-res.com/abstracts/meps/v481/p239-248/> [Accessed 20 Mar. 2019].

Minchin, D., 2006. The transport and the spread of living aquatic species. In: J. Davenport and J.L. Davenport, eds., *The Ecology of Transportation: Managing Mobility for the Environment. Environmental Pollution, vol 10*. [online] Springer, Dordrecht, pp.77–97. Available at: <http://link.springer.com/10.1007/1-4020-4504-2_5> [Accessed 14 Jun. 2019].

Minchin, D., FLOERL, O., Savini, D. and Occhipinti-Ambrogi, A., 2006. Small craft and the spread of exotic species. In: J. Davenport and J.L. Davenport, eds., *The Ecology of Transportation: Managing Mobility for the Environment. Environmental Pollution, vol 10*. [online] Springer, Dordrecht, pp.99–118. Available at: <http://link.springer.com/10.1007/1-4020-4504-2_6> [Accessed 14 Jun. 2019].

Miralles, L., Gomez-Agenjo, M., Rayon-Viña, F., Gyraitė, G. and Garcia-Vazquez, E., 2018. Alert calling in port areas: Marine litter as possible secondary dispersal vector for hitchhiking invasive species. *Journal for Nature Conservation*, [online] 42, pp.12–18. Available at: <https://www.sciencedirect.com/science/article/pii/S161713811730198X> [Accessed 10 Sep. 2018].

Monteclaro, H.M., Anraku, K. and Matsuoka, T., 2010. Response properties of crayfish antennules to hydrodynamic stimuli: functional differences in the lateral and medial flagella. *The Journal of experimental biology*, [online] 213(Pt 21), pp.3683–91. Available at: <http://www.ncbi.nlm.nih.gov/pubmed/20952616> [Accessed 22 Jun. 2019].

Mooney, T.A., Nachtigall, P.E., Breese, M., Vlachos, S. and Au, W.W.L., 2009. Predicting temporary threshold shifts in a bottlenose dolphin (Tursiops truncatus): The effects of noise level and duration. *The Journal of the Acoustical Society of America*, [online] 125(3), pp.1816–1826. Available at: <http://scitation.aip.org/content/asa/journal/jasa/125/3/10.1121/1.3068456>.

Moore, M., Andrews, R., Austin, T., Bailey, J., Costidis, A., George, C., Jackson, K., Pitchford, T., Landry, S., Ligon, A., Mclellan, W., Morin, D., Smith, J., Rotstein, D., Rowles, T., Slay, C. and Walsh, M., 2013. Rope trauma, sedation, disentanglement, and monitoring-tag associated lesions in a terminally entangled North Atlantic right whale (Eubalaena glacialis). *Marine Mammal Science*, 29(2), pp.98–113.

Mueller-Blenkle, C., Jones, E., Reid, D., Lüdemann, K., Kafemann, R. and Elepfandt, A., 2008. Reactions of cod gadus morhua to low- frequency sound resembling offshore wind turbine noise emissions. *Bioacoustics*, [online] 17(1–3), pp.207–209. Available at: <http://www.tandfonline.com/doi/abs/10.1080/09524622.2008.9753819> [Accessed 14 Feb. 2019].

Mueller-Blenkle, C., Mcgregor, P.K., Gill, A.B., Andersson, M.H., Metcalfe, J., Bendall, V., Sigray, P., Wood, D.T. and Thomsen, F., 2010. *Effects of Pile-Driving Noise on the Behaviour of Marine Fish.* [online] *COWRIE Ref: Fish 06-08, Technical Report*, Available at: <http://mhk.pnl.gov/wiki/index.php/Effects_of_Pile-Driving_Noise_on_the_Behavior_of_Marine_Fish> [Accessed 3 Apr. 2019].

Nabe-Nielsen, J., van Beest, F.M., Grimm, V., Sibly, R.M., Teilmann, J. and Thompson, P.M., 2018. Predicting the impacts of anthropogenic disturbances on marine populations. *Conservation Letters*, 11(5), pp.1–8.

Nabe-Nielsen, J., Sibly, R.M., Tougaard, J., Teilmann, J. and Sveegaard, S., 2014. Effects of noise and by-catch on a Danish harbour porpoise population. *Ecological Modelling*, [online] 272, pp.242–251. Available at: <https://www.sciencedirect.com/science/article/pii/S0304380013004675> [Accessed 1 Mar. 2019].

National Marine Fisheries Service, 2016. *Technical Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing: Underwater Acoustic Thresholds for Onset of Permanent and Temporary Threshold Shifts*. [online] *U.S. Department of Commerce. NOAA Technical Memorandum NMFS-OPR-55*. Available at: <https://repository.library.noaa.gov/view/noaa/15850>.

National Parks & Wildlife Service, 2014. *Guidance to Manage the Risk to Marine Mammals from Man-made Sound Sources in Irish Waters*. [online] Department of Arts, Heritage and the Gaeltacht. Available at: <https://www.npws.ie/sites/default/files/general/Underwater sound guidance_Jan 2014.pdf>.

National Research Council, 2003. *Ocean Noise and Marine Mammals*. [online] *Ocean Noise and Marine Mammals*. Washington, D.C.: National Academies Press. Available at: <http://www.ncbi.nlm.nih.gov/pubmed/25057640> [Accessed 15 Jun. 2019].

Nedelec, S.L., Mills, S.C., Lecchini, D., Nedelec, B., Simpson, S.D. and Radford, A.N., 2016. Repeated exposure to noise increases tolerance in a coral reef fish. *Environmental Pollution*, [online] 216, pp.428–436. Available at: <https://www.sciencedirect.com/science/article/pii/S0269749116304468> [Accessed 14 May 2019].

Nedelec, S.L., Radford, A.N., Pearl, L., Nedelec, B., McCormick, M.I., Meekan, M.G. and Simpson, S.D., 2017. Motorboat noise impacts parental behaviour and offspring survival in a reef fish. *Proceedings of the Royal Society B: Biological Sciences*, [online] 284(1856), p.20170143. Available at: <http://rspb.royalsocietypublishing.org/lookup/doi/10.1098/rspb.2017.0143> [Accessed 5 Jun. 2019].

Nedwell, J. and Howell, D., 2004. *A review of offshore windfarm related underwater noise sources*. [online] *COWRIE Tech. Rep. 544R0308*. Hampshire, UK. Available at: <http://www.ncbi.nlm.nih.gov/pubmed/21498835>.

Nehls, G., Rose, A., Diederichs, A., Bellmann, M. and Pehlke, H., 2016. Noise Mitigation During Pile Driving Efficiently Reduces Disturbance of Marine Mammals. In: A.N. Popper and A.D. Hawkins, eds., *The Effects of Noise on Aquatic Life II*. [online] New York: Springer-Verlag, pp.755–762. Available at: <http://link.springer.com/10.1007/978-1-4939-2981-8_92> [Accessed 20 Feb. 2019].

Neo, Y.Y., Hubert, J., Bolle, L., Winter, H.V., ten Cate, C. and Slabbekoorn, H., 2016. Sound exposure changes European seabass behaviour in a large outdoor floating pen: Effects of temporal structure and a ramp-up procedure. *Environmental Pollution*, [online] 214, pp.26–34. Available at: <https://www.sciencedirect.com/science/article/pii/S0269749116302640> [Accessed 3 Jun. 2019].

Neo, Y.Y., Hubert, J., Bolle, L.J., Winter, H.V. and Slabbekoorn, H., 2018. European seabass respond more strongly to noise exposure at night and habituate over repeated trials of sound exposure. *Environmental Pollution*, [online] 239, pp.367–374. Available at: <https://www.sciencedirect.com/science/article/pii/S0269749117341817> [Accessed 3 Jun. 2019].

Neo, Y.Y., Seitz, J., Kastelein, R.A., Winter, H.V., ten Cate, C. and Slabbekoorn, H., 2014. Temporal structure of sound affects behavioural recovery from noise impact in European seabass. *Biological Conservation*, [online] 178, pp.65–73. Available at: <https://www.sciencedirect.com/science/article/pii/S0006320714002742> [Accessed 14 May 2019].

Neo, Y.Y., Ufkes, E., Kastelein, R.A., Winter, H. V., ten Cate, C. and Slabbekoorn, H., 2015. Impulsive sounds change European seabass swimming patterns: Influence of pulse repetition interval. *Marine Pollution Bulletin*, [online] 97(1–2), pp.111–117. Available at: <http://dx.doi.org/10.1016/j.marpolbul.2015.06.027>.

Ng, A.K.Y. and Song, S., 2010. The environmental impacts of pollutants generated by routine shipping operations on ports. *Ocean & Coastal Management*, [online] 53(5–6), pp.301–311. Available at: <https://www.sciencedirect.com/science/article/pii/S0964569110000372> [Accessed 14 Jun. 2019].

Nichols, T.A., Anderson, T.W. and Širović, A., 2015. Intermittent Noise Induces Physiological Stress in a Coastal Marine Fish. *PLOS ONE*, [online] 10(9), p.e0139157. Available at: <http://dx.plos.org/10.1371/journal.pone.0139157> [Accessed 4 Jun. 2019].

Normandeau, Exponent, Tricas, T. and Gill, A., 2011. *Effects of EMFs From Undersea Power Cables on Elasmobranchs and other Marine Species. OCS Study BOEMRE 2011-09*. [online] Camarilli, CA. Available at: <http://www.gomr.boemre.gov/homepg/espis/espisfront.asp>.

O'Higgins, T. and O'Hagan, A.M., 2019. A return to the tragedy of the commons? Brexit trade-offs and spatial analysis, an Irish perspective. *Marine Policy*, [online] 106, p.103524. Available at: <https://www.sciencedirect.com/science/article/pii/S0308597X19300181> [Accessed 21 Jun. 2019].

O'Sullivan, D., Lordan, C., Doyle, J., Berry, A. and Lyons, K., 2014. *Sediment characteristics and local hydrodynamics and their influence on the population of Nephrops around Ireland*. *Irish Fisheries Investigations No. 26*. Rinville, Oranmore, Co. Galway.

Oakley, J.A., Williams, A.T. and Thomas, T., 2017. Reactions of harbour porpoise (Phocoena phocoena) to vessel traffic in the coastal waters of South West Wales, UK. *Ocean & Coastal Management*, [online] 138, pp.158–169. Available at: <https://www.sciencedirect.com/science/article/pii/S0964569117300170> [Accessed 21 Feb. 2019].

Oh, K.-Y., Nam, W., Ryu, M.S., Kim, J.-Y. and Epureanu, B.I., 2018. A review of foundations of offshore wind energy convertors: Current status and future perspectives. *Renewable and Sustainable Energy Reviews*, [online] 88, pp.16–36. Available at: <https://www.sciencedirect.com/science/article/pii/S136403211830025X> [Accessed 23 Jun. 2019].

Overholtz, W.J. and Friedland, K.D., 2002. Recovery of the Gulf of Maine – Georges Bank Atlantic herring (Clupea harengus) complex: perspectives based on bottom trawl survey data. *Fisheries Bulletin*, 100(3), pp.593–608.

Overholtz, W.J. and Link, J.S., 2007. Consumption impacts by marine mammals , fish , and seabirds on the Gulf of Maine – Georges Bank Atlantic herring (Clupea harengus) complex during the years 1977 – 2002. *ICES Journal of Marine Science*, 64, pp.83–96.

Parry, G.D. and Gason, A., 2006. The effect of seismic surveys on catch rates of rock lobsters in western Victoria, Australia. *Fisheries Research*, [online] 79(3), pp.272–284. Available at: <https://www.sciencedirect.com/science/article/pii/S0165783606001330> [Accessed 22 Jun. 2019].

Parsons, E.C.M., Dolman, S.J., Wright, A.J., Rose, N.A. and Burns, W.C.G., 2008. Navy sonar and cetaceans: Just how much does the gun need to smoke before we act? *Marine Pollution Bulletin*, [online] 56(7), pp.1248–1257. Available at: <http://linkinghub.elsevier.com/retrieve/pii/S0025326X08002221>.

Payne, J., Andrews, C., Fancey, L., White, D. and Christian, J., 2008. Potential Effects of Seismic Energy on Fish and Shellfish: An Update Since. [online] 3848. Available at: <http://biblio.uqar.ca/archives/30137336.pdf>.

Peng, C., Zhao, X. and Liu, G., 2015. *Noise in the sea and its impacts on marine organisms*. *International Journal of Environmental Research and Public Health*, .

Pershing, A.J., Alexander, M.A., Hernandez, C.M., Kerr, L.A., Le Bris, A., Mills, K.E., Nye, J.A., Record, N.R., Scannell, H.A., Scott, J.D., Sherwood, G.D. and Thomas, A.C., 2015. Slow adaptation in the face of rapid warming leads to collapse of the Gulf of Maine cod fishery. *Science*, [online] 350(6262), pp.809–812. Available at: <http://www.sciencemag.org/content/350/6262/809.abstract%5Cnhttp://www.sciencemag.org/cgi/doi/10.1126/science.aac9819>.

Pikitch, E., Boersma, P.D., Boyd, I., Conover, D., Cury, P., Essington, T., Heppell, S., Houde, E., Mangel, M., Pauly, D., Plaganyi-Lloyd, E., Sainsbury, K. and Steneck, R., 2012. *Little Fish, Big Impact: Managing a Crucial Link in Ocean Food Webs*. [online] Washington, DC: Lenfest Ocean Program. Available at: <https://publications.csiro.au/rpr/pub?list=BRO&pid=csiro:EP124359&sb=RECENT&n=10&rpp=10&page=292&tr=4231&dr=all&dc4.browseYear=2012> [Accessed 21 Jun. 2019].

Pine, M.K., Jeffs, A.G., Wang, D. and Radford, C.A., 2016. The potential for vessel noise to mask biologically important sounds within ecologically significant embayments. *Ocean and Coastal Management*, 127.

Pirotta, E., Merchant, N.D., Thompson, P.M., Barton, T.R. and Lusseau, D., 2015. Quantifying the effect of boat disturbance on bottlenose dolphin foraging activity. *Biological Conservation*, [online] 181, pp.82–89. Available at: <https://www.sciencedirect.com/science/article/pii/S0006320714004200> [Accessed 21 Feb. 2019].

Popper, A.N. and Fay, R.R., 1993. Sound Detection and Processing by Fish: Critical Review and Major Research Questions (Part 1 of 2). *Brain, Behavior and Evolution*, [online] 41(1), pp.14–25. Available at: <http://www.ncbi.nlm.nih.gov/pubmed/8431753> [Accessed 28 May 2019].

Popper, A.N. and Fay, R.R., 2011. Rethinking sound detection by fishes. *Hearing Research*, [online] 273(1–2), pp.25–36. Available at: <https://linkinghub.elsevier.com/retrieve/pii/S037859550900313X> [Accessed 7 May 2019].

Popper, A.N. and Hastings, M.C., 2009a. The effects of anthropogenic sources of sound on fishes. *Journal of Fish Biology*, [online] 75, pp.455–489. Available at: <http://doi.wiley.com/10.1111/j.1095-8649.2009.02319.x> [Accessed 2 Apr. 2019].

Popper, A.N. and Hastings, M.C., 2009b. The effects of human-generated sound on fish. *Integrative Zoology*, [online] 4(1), pp.43–52. Available at: <http://doi.wiley.com/10.1111/j.1749-4877.2008.00134.x>.

Popper, A.N. and Hawkins, A.D., 2019. An overview of fish bioacoustics and the impacts of anthropogenic sounds on fishes. *Journal of Fish Biology*, [online] pp.1–22. Available at: <https://onlinelibrary.wiley.com/doi/abs/10.1111/jfb.13948> [Accessed 7 May 2019].

Popper, A.N., Hawkins, A.D., Fay, R.R., Mann, D.A., Bartol, S., Carlson, T.J., Coombs, S., Ellison, W.T., Gentry, R.L., Halvorsen, M.B., Løkkeborg, S., Rogers, P.H., Southall, B.L., Zeddies, D.G. and Tavolga, W.N., 2014. *ASA S3/SC1.4 TR-2014 Sound Exposure Guidelines for Fishes and Sea Turtles: A Technical Report prepared by ANSI-Accredited Standards Committee S3/SC1 and registered with ANSI*. SpringerBriefs in Oceanography. [online] Cham: Springer International Publishing. Available at: <http://link.springer.com/10.1007/978-3-319-06659-2>.

Popper, A.N., Hawkins, A.D. and Halvorsen, M.B., 2019. *Anthropogenic Sound and Fishes*. *WSDOT Research Report*. Olympia, WA: WSDOT Research Report.

Popper, A.N., Salmon, M. and Horch, K.W., 2001. Acoustic detection and communication by decapod crustaceans. *Journal of Comparative Physiology A: Sensory, Neural, and Behavioral Physiology*, [online] 187(2), pp.83–89. Available at: <http://link.springer.com/10.1007/s003590100184> [Accessed 22 Jun. 2019].

Popper, A.N., Smith, M.E., Cott, P.A., Hanna, B.W., MacGillivray, A.O., Austin, M.E. and Mann, D.A., 2005. Effects of exposure to seismic airgun use on hearing of three fish species. *The Journal of the Acoustical Society of America*, [online] 117(6), pp.3958–3971. Available at: <http://asa.scitation.org/doi/10.1121/1.1904386> [Accessed 24 May 2019].

Purser, J., Bruintjes, R., Simpson, S.D. and Radford, A.N., 2016. Condition-dependent physiological and behavioural responses to anthropogenic noise. *Physiology & Behavior*, [online] 155, pp.157–161. Available at: <https://www.sciencedirect.com/science/article/pii/S0031938415302055> [Accessed 3 Jun. 2019].

Purser, J. and Radford, A.N., 2011. Acoustic Noise Induces Attention Shifts and Reduces Foraging Performance in Three-Spined Sticklebacks (Gasterosteus aculeatus). *PLoS ONE*, [online] 6(2), p.e17478. Available at: <https://dx.plos.org/10.1371/journal.pone.0017478> [Accessed 4 Jun. 2019].

Rabaut, M., Van de Moortel, L., Vincx, M. and Degraer, S., 2010. Biogenic reefs as structuring factor in Pleuronectes platessa (Plaice) nursery. *Journal of Sea Research*, [online] 64(1–2), pp.102–106. Available at: <https://www.sciencedirect.com/science/article/pii/S1385110109001075> [Accessed 13 Jun. 2019].

Radford, A.N., Kerridge, E. and Simpson, S.D., 2014. Acoustic communication in a noisy world: can fish compete with anthropogenic noise? *Behavioral Ecology*, [online] 25(5), pp.1022–1030. Available at: <https://academic.oup.com/beheco/article-lookup/doi/10.1093/beheco/aru029> [Accessed 2 Apr. 2019].

Radford, A.N., Lèbre, L., Lecaillon, G., Nedelec, S.L. and Simpson, S.D., 2016. Repeated exposure reduces the response to impulsive noise in European seabass. *Global Change Biology*, [online] 22(10), pp.3349–3360. Available at: <http://doi.wiley.com/10.1111/gcb.13352> [Accessed 2 Apr. 2019].

Raoux, A., Dambacher, J.M., Pezy, J.-P., Mazé, C., Dauvin, J.-C. and Niquil, N., 2018. Assessing cumulative socio-ecological impacts of offshore wind farm development in the Bay of Seine (English Channel). *Marine Policy*, [online] 89, pp.11–20. Available at: <https://linkinghub.elsevier.com/retrieve/pii/S0308597X1730444X>.

Raoux, A., Lassalle, G., Pezy, J.P., Tecchio, S., Safi, G., Ernande, B., Mazé, C., Loc'h, F. Le, Lequesne, J., Girardin, V., Dauvin, J.C. and Niquil, N., 2019. Measuring sensitivity of two OSPAR indicators for a coastal food web model under offshore wind farm construction. *Ecological Indicators*, 96(July 2018), pp.728–738.

Raoux, A., Tecchio, S., Pezy, J.P., Lassalle, G., Degraer, S., Wilhelmsson, D., Cachera, M., Ernande, B., Le Guen, C., Haraldsson, M., Grangeré, K., Le Loc'h, F., Dauvin, J.C. and Niquil, N., 2017. Benthic and fish aggregation inside an offshore wind farm: Which effects on the trophic web functioning? *Ecological Indicators*, 72, pp.33–46.

Reid, J.B., Evans, P.G.H. and Northridge, S.P., 2003. *Atlas of cetacean distribution in north-west European waters,*. [online] Peterborough: Joint Nature Conservation Committee. Available at: <http://doi.wiley.com/10.1002/aqc.615>.

Reubens, J.T., Braeckman, U., Vanaverbeke, J., Van Colen, C., Degraer, S. and Vincx, M., 2013a. Aggregation at windmill artificial reefs: CPUE of Atlantic cod (Gadus morhua) and pouting (Trisopterus luscus) at different habitats in the Belgian part of the North Sea. *Fisheries Research*, [online] 139, pp.28–34. Available at: <https://www.sciencedirect.com/science/article/pii/S016578361200327X> [Accessed 16 May 2019].

Reubens, J.T., Degraer, S. and Vincx, M., 2011. Aggregation and feeding behaviour of pouting (Trisopterus luscus) at wind turbines in the Belgian part of the North Sea. *Fisheries Research*, [online] 108(1), pp.223–227. Available at: <http://dx.doi.org/10.1016/j.fishres.2010.11.025>.

Reubens, J.T., Degraer, S. and Vincx, M., 2014. The ecology of benthopelagic fishes at offshore wind farms: A synthesis of 4 years of research. *Hydrobiologia*, 727(1), pp.121–136.

Reubens, J.T., Pasotti, F., Degraer, S. and Vincx, M., 2013b. Residency, site fidelity and habitat use of atlantic cod (Gadus morhua) at an offshore wind farm using acoustic telemetry. *Marine Environmental Research*, [online] 90, pp.128–135. Available at: <http://dx.doi.org/10.1016/j.marenvres.2013.07.001>.

Reubens, J.T., De Rijcke, M., Degraer, S. and Vincx, M., 2014. Diel variation in feeding and movement patterns of juvenile Atlantic cod at offshore wind farms. *Journal of Sea Research*, [online] 85, pp.214–221. Available at: <http://dx.doi.org/10.1016/j.seares.2013.05.005>.

Reubens, J.T., Vandendriessche, S., Zenner, A.N., Degraer, S. and Vincx, M., 2013c. Offshore wind farms as productive sites or ecological traps for gadoid fishes? - Impact on growth, condition index and diet composition. *Marine Environmental Research*, 90, pp.66–74.

Reyff, J.A., 2016. Underwater Sound Propagation from Marine Pile Driving. In: A. Popper and A. Hawkins, eds., *The Effects of Noise on Aquatic Life II*. [online] Springer, New York, NY, pp.909–915. Available at: <http://link.springer.com/10.1007/978-1-4939-2981-8_112> [Accessed 22 Jun. 2019].

Richardson, W.J., Greene, C.R., Malme, C.I., Thomson, D.H., Moore, S.E. and Wiirsig, B., 2013. *Marine Mammals and Noise*. *Marine Mammals and Noise*. Elsevier Inc.

Roach, M., Cohen, M., Forster, R., Revill, A.S. and Johnson, M., 2018. The effects of temporary exclusion of activity due to wind farm construction on a lobster (Homarus gammarus) fishery suggests a potential management approach. *ICES Journal of Marine Science*, 75(4), pp.1416–1426.

Roberts, L., Cheesman, S., Elliott, M. and Breithaupt, T., 2016. Sensitivity of Pagurus bernhardus (L.) to substrate-borne vibration and anthropogenic noise. *Journal of Experimental Marine Biology and Ecology*, [online] 474, pp.185–194. Available at: <https://www.sciencedirect.com/science/article/pii/S0022098115300277> [Accessed 3 Jun. 2019].

Roberts, L., Pérez-Domínguez, R. and Elliott, M., 2016. Use of baited remote underwater video (BRUV) and motion analysis for studying the impacts of underwater noise upon free ranging fish and implications for marine energy management. *Marine Pollution Bulletin*, [online] 112(1–2), pp.75–85. Available at: <https://www.sciencedirect.com/science/article/pii/S0025326X16306713> [Accessed 4 Jun. 2019].

Rogan, E. and Berrow, S.D., 1995. The management of Irish waters as a whale and dolphin sanctuary. *Developments in Marine Biology*, [online] 4, pp.671–681. Available at: <https://www.sciencedirect.com/science/article/pii/S0163699506800661?via%3Dihub> [Accessed 16 Jun. 2019].

Rogan, E., Breen, P., Mackey, M., Cañadas, A., Scheidat, M., Geelhoed, S. and Jessopp, M., 2018. *Aerial Surveys of Cetaceans and Seabirds in Irish waters : Occurrence, distribution and abundance in 2015-2017*. Dublin, Ireland.

Rose, G. a. and Rowe, S., 2015. Northern cod comeback. *Canadian Journal of Fisheries and Aquatic Sciences*, [online] 72(12), pp.1789–1798. Available at: <http://www.nrcresearchpress.com/doi/10.1139/cjfas-2015-0346>.

Rose, G. and O'Driscoll, R.L., 2002. Capelin are good for cod: can the northern stock rebuild without them? *ICES Journal of Marine Science*, [online] 59(5), pp.1018–1026. Available at: <https://academic.oup.com/icesjms/article-lookup/doi/10.1006/jmsc.2002.1252> [Accessed 23 Jun. 2019].

Rose, G.A., 2004. Reconciling overfishing and climate change with stock dynamics of Atlantic cod (*Gadus morhua*) over 500 years. *Canadian Journal of Fisheries and Aquatic Sciences*, [online] 61(9), pp.1553–1557. Available at: <http://www.nrcresearchpress.com/doi/10.1139/f04-173> [Accessed 23 Jun. 2019].

Ross, D., 2005. Ship Sources of Ambient Noise. *IEEE Journal of Oceanic Engineering*, [online] 30(2), pp.257–261. Available at: <http://ieeexplore.ieee.org/document/1522502/> [Accessed 5 Jun. 2019].

Russell, D.J.F., Brasseur, S.M.J.M., Thompson, D., Hastie, G.D., Janik, V.M., Aarts, G., McClintock, B.T., Matthiopoulos, J., Moss, S.E.W. and McConnell, B., 2014. Marine mammals trace anthropogenic structures at sea. *Current Biology*, 24(14), pp.638–639.

Russell, D.J.F., Hastie, G.D., Thompson, D., Janik, V.M., Hammond, P.S., Scott-Hayward, L.A.S., Matthiopoulos, J., Jones, E.L. and McConnell, B.J., 2016. Avoidance of wind farms by harbour seals is limited to pile driving activities. *Journal of Applied Ecology*, 53(6), pp.1642–1652.

Ryan, C., Berrow, S.D., Mchugh, B., O'Donnell, C., Trueman, C.N. and O'Connor, I., 2014. *Prey preferences of sympatric fin (Balaenoptera physalus) and humpback (Megaptera novaeangliae) whales revealed by stable isotope mixing models*. *Marine Mammal Science*, .

Ryan, C., McHugh, B., Trueman, C.N., Sabin, R., Deaville, R., Harrod, C., Berrow, S.D. and O'Connor, I., 2013. Stable isotope analysis of baleen reveals resource partitioning among sympatric rorquals and population structure in fin whales. *Marine Ecology Progress Series*, 479, pp.251–261.

Sanabria-Fernandez, J.A., Lazzari, N., Riera, R. and Becerro, M.A., 2018. Building up marine biodiversity loss: Artificial substrates hold lower number and abundance of low occupancy benthic and sessile species. *Marine Environmental Research*, [online] 140, pp.190–199. Available at: <https://www.sciencedirect.com/science/article/pii/S0141113618300606> [Accessed 9 Jun. 2019].

Scheidat, M., Tougaard, J., Brasseur, S., Carstensen, J., van Polanen Petel, T., Teilmann, J. and Reijnders, P., 2011. Harbour porpoises (Phocoena phocoena) and wind farms: a case study in the Dutch North Sea. *Environmental Research Letters*, [online] 6(2), p.025102. Available at: <http://stacks.iop.org/1748-9326/6/i=2/a=025102?key=crossref.32cfae25976538501b44e42e503686ab>.

Schuster, E., Bulling, L. and Köppel, J., 2015. Consolidating the State of Knowledge: A Synoptical Review of Wind Energy's Wildlife Effects. *Environmental Management*, [online] 56(2), pp.300–331. Available at: <http://link.springer.com/10.1007/s00267-015-0501-5> [Accessed 15 Mar. 2019].

Sclavounos, P.D., Lee, S. and DiPietro, J., 2010. Floating offshore wind turbines: tension leg platform and taught leg buoy concepts supporting 3–5 mw wind turbines. *European Wind Energy Conference (EWEC)*, [online] pp.1–7. Available at: <http://web.mit.edu/flowlab/pdf/EWEC2010.pdf>.

Scott, K., Harsanyi, P. and Lyndon, A.R., 2018. Understanding the effects of electromagnetic field emissions from Marine Renewable Energy Devices (MREDs) on the commercially important edible crab, Cancer pagurus (L.). *Marine Pollution Bulletin*, [online] 131(May), pp.580–588. Available at: <https://doi.org/10.1016/j.marpolbul.2018.04.062>.

Seuront, L. and Cribb, N., 2011. Fractal analysis reveals pernicious stress levels related to boat presence and type in the Indo–Pacific bottlenose dolphin, Tursiops aduncus. *Physica A: Statistical Mechanics and its Applications*, [online] 390(12), pp.2333–2339. Available at: <https://www.sciencedirect.com/science/article/pii/S0378437111001233> [Accessed 21 Feb. 2019].

Shafiei Sabet, S., Wesdorp, K., Campbell, J., Snelderwaard, P. and Slabbekoorn, H., 2016. Behavioural responses to sound exposure in captivity by two fish species with different hearing ability. *Animal Behaviour*, [online] 116, pp.1–11. Available at: <https://www.sciencedirect.com/science/article/pii/S000334721630001X> [Accessed 3 Jun. 2019].

Sierra-Flores, R., Atack, T., Migaud, H. and Davie, A., 2015. Stress response to anthropogenic noise in Atlantic cod Gadus morhua L. *Aquacultural Engineering*, [online] 67, pp.67–76. Available at: <https://www.sciencedirect.com/science/article/pii/S0144860915000503> [Accessed 27 May 2019].

Sigray, P. and Andersson, M.H., 2011. Particle motion measured at an operational wind turbine in relation to hearing sensitivity in fish. *The Journal of the Acoustical Society of America*, [online] 130(1), pp.200–207. Available at: <http://asa.scitation.org/doi/10.1121/1.3596464>.

Simmonds, M. and Dolman, S., 2008. All at sea: Renewable Energy Production in the Context of Marine Nature Conservation. *Offshore wind farms and marine mammals impacts and methodologies for assesing impacts - Proceedings of the Ascobans/ECS Workshop*, (49), pp.6–11.

Simmonds, M.P. and Brown, V.C., 2010. Is there a conflict between cetacean conservation and marine renewable-energy developments? *Wildlife Research*, 37(8), p.688.

Simpson, S.D., Purser, J. and Radford, A.N., 2015. Anthropogenic noise compromises antipredator behaviour in European eels. *Global Change Biology*, [online] 21(2), pp.586–593. Available at: <http://doi.wiley.com/10.1111/gcb.12685> [Accessed 5 Jun. 2019].

Slabbekoorn, H., Bouton, N., van Opzeeland, I., Coers, A., ten Cate, C. and Popper, A.N., 2010. A noisy spring: The impact of globally rising underwater sound levels on fish. *Trends in Ecology & Evolution*, [online] 25(7), pp.419–427. Available at: <https://linkinghub.elsevier.com/retrieve/pii/S0169534710000832> [Accessed 2 Apr. 2019].

Slabbekoorn, H., Dooling, R.J., Popper, A.N. and Fay, R.R. eds., 2018. *Effects of Anthropogenic Noise* *on Animals*. Springer Handbook of Auditory Research. [online] New York, NY: Springer New York. Available at: <http://link.springer.com/10.1007/978-1-4939-8574-6> [Accessed 28 May 2019].

Smith, M.E., Schuck, J.B., Gilley, R.R. and Rogers, B.D., 2011. Structural and functional effects of acoustic exposure in goldfish: evidence for tonotopy in the teleost saccule. *BMC Neuroscience*, [online] 12(1), p.19. Available at: <https://bmcneurosci.biomedcentral.com/articles/10.1186/1471-2202-12-19> [Accessed 28 May 2019].

Solan, M., Hauton, C., Godbold, J.A., Wood, C.L., Leighton, T.G. and White, P., 2016. Anthropogenic sources of underwater sound can modify how sediment-dwelling invertebrates mediate ecosystem properties. *Scientific Reports*, [online] 6(1), p.20540. Available at: <http://www.nature.com/articles/srep20540> [Accessed 5 Jun. 2019].

Southall, B.L., Bowles, A.E., Ellison, W.T., Finneran, J.J., Gentry, R.L., Greene Jr., C.R., Kastak, D., Ketten, D.R., Miller, J.H., Nachtigall, P.E., Richardson, J.W., Thomas, J.A. and Tyack, P.L., 2007. Marine mammal noise criteria: initial scientific recommmendations. *Aquatic Mammals*, [online] 33(4), pp.411–522. Available at: <http://csi.whoi.edu/content/marine-mammal-noise-exposure-criteria-initial-scientific-recommendations-0>.

Southall, B.L., Finneran, J.J., Reichmuth, C., Nachtigall, P.E., Ketten, D.R., Bowles, A.E., Ellison, W.T., Nowacek, D.P. and Tyack, P.L., 2019. Marine Mammal Noise Exposure Criteria: Updated Scientific Recommendations for Residual Hearing Effects. *Aquatic Mammals*, 45(2), pp.125–232.

Spiga, I., 2016. Acoustic Response to Playback of Pile-Driving Sounds by Snapping Shrimp. [online] Springer, New York, NY, pp.1081–1088. Available at: <http://link.springer.com/10.1007/978-1-4939-2981-8_134> [Accessed 22 Jun. 2019].

Spiga, I., Aldred, N. and Caldwell, G.S., 2017. Anthropogenic noise compromises the anti-predator behaviour of the European seabass, Dicentrarchus labrax (L.). *Marine Pollution Bulletin*, [online] 122(1–2), pp.297–305. Available at: <https://www.sciencedirect.com/science/article/pii/S0025326X17305647> [Accessed 21 Feb. 2019].

Spiga, I., Caldwell, G.S. and Bruintjes, R., 2016. Influence of Pile Driving on the Clearance Rate of the Blue Mussel, Mytilus edulis (L.). In: *Proceedings of Meetings on Acoustics*. [online] Acoustical Society of America, p.040005. Available at: <http://asa.scitation.org/doi/abs/10.1121/2.0000277> [Accessed 28 May 2019].

Stelfox, M., Hudgins, J. and Sweet, M., 2016. A review of ghost gear entanglement amongst marine mammals, reptiles and elasmobranchs. *Marine Pollution Bulletin*, [online] 111(1–2), pp.6–17. Available at: <https://linkinghub.elsevier.com/retrieve/pii/S0025326X16304386> [Accessed 10 Sep. 2018].

Stenberg, C., Støttrup, J.G., Van Deurs, M., Berg, C.W., Dinesen, G.E., Mosegaard, H., Grome, T.M. and Leonhard, S.B., 2015. Long-term effects of an offshore wind farm in the North Sea on fish communities. *Marine Ecology Progress Series*, 528, pp.257–265.

Sutton, G., Jessopp, M., Clorennec, D. and Folegot, T., 2013. *Mapping the Spatio-temporal Distribution of Underwater Noise in Irish Waters. STRIVE Report Series No . 121. EPA STRIVE Programme 2007-2013*. Wexford, Ireland: EPA STRIVE Programme 2007-2013.

Taormina, B., Bald, J., Want, A., Thouzeau, G., Lejart, M., Desroy, N. and Carlier, A., 2018. A review of potential impacts of submarine power cables on the marine environment: Knowledge gaps, recommendations and future directions. *Renewable and Sustainable Energy Reviews*, [online] 96(August), pp.380–391. Available at: <https://doi.org/10.1016/j.rser.2018.07.026>.

Teilmann, J. and Carstensen, J., 2012. Negative long term effects on harbour porpoises from a large scale offshore wind farm in the Baltic - Evidence of slow recovery. *Environmental Research Letters*, 7(4).

Tennessen, J.B., Parks, S.E. and Langkilde, T.L., 2016. Anthropogenic Noise and Physiological Stress in Wildlife. In: A. Popper and A. Hawkins, eds., *The Effects of Noise on Aquatic Life II*. [online] Springer, New York, NY, pp.1145–1148. Available at: <http://link.springer.com/10.1007/978-1-4939-2981-8_142> [Accessed 22 Jun. 2019].

Thompson, P.M., Lusseau, D., Barton, T., Simmons, D., Rusin, J. and Bailey, H., 2010. Assessing the responses of coastal cetaceans to the construction of offshore wind turbines. *Marine Pollution Bulletin*, [online] 60(8), pp.1200–1208. Available at: <http://dx.doi.org/10.1016/j.marpolbul.2010.03.030> [Accessed 21 Feb. 2019].

Thomsen, F., Luedemann, K., Piper, W., Judd, A. and Kafemann, R., 2008. Potential effects of offshore wind farm noise on fish. *Bioacoustics*, [online] 17(1–3), pp.221–223. Available at: <http://www.tandfonline.com/doi/abs/10.1080/09524622.2008.9753825> [Accessed 14 Feb. 2019].

Tidau, S. and Briffa, M., 2016. Review on behavioral impacts of aquatic noise on crustaceans. In: *Proceedings of Meetings on Acoustics*. [online] Acoustical Society of America, p.15. Available at: <http://asa.scitation.org/doi/abs/10.1121/2.0000302> [Accessed 4 Jun. 2019].

Tougaard, J. and Beedholm, K., 2019. Practical implementation of auditory time and frequency weighting in marine bioacoustics. *Applied Acoustics*, [online] 145, pp.137–143. Available at: <https://linkinghub.elsevier.com/retrieve/pii/S0003682X18303967>.

Tougaard, J., Carstensen, J., Teilmann, J., Skov, H. and Rasmussen, P., 2009. Pile driving zone of responsiveness extends beyond 20 km for harbor porpoises (*Phocoena phocoena* (L.)). *The Journal of the Acoustical Society of America*, [online] 126(1), pp.11–14. Available at: <http://asa.scitation.org/doi/10.1121/1.3132523> [Accessed 1 Mar. 2019].

Tougaard, J. and Mikaelsen, M.A., 2018. *Effects of larger turbines for the offshore wind farm at Krieger's Flak, Sweden. Assessment of impact on marine mammals. Aarhus University, DCE – Danish Centre for Environment and Energy, 112 pp. Scientific Report No. 286*. [online] Available at: <http://dce2.au.dk/pub/SR286.pdf>.

Tougaard, J., Wright, A.J. and Madsen, P.T., 2015. Cetacean noise criteria revisited in the light of proposed exposure limits for harbour porpoises. *Marine Pollution Bulletin*, [online] 90(1–2), pp.196–208. Available at: <https://www.sciencedirect.com/science/article/pii/S0025326X14007358> [Accessed 21 Feb. 2019].

De Troch, M., Reubens, J.T., Heirman, E., Degraer, S. and Vincx, M., 2013. Energy profiling of demersal fish: A case-study in wind farm artificial reefs. *Marine Environmental Research*, 92, pp.224–233.

Tyack, P., 2009. Acoustic playback experiments to study behavioral responses of free-ranging marine animals to anthropogenic sound. *Marine Ecology Progress Series*, [online] 395, pp.187–200. Available at: <http://www.int-res.com/abstracts/meps/v395/p187-200/> [Accessed 22 Feb. 2019].

Vaissière, A.C., Levrel, H., Pioch, S. and Carlier, A., 2014. Biodiversity offsets for offshore wind farm projects: The current situation in Europe. *Marine Policy*, [online] 48, pp.172–183. Available at: <http://dx.doi.org/10.1016/j.marpol.2014.03.023>.

Vallejo, G.C., Grellier, K., Nelson, E.J., McGregor, R.M., Canning, S.J., Caryl, F.M. and McLean, N., 2017. Responses of two marine top predators to an offshore wind farm. *Ecology and Evolution*, [online] (July), pp.1–11. Available at: <http://doi.wiley.com/10.1002/ece3.3389>.

Vandendriessche, S., Derweduwen, J. and Hostens, K., 2015. Equivocal effects of offshore wind farms in Belgium on soft substrate epibenthos and fish assemblages. *Hydrobiologia*, [online] 756(1), pp.19–35. Available at: <http://link.springer.com/10.1007/s10750-014-1997-z> [Accessed 3 Apr. 2019].

Vaughan, D., 2017. Fishing effort displacement and the consequences of implementing Marine Protected Area management – An English perspective. *Marine Policy*, [online] 84, pp.228–234. Available at: <https://www.sciencedirect.com/science/article/pii/S0308597X16307588> [Accessed 13 Jun. 2019].

Verfuss, U.K., Sparling, C.E., Arnot, C., Judd, A. and Coyle, M., 2016. Review of Offshore Wind Farm Impact Monitoring and Mitigation with Regard to Marine Mammals. In: A.N. Popper and A.D. Hawkins, eds., *The Effects of Noise on Aquatic Life II*. [online] New York: Spinger-Verlag, pp.1175–1182. Available at: <http://link.springer.com/10.1007/978-1-4939-2981-8_147> [Accessed 20 Feb. 2019].

Wahlberg, M. and Westerberg, H., 2005. Hearing in Fish and Reaction to Offshore Wind. *Marine Ecology Progress Series*, 288(2001), pp.295–309.

Wall, D., Murray, C., O'Brien, J., Kavanagh, L., Ryan, C., Glanville, B., Williams, D., Enlander, I., O'Connor, I., Mcgrath, D., Whooley, P., Berrow, S.D. and Wilson, C., 2013. *Atlas of the Distribution and Relative Abundance of Marine Mammals in Irish Offshore Waters : 2005 – 2011*. *The Irish Whale and Dolphin Group*.

Westerberg, H. and Legenfelt, I., 2008. Sub-sea power cables and the migration behaviour of the European eel. *Fisheries Management and Ecology*, [online] 15(5–6), pp.369–375. Available at: <http://doi.wiley.com/10.1111/j.1365-2400.2008.00630.x> [Accessed 11 Jun. 2019].

White, M., Mohn, C. and Orren, M.J., 1998. Nutrient distributions across the Porcupine Bank. *ICES Journal of Marine Science*, [online] 55(6), pp.1082–1094. Available at: <//000077980200007>.

Whitfield, A.K.K. and Becker, A., 2014. Impacts of recreational motorboats on fishes: A review. *Marine Pollution Bulletin*, [online] 83(1), pp.24–31. Available at: <http://dx.doi.org/10.1016/j.marpolbul.2014.03.055> [Accessed 14 May 2019].

Wilber, D.H., Carey, D.A. and Griffin, M., 2018. Flatfish habitat use near North America's first offshore wind farm. *Journal of Sea Research*, 139(November 2017), pp.24–32.

Wilhelmsson, D. and Langhamer, O., 2014. The Influence of Fisheries Exclusion and Addition of Hard Substrata on Fish and Crustaceans. In: M.A. Shields and M.R. Payne, eds., *Marine Renewable Energy Technology and Environmental Interactions*. [online] Dordrecht: Springer, Dordrecht, pp.49–60. Available at: <http://link.springer.com/10.1007/978-94-017-8002-5_5> [Accessed 7 May 2019].

Wilhelmsson, D. and Malm, T., 2008. Fouling assemblages on offshore wind power plants and adjacent substrata. *Estuarine, Coastal and Shelf Science*, [online] 79(3), pp.459–466. Available at: <https://www.sciencedirect.com/science/article/pii/S0272771408001911> [Accessed 3 Apr. 2019].

Wilhelmsson, D., Malm, T. and Ohman, M., 2006. The influence of offshore windpower on demersal fish. *ICES Journal of Marine Science*, [online] 63(5), pp.775–784. Available at: <https://academic.oup.com/icesjms/article-lookup/doi/10.1016/j.icesjms.2006.02.001> [Accessed 3 Apr. 2019].

Williams, R., Veirs, S., Veirs, V., Ashe, E. and Mastick, N., 2019. Approaches to reduce noise from ships operating in important killer whale habitats. *Marine Pollution Bulletin*, [online] 139(May 2018), pp.459–469. Available at: <https://doi.org/10.1016/j.marpolbul.2018.05.015>.

Williams, R., Wright, A.J., Ashe, E., Blight, L.K., Bruintjes, R., Canessa, R., Clark, C.W., Cullis-Suzuki, S., Dakin, D.T., Erbe, C., Hammond, P.S., Merchant, N.D., O'Hara, P.D., Purser, J., Radford, A.N., Simpson, S.D., Thomas, L. and Wale, M.A., 2015. Impacts of anthropogenic noise on marine life:

Publication patterns, new discoveries, and future directions in research and management. *Ocean & Coastal Management*, [online] 115, pp.17–24. Available at: <http://dx.doi.org/10.1016/j.ocecoaman.2015.05.021>.

WindEurope, 2018. *World's first offshore wind farm without subsidies to be built in the Netherlands | WindEurope*. [online] Available at: <https://windeurope.org/newsroom/press-releases/worlds-first-offshore-wind-farm-without-subsidies-to-be-built-in-the-netherlands/> [Accessed 24 Jun. 2019].

WindEurope, 2019. *Offshore wind in Europe. Key trends and statistics 2018*. [online] Available at: <https://windeurope.org/wp-content/uploads/files/about-wind/statistics/WindEurope-Annual-Offshore-Statistics-2018.pdf>.

Wingfield, J.E., O'Brien, M., Lyubchich, V., Roberts, J.J., Halpin, P.N., Rice, A.N. and Bailey, H., 2017. Year-round spatiotemporal distribution of harbour porpoises within and around the Maryland wind energy area. *PLoS ONE*, 12(5), pp.1–18.

Wisniewska, D.M., Johnson, M., Teilmann, J., Siebert, U., Galatius, A., Dietz, R. and Madsen, P.T., 2018. High rates of vessel noise disrupt foraging in wild harbour porpoises (Phocoena phocoena). *Proceedings of the Royal Society B: Biological Sciences*, [online] 285(1872), p.20172314. Available at: <http://rspb.royalsocietypublishing.org/lookup/doi/10.1098/rspb.2017.2314>.

Wright, A.J., Deak, T. and Parsons, E.C.M., 2011. Size matters: Management of stress responses and chronic stress in beaked whales and other marine mammals may require larger exclusion zones. *Marine Pollution Bulletin*, [online] 63(1–4), pp.5–9. Available at: <http://linkinghub.elsevier.com/retrieve/pii/S0025326X09005013>.

Wu, X., Hu, Y., Li, Y., Yang, J., Duan, L., Wang, T., Adcock, T., Jiang, Z., Gao, Z., Lin, Z., Borthwick, A. and Liao, S., 2019. Foundations of offshore wind turbines: A review. *Renewable and Sustainable Energy Reviews*, [online] 104, pp.379–393. Available at: <https://www.sciencedirect.com/science/article/pii/S1364032119300127> [Accessed 22 Jun. 2019].

Wyman, M.T., Peter Klimley, A., Battleson, R.D., Agosta, T. V., Chapman, E.D., Haverkamp, P.J., Pagel, M.D. and Kavet, R., 2018. Behavioral responses by migrating juvenile salmonids to a subsea high-voltage DC power cable. *Marine Biology*, [online] 165(8), p.134. Available at: <http://link.springer.com/10.1007/s00227-018-3385-0> [Accessed 7 May 2019].