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Marine noise pollution and its impacts on fish: priorities, models, and methods for mitigation

by

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Thesis Abstract

The effects of noise on aquatic life is a topic of growing international concern. Underwater noise can impact both the physiology and behaviour of fish species on a wide-ranging scale, from minor changes and adaptations to major injury and death. Future mitigation of anthropogenic noise in the ocean is dependent on greater awareness of the effects of noise, the amount of risk, and degree of harm, likely to affect fish populations. Currently, there is a lack of incentive for mitigation measures to be put in place. Knowledge and evidence of the impacts of anthropogenic noise on fish is rapidly increasing (Figure 1.2) but with over 32,000 species of fish of differing conservation and commercial importance, it is extremely difficult to decide where to focus research for maximum benefit (Hawkins *et al.*, 2015). Predictions and assumptions about potential impacts lack accuracy as variations in experimental equipment and techniques, lack of agreed standards, different algorithms for analysis, ambiguous and interchangeable terminology, and different quantities, units and metrics, all lead to incongruities (ISVR Consulting, 2004; Barlow *et al.*, 2014; Rogers *et al.*, 2016). Often it is not possible to compare studies or make generalisations (OSPAR, 2009; Wilcock *et al.*, 2014). Here the aim is to aid the mitigation process by directing research priorities toward the most vulnerable fish species, and developing models and tools that allow for informed and cost-effective mitigation methods in a bid to reduce the effects of anthropogenic noise from marine traffic.

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Declaration of Authorship

I, Sarah Tegan Victoria Neenan declare that this thesis and the work presented in it is my own and has been generated by me as the result of my own original research.

Marine noise pollution and its impacts on fish: priorities, models, and methods for mitigation

I confirm that:

1. This work was done wholly or mainly while in candidature for a research degree at this University;
2. Where any part of this thesis has previously been submitted for a degree or any other qualification at this University or any other institution, this has been clearly stated;
3. Where I have consulted the published work of others, this is always clearly attributed;
4. Where I have quoted from the work of others, the source is always given. With the exception of such quotations, this thesis is entirely my own work;
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6. Where the thesis is based on work done by myself jointly with others, I have made clear exactly what was done by others and what I have contributed myself;
7. Parts of this work have been published as:

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Signed: ....

Date: ....

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Thesis overview

This research project was sponsored by FugroEMU Limited (Trafalgar Wharf, Portsmouth) as there is an industry need for an accurate method of predicting the possible harmful effects of underwater noise on fish populations. Clients undertaking new projects within the marine environment need to comply with legal statutes, but choosing a suitable and beneficial mitigation method to implement is difficult. Research into mitigation options is often long-term and can impact on everyday noise-producing activities. New innovative tools are required that can provide a reliable and cost-effective method of ensuring that any damage to underwater ecosystems is minimised.

The models created here are flexible, adaptable, replicable (to combat data reporting errors) and can incorporate any chosen fish species, within any location. Such adaptability means they can be widely applied within the sphere of mitigation management. These models can easily be made available as a user-friendly online tool that provides for widespread dissemination of the data, and ensures information for decision makers is readily available. It is hoped that the methods designed and presented within this project will aid in the prioritisation of fish research, provide a novel method of predicting successful mitigation strategies, and help inform marine developers and policy decision-makers of the potential impacts of vessel noise on marine fish species.

This project delivers novelty in a number of areas. The Prioritisation Index (and potential for accompanying online tool) offers a new framework for prioritising research and mitigation, demonstrated here in terms of noise pollution. The Index is flexible and can easily be applied during Impact Assessments to identify and allow for mitigation of the most vulnerable species. The innovative modelling approach to create an ocean vessel noise exposure map (and potential software tool) combines and builds upon previous research to further understand vessel associated noise emissions. It also provides a means of predicting future vessel noise exposure trends, and the potential impacts those trends may have on marine fish species.

Glossary

Auditory Brainstem Response	ABR	A non-invasive method of collecting audiogram data, by placing small electrodes on the subject's head to measure the auditory evoked potentials in response to a noise stimulus
Automatic Identification System	AIS	An automatic tracking system used on vessels that broadcasts ship information and position data via VHF radio
AWK		A programming language used for text processing and data extraction
Blue biotechnology		A term used to describe the combination of marine organisms with technology to produce medicines, energy sources and other products.
Conspecific		A member of the same species
Cumulative Sound Exposure Level	SELcum	The total sound exposure level determined for an extended period or number of events.
Decibel level at 1 m	dB re 1 m	Units for Transmission Loss
Environmental Impact Assessment	EIA	A policy/management tool which identifies and evaluates the potential impacts (effects) of a proposed development and makes sure that mitigation methods are properly understood by the relevant authority before development decisions are made
Geographic Information System	GIS	A computer system designed for visualisation, manipulation and analysation of geographic data
Good Ecological Status	GEcS	GEcS ensures that uses of the water environment do not alter the structure and functioning of aquatic plant and animal communities
Good Environmental Status	GES	GES ensure that uses of marine resources are conducted at a sustainable level ensuring their continuity for future generations
Hertz	Hz	Unit of frequency
Inverse Distance Weighted	IDW	A method for predicting a value for any unmeasured location using measured values of surrounding locations
Marine Mobile Service Identity	MMSI	A vessel's unique identifier
Masking		When the detection of one sound is impaired by the presence of another
MicroPascals	μPa	One millionth of a Pascal (the unit for pressure defined as one newton per square metre)
Noise		Any anthropogenic sound source present in the ocean, either intentionally or unintentionally produced

Particle Velocity Level	PVL	The velocity of a particle in a medium as it transmits a wave, measured using a logarithmic decibel scale
Permanent Threshold Shift	PTS	A permanent shift in the auditory threshold which is not reversible
Propagation Loss	PL	The drop in sound energy level as it propagates from one point in the ocean to another (the ratio of the acoustic intensity at the particular point, to the level at some reference position). Synonym for Transmission Loss
Received Level	RL	The sound level at the listeners' (or receivers') actual position, which is usually considerably more distant that the reference source level of 1 m. RL can be calculated by subtracting the PL from the SL
Revolutions per minute	RPM	A unit of frequency used to measure rotational speed
Root Mean Square	RMS	The log transformed square root of the average square pressure of a signal over a specific time interval (dB re µPa rms)
Signal		An informative sound, for example aiding in communication or orientation
Single Pulse Sound Exposure Level	SELss	Sound Exposure Level for a single event. Used for recording pile driving noise emissions.
Sound		General term used to describe the effect a vibrating object has on the surrounding environment
Sound Exposure Level	SEL	(1 $\mu Pa^2 s$)
Sound Pressure Level	SPL	The difference between the instantaneous total pressure and the pressure that would exist in the absence of sound (dB re µPa). This is the most suitable metric for continuous sounds such as that from shipping
Soundscape		An acoustic environment consisting of natural sounds (including biotic (e.g. marine species) and abiotic (e.g. weather)) and anthropogenic sounds
Source Level	SL	Source level represents the noise level at a distance of one meter from the source, referenced to one microPascal. commonly seen expressed as dB re 1 µPa at 1 m (understood as dB re 1 µPa referred to 1 m)
Temporary Threshold Shift	TTS	A temporary shift in the auditory threshold which is reversible
Transmission Loss	TL	Synonym for Propagation Loss
Vessel Monitoring System	VMS	A satellite-based system for monitoring, control and surveillance of fishing vessels

Abbreviations

ACCOBAMS	Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea and contiguous Atlantic area
CIA	Cumulative Impact Assessment
dph	Days post hatch
EIA	Environmental Impact Assessment
Eq	Equation
FAO	Food and Agriculture Organization
ICES	International Council for the Exploration of the Sea
IUCN	International Union for Conservation of Nature
MCZ	Marine Conservation Zone
MMO	Marine Management Organisation
MPA	Marine Protected Area
MSFD	Marine Strategy Framework Directive
n.d.	No date
SONIC	Suppression Of underwater Noise Induced by Cavitation
SSLM	Ship Source Level Model
WFD	Water Framework Directive

Thesis outline

In Chapter One a comprehensive literature review is undertaken. The effects of anthropogenic noise on marine fish are discussed, with particular emphasis on the impacts from shipping noise emissions. Concerns and knowledge gaps within marine noise pollution research on fish are uncovered, and areas identified where new and innovative methods would aid the mitigation process. The importance of prioritising research was clear, and yet no prioritisation frameworks for identifying fish species vulnerable to noise pollution were found. Such models provide a method of identifying the biological information required to reduce the effects of noise pollution and help design successful focused mitigation measures. Modelling and mapping methods are identified as helpful tools for use in research to aid mitigation. The aims and objectives of the thesis are stated in this chapter.

In Chapter Two a method of establishing priorities for mitigation research, using value and susceptibility indicators, is discussed. The framework presented provides a new semi-quantitative approach of prioritising and directing research attention toward those fish species most affected by noise pollution in terms of their populations, the ecosystem, and value to human society. This will assist researchers and policy-makers to make informed decisions with regard to prioritising future research and mitigation.

In Chapter Three an innovative AIS-based tool to model source level noise exposure from shipping is constructed. It is used to illustrate historical cumulative exposure levels for UK waters. Much of the research on noise impacts to date has focused on specific species rather than specific habitats but some habitats suffer more noise exposure than others. The model created delivers a method of researching source-based mitigation and predictions of noise trends that identifies areas exposed to high levels of shipping noise, and provides prioritisation by geographic areas. This chapter also introduces the concept of changing the sound source to prevent the initial impacts, and thus damage to marine life, from occurring. The model developed here, maps shipping movements, estimates the likely source level of each vessel, and provides a method of predicting potential source levels under various conditions.

In Chapter Four the predictive modelling ability of the AIS-based model produced in Chapter 3 is investigated. Analysing the efficiency of mitigation measures using models is beneficial as strategies cost money and effort to implement. Large-scale mitigation efforts require strong evidence to support them. Five proposed future mitigation scenarios are discussed. The chapter compares the historical trend reported in Chapter 3 to the predicted future trends for the five different mitigation scenarios, to see which, if any, are feasible and worthwhile. Two forms of mitigation, source level reduction and geographic mitigation are examined. The models are used to demonstrate a cost-effective method of obtaining evidence to guide mitigation decisions. The potential application of AIS models as a method for mitigation is further reviewed.

In Chapter Five the AIS-based model is used to highlight the impacts of vessel noise on species' populations and identifies the potential for migration routes to be affected by noise barriers (using European eel and Atlantic salmon as case studies). A combination of biological information, modelling and mapping methods are used to investigate and predict the noise impacts that could negatively impact species' migrations. This demonstrates the broader use of AIS models in mitigation, as it encompasses priority modelling, source-level modelling, biological modelling and predictive modelling in one single application. This allows predictions and recommendations to be made to mitigate the impacts that migrating species may face from vessel noise pollution.

In Chapter Six priorities, models, and methods for mitigation relating to noise pollution impacts on fish in the marine environment are discussed.

1. Chapter One: Literature Review

Many fish species endure a perilous existence from a variety of natural and anthropogenic (man-made) threats. To help reduce the decline of marine species, anthropogenic threats need to be managed and controlled as efficiently and effectively as possible within the constraints of available resources. Conservation of fish species is important for the sustainability of species for future generations, and to prevent the collapse of fisheries. Overexploitation, habitat degradation, chemical pollution, and invasive species (Jackson *et al.*, 2001; van der Oost *et al.*, 2003; Munday, 2004) all threaten populations. Austin (1998) wrote that pollutants can have a devastating effect on marine life, as the impacts can result in rapid, large scale mortality, and/or can occur for long periods after initial exposure. Pollutants as a whole have received a lot of attention, but the effects of noise pollution in particular have been largely neglected until recent years. Noise pollution in the oceans can affect individual organisms, populations, food chains or whole ecosystems, and continues to pollute the marine environment every day. The vast majority of fish have the potential to be negatively affected by noise pollution (Cox *et al.*, 2016). It is, therefore, important to know the consequences of noise pollution in order to conserve species successfully (Slabbekoorn *et al.*, 2010). Research on fish thus far has shown noise exposure can result in: reduced fitness; increased predation rate; reduced foraging efficiency; increased occurrence of confused and displaced individuals and schools; and in the case of vocal fishes, disrupted acoustical communication (McCauley *et al.*, 2003; Sarà *et al.*, 2007; Purser and Radford, 2011; Voellmy *et al.*, 2014b; Luczkovich *et al.*, 2016). Populations impacted by noise show a decline in survival rates, altered dispersion behaviour, and lower reproductive success (Meier and Horseman, 1977; Sarà *et al.*, 2007; Simpson *et al.*, 2016b; Herbert-read *et al.*, 2017; Nedelec *et al.*, 2017b). Any marked reduction in commercial and popular tourism species could have serious repercussions for the economy and welfare of maritime nations. To help determine the current effects of anthropogenic noise on marine fish, in particular those directly attributable to the activities of marine shipping, an in-depth literature review was conducted. The relevant facts and findings have been extracted and reported in this chapter.

1.1 The definition of noise

Before the topic of marine noise pollution impacts can be discussed, it is important to understand the actual definition of 'noise' in the marine environment. There is a lack of consistency within the literature associated with marine noise pollution research. In the context of the responses of fish to acoustic signals, in a laboratory there is little question about what to classify as a signal and what constitutes noise (Braun, 2015). The signal is the specific stimulus (or stimuli) presented to the subject in an attempt to elicit some response, either behaviourally conditioned or physiologically monitored, and noise refers to any remaining ambient energy. However, when discussing noise in the natural environment, the definitions become more contentious.

Previous research on noise pollution in the marine environment has generated definitions that have become widely accepted. 'Sound' is a general term used to describe disturbance in pressure that propagates through a medium, and is also used as a broad description of acoustic energy (Southall, 2005). 'Sound' can then be subdivided into 'signals' which contain biologically significant information (such as the location of a mate or predator), and 'noise' which is used to describe sound from a diffuse array of sources that does not convey biologically significant information (Southall, 2005). However, in the natural environment, multiple sensory information streams are present simultaneously and so a sound can be both an informative signal, and a distracting 'noise' that decreases efficiency or performance of sensory tasks, at the same time. Human-generated sounds, such as those emitted from vessels, might be viewed as: an ambient distractor limiting acoustic cues; a chronic stressor impairing neural and behavioural function; a legitimate warning signal that a potentially harmful activity is about to occur; or an informative signal for navigation that animals might learn to recognise and use (Braun, 2015). To add to the contention, some authors argue that the term 'soundscape' should replace 'noise' as the designator for the ambient combination of all sound sources surrounding a listener (Pijanowski *et al.*, 2011). The Marine Strategy Framework Directive defined 'noise' as any sound that has the potential to negatively impact marine life (Graaf *et al.*, 2012), but others have referred to noise as only accidental sound, as opposed to intentional sounds which are not considered as noise (Universitat Politècnica de Catalunya, 2011). Defining noise in terms of accidental sounds alone would exclude such activities as seismic surveys, sonar, or fishing using echosounders as noise pollution sources.

As there is not one universally accepted definition of sound and noise, in this thesis all anthropogenic sound sources will be considered as generating 'noise', regardless of whether the noise was intentionally produced (such as seismic exploration, sonars, acoustic deterrents) or an unintended by-product of human activity (such as maritime shipping and construction), and that 'noise' has the potential to negatively impact marine life.

1.2 Marine noise pollution and shipping

Human activities generate noise in the marine environment. These noises are often louder, more frequent and more common than natural acoustic stimuli (Patricelli and Blickley, 2006; Popper and Hastings, 2009a). Noise production may have an explicit purpose, such as locating submerged objects or fish shoals, or measuring environmental features, or it may be an unwelcome by-product of industrial activities such as the construction of infrastructure or the movement of vessels. Anthropogenic noise can potentially directly affect any animal it comes into contact with that is capable of hearing it (Slabbekoorn *et al.*, 2010). The majority of marine fishes evaluated for hearing capabilities so far are most sensitive to sounds occurring in a low frequency (20–1000 Hz) range (*see* Ladich and Fay's (2013) review), indicating considerable spectral overlap between sensitivity of fish hearing and the frequency of peak noise levels produced by shipping activities. Man-made noise can be especially detrimental to fish if it occurs at frequencies within their hearing thresholds (Scholik and Yan, 2002).

In 2003 the National Research Council grouped anthropogenic noise sources into six categories comprised of explosions, industrial activity, shipping, seismic surveying, sonars, and miscellaneous (National Research Council, 2003). Underwater noise pollution sources of greatest concern being explosions, shipping, sonars, air guns, dredgers, ocean science studies, hydroelectric dams, fishing equipment with acoustic deterrent devices, and noises associated with oil and gas production (Richardson *et al.*, 1995; Richardson and Würsig, 1997).

The most widespread, but not necessarily the highest impact, source of marine anthropogenic underwater noise pollution is from shipping (Firestone and Jarvis, 2007; Jensen *et al.*, 2009). Vessels contribute to noise in the underwater environment through hydraulic flow over the hull, turbulence around various external ship elements, propulsion, propeller singing, propeller cavitation (which produces significant low-frequency noise because of bubble creation and collapse from the rotation of the propeller blade), and the use of other on-board machinery (e.g. rotating machinery of the vessels' engines, generators, and on board navigational sonar) (Richardson *et al.*, 1995; Erbe, 2002; Southall, 2005; Hildebrand, 2009; Harris, 2017). Small craft with high speed engines and propellers generally produce higher frequency noise (Erbe, 2002, 2013), whereas large vessels (e.g. cruise ships, container ships) generate lower frequency noise (<1,000 Hz) because of their size and their large, lower RPM (revolutions per minute) engines and propellers (Arveson and Vendittis, 2000; McKenna *et al.*, 2012).

The broadband and tonal components produced by propeller cavitation account for 80-85 per cent of ship-radiated noise power (Ross, 1976). Peak spectral densities (of a monopole source at 1 m) for individual ships have been seen to range from 195 dB re 1 $\mu Pa^2.m^2Hz^{-1}$ for fast moving super tankers, to 140 dB re 1 $\mu Pa^2.m^2Hz^{-1}$ for small fishing vessels (Wagstaff, 1973). The majority of ships produce a spectrum of noise that peaks in the low frequency region (5 to 500 Hz) (Hildebrand, 2004). As with other noise sources, the exact noise emissions depend on various factors such as vessel type, size, age, and operational mode, as well as the propeller depth, speed, load and pitch angle (OSPAR Commission, 2009a). All are important when considering long-range propagation (Amoser *et al.*, 2004; Southall, 2005).

The external environment can also influence the noise levels of a vessel. Marine traffic noise produced at high latitudes is particularly efficient at propagating over large distances because in these regions the oceanic sound channel (the zone of most efficient noise propagation) reaches the ocean surface (Leighton, 1998). High frequency noise propagates further in shallow water than low frequency noise (Amoser and Ladich, 2005). Shipping noise can also make use of 'down-slope conversion' as noise propagating down the continental slope can readily enter the deep sound channel (McDonald *et al.*, 2006). The shipping lanes that traverse the continental slope are ideal sites for the efficient propagation of noise into the deep sound channel.

Low-frequency (defined by Celi *et al.* (2016)to be 6–3,000 Hz) ocean noise experiences low levels of absorption, allowing for long-range propagation over hundreds and thousands of kilometres (Brekhovskikh, 2003), the primary source of which is commercial shipping (Hildebrand, 2009). Less research and mitigation focus has been directed towards the chronic emissions of vessels, that the wider known intense, impulsive noise sources that have been suggested to cause mass strandings in marine mammals (Harris, 2017). This is a concern, because vessels are responsible for a major part of man-made noise pollution in coastal areas (Codarin *et al.*, 2009; Picciulin *et al.*, 2010), and are especially prevalent along major shipping channels, for instance large numbers of super tankers carry oil from Alaska to California (e.g. Wales and Heitmeyer, 2002). Noise from commercial shipping is generally confined to ports, harbours, and shipping lanes, whilst noise from other vessels (military vessels, fishing fleets, scientific research ships, and recreational craft) is more widely distributed (Firestone and Jarvis, 2007).

Higher levels of marine traffic have led to low-frequency background noise in the ocean increasing since the 1950's (Malakoff, 2010). Ross (1976) presented data that indicated noise levels had increased by 15 dB between 1950 and 1975 as a direct result of shipping activities. Several factors have contributed to this increase. The worldwide commercial fleet, consisting of tankers, dry bulk vessels, container ships, and other large ocean-going vessels, grew from approximately 30,000 vessels (~85,000,000 gross tons) in 1950 to 89,899 vessels (~605,000,000 gross tons) in 2003 (Ross *et al.*, 1993; Southall, 2005). As of 2014, the size of the world-wide fleet of large vessels (those of 100 gross tonnes or more) was 1669.7 million deadweight tonnage (M dwt) (Department for Transport, 2015). As of 2015, the global number of merchant ships alone, excluding passenger ships and fishing vessels, was estimated at just over 50,000 vessels (Harris, 2017). IHS Fairplay, a maritime news magazine and the leading source of critical maritime and trade insight, predicts that by 2030 the world fleet will be in excess of 2,500 M dwt (Pålsson, 2011; Det Norske Veritas AS, 2012). This equates to an average of 1,700 to 2,000 new vessels being built each year. Furthermore, port turn-around times are shorter. New efficiencies at container terminals such as modernisation of equipment allows for quicker operations (e.g. double cycling, tandem and multiple lift cranes) resulting in port turnaround times reducing from many days and even weeks to a matter of hours (Ducruet and Merk, 2013; Slack and Comtoise, 2015), so vessels spend more days per year at sea. Vessels also have greater average speeds, propulsion power, and propeller tip speeds, which all contribute towards greater ocean ambient noise levels (Ross, 1976; Ross *et al.*, 1993; Southall, 2005; McDonald *et al.*, 2006). In addition to the commercial fleet, there are also personal craft to consider. The numbers of these are less well known, but in the USA alone there are more than 12 million registered powerboats (National Marine Manufacturers Association, 2004), with a further 2.5 million in Canada (Canada National Marine Manufactures Association, 2006).

According to data from a number of researchers, the background noise level in the world's oceans is doubling every decade (National Research Council, 2003). As a result of increases in the volume of shipping and technological improvements in exploration and extraction of marine resources, anthropogenic noise levels within the marine environment are predicted to continue rising over the coming decades (United Nations Environment Programme, 2012). Rossi *et al.* (2016) claimed that the increasing number of vessels and ships represents one of the most evident threats for marine organisms.

Although ocean noise levels are said to have increased over the past decades, and with further rises predicted, the actual evidence of increased noise is limited to studies in the Pacific and Indian Oceans (Andrew *et al.*, 2011; Miksis-Olds *et al.*, 2013). Other oceans have not been studied in relation to marine noise trends, and so some researchers are unsure of whether this is in fact a global problem. Ambient noise levels in the North Pacific have increased. McDonald *et al.* (2006) reported a 10-12 dB rise in the 30-50 Hz frequency band over the past 50 years but the rate slowed from 0.55 dB/year to 0.2 dB/year in the 1980s (Miksis-Olds, 2016). Although overall trends are increasing, localised areas have shown decreasing noise trends, suggesting that the previously thought hypothesis of ocean sound increasing in a uniform manner may not be supported (Miksis-Olds *et al.*, 2013; Miksis-Olds and Nichols, 2016). It was concluded by Miksis-Olds *et al.* (2013) that the greatest increase in ocean noise was recorded in the 85–105 Hz band, suggesting that shipping noise is a large contributor to the increase, and many other researchers were of the same opinion (e.g. Andrew *et al.*, 2002; McDonald *et al.*, 2006; McKenna *et al.*, 2012). However, more recent research has seen sound levels decrease even though ship traffic is still increasing (Wilcock *et al.*, 2014).

1.3 Noise perceptions in fish

The turbid conditions present in much of the underwater environment results in many species relying upon sound, rather than vision, to decipher information about their surroundings. Fish interpret changes in ambient biotic and abiotic sounds from sound-scattering objects to create an 'acoustic view' (Popper and Fay, 1973; Bregman, 1990; Popper *et al.*, 2004). Sound can inform an individual as to the whereabouts of food, competitors, predators, and potential mates through the perception of intended and/or unintended acoustic signals in the environment (Myrberg, 1978). Abiotic – non-biological – sources encompass the sound of waves on the shore, geological events, current and winds, and raindrops on the water surface. Biotic – biological – sources include the sounds produced by conspecifics (other members of the same species), other fish species, marine mammals and invertebrates. Snapping shrimp are possibly the most ubiquitous source of background biotic noise in some parts of the ocean (Lagardère *et al.*, 1994; Tolimieri *et al.*, 2000; Popper and Hastings, 2009a, 2009b). Sound is an ideal means of communication in the aquatic environment for distances over several metres because, at a given frequency, the absorption of sound by water is far less than the absorption of light (Leighton, 2007). Furthermore, although the absolute value of the sound speed in water does not necessarily make sound a better transmitter over large distances in water, the variations from the average sound speed in water can be very great, leading to the formation of sound channels where the geometrical losses are much less than they would be if spherical spreading were to occur (Leighton, 1998). Even fauna that do not transmit sound over long distances underwater, and instead only communicate acoustically over short distances, can be affected by the efficiency of long distance acoustic transmission because it can bring noise from distant sources to their location, where they may be adversely affected by its presence. The other senses (vision, touch, smell and taste) are limited in range and/or speed of signal transmission (Tasker *et al.*, 2010). As fish use hearing to perceive and navigate their environment, the ability to create an auditory scene and correctly interpret the acoustic information gleaned, is crucial for their survival (Myrberg, 2001).

Fish use two independent but related sensory systems to perceive sound; the inner ear and the lateral line (Kikuchi, 2010). Together, the inner ear and the lateral line make up the octavo-lateralis system, which provides fish with balance, hearing, and the ability to feel vibrations from a distance (Popper, 2003). All fish perceive sound through use of the lateral line, with only certain fish species hearing via the inner ear (Zeddies *et al.*, 2013).

The lateral line system is used to detect local low frequency (below 150 Hz) water flow relative to the body, whilst the inner ear is equipped to detect motion and can sense sound pressure from an acoustic stimulus above about 50 Hz (Chapman and Hawkins, 1973; Sand, 1984; Andersson, 2011). The sensory hair cells found in the fish ear and in the lateral line rely on the movement of specialised hair cells to detect vibrations originating from other structures (Fay and Popper, 2000; Popper, 2003). The sound waves can travel through the body of the fish as fish tissue has an acoustic impedance similar to that of the surrounding water. Structures of differing density to the fish's body (such as otoliths or the air-filled swim bladder) are denser and so move slower in response to a sound wave causing displacement of the structures, which is interpreted by the brain as sound perception (Discovery of Sound in the Sea, 2017).

Some fish have evolved specialised hearing apparatus to improve their hearing capabilities. Ladich and Popper (2004) noted that sound pressure sensitivity has evolved simultaneously in many different fish families. At least a third of all teleost species have developed structures for sound pressure detection (Ladich and Schulz-Mirbach, 2016). The gasbladder (von Frisch, 1938; Yan *et al.*, 2000), gas-holding auditory ancillary structures, supra-branchial chambers (Ladich and Yan, 1998), otic gasbladders (von Frisch, 1938; Yan and Curtsinger, 2000), and the otic bullae (Denton *et al.*, 1979; Blaxter *et al.*, 1981) are some examples of specialised adaptations that have evolved to improve hearing capabilities (Figure 1.1). It has also been noted that the hearing sensitivity of fish depends on the degree of coupling or the proximity of these specialised structures to the fish's inner ear (Scholik and Yan, 2002).

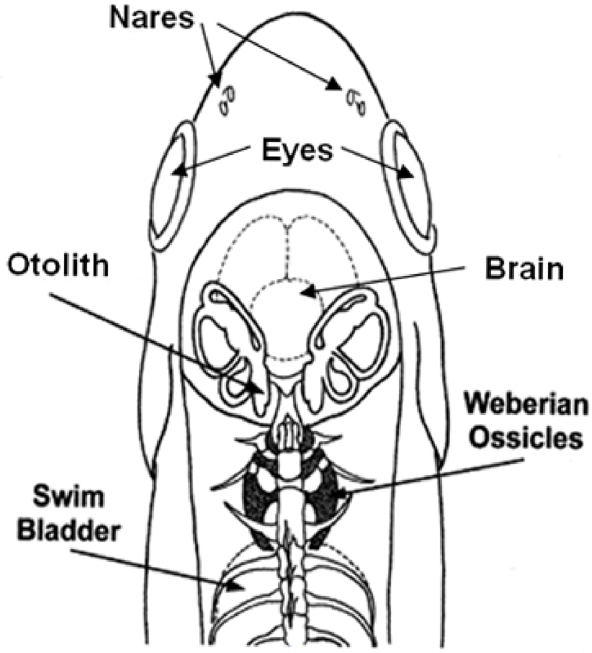

Figure 1.1. The ear anatomy and hearing apparatus of a fish. Specialised hearing apparatus such as the swim bladder and weberian ossicles allow fish to hear a broader frequency range. Species with the specialised apparatus are likely to suffer higher risk of noise impacts than those without (Popper *et al.*, 2014). Image from the Department of Environmental Conservation (*n.d.*).

The role that particle motion plays in the biology and ecology of species is still largely unknown (Nedelec *et al.*, 2016a). It is known, however, that if a noise source is more than a few metres away from an organism, noise may have less impact on species relying on particle motion, because it can only be detected over short distances, in a small frequency range and at higher sound intensities (Kunc *et al.*, 2016). Species relying on sound pressure detection, on the other hand, can detect sound pressure changes over large distances, and thus may be more vulnerable to increasing noise levels than species relying on particle motion alone.

In the past, fish species have been categorised either as a hearing generalist or a hearing specialist (depending on their hearing apparatus and consequent capabilities). However, these distinctions are becoming less used. Generalists are able to perceive narrow, low frequency sounds (up to 500–1,000 Hz), with optimal hearing ranging from 100 to 400 Hz. Hearing specialists tend to have the adaptations that improve their hearing capabilities, such as direct coupling devices or auditory ancillary structures (Scholik and Yan, 2002), and can hear over a much wider frequency range, detecting sounds over 3,000 Hz, with peak hearing sensitivity ranging from about 300 to 1,000 Hz. Some species are able to detect ultrasonic sounds over 200 kHz, such as fish in the family *Alosinae* (shads), who can detect and avoid echo-locating whales (Tavolga and Wodinsky, 1963; Popper and Fay, 1993, 1999; Mann *et al.*, 2001; Amoser and Ladich, 2003; Popper *et al.*, 2004; Dokseater, 2009). At the opposite end of the spectrum, field studies on European eel (*Anguilla anguilla*) and juvenile salmonids showed the ability to detect and avoid infrasound (< 20 Hz), in order to sense the hydrodynamic noise generated by approaching predators (Knudsen and Schreck, 1997; Sand *et al.*, 2000).

1.4 Impacts of noise pollution on fish

It is said that the most severe impact of all human activities is species extinction or loss of endemic populations, and marine species are increasingly being impacted by habitat change and the loss of crucial environmental resources (Braun, 2015). The paleontological record shows that organisms have coped with environmental distresses through great flexibility in response to social and physical environment challenges (Wingfield, 2013). For fish species survival, they need to endure perturbations of the environment – such as habitat change, invasive species and climate change – by having the ability to adapt and alter both physiological and behavioural processes. However, anthropogenic changes are often so rapid that evolutionary processes may struggle to keep pace with the ongoing changes, and if species cannot adapt or react quickly, then their survival could become compromised (Gomulkiewicz, 1995; Chevin and Lande, 2010). To prevent or minimise further damage or loss of species, we need to know the effects of our actions on wild fish populations, and whether fish have the inherent ability to respond adequately to avoid harm. If not, they will be totally reliant on mitigation measures to help them survive.

The impact of noise on aquatic species varies according to the properties of that noise. The frequency spectrum, the sound pressure level (SPL) at the receiver, duration, rise and fall times, and repetition rate can all affect the level of impact. If the source level increases then the noise has more energy and is likely to lead to greater impact, and the range of the noise's propagation increases potentially affecting a larger number of organisms (Hawkins *et al.*, 2014). The level of background noise present in the environment can alter the extent to which a sound can be distinguished (Holt and Johnston, 2014; Luczkovich *et al.*, 2016). The environmental features the noise moves through, such as bathymetry, temperature, and nature of the seabed can also affect the level of noise reaching the organisms (Farcas *et al.*, 2016). Finally, the hearing properties of the species themselves, such as species-specific or individual variation and sensitivity to noise, which may change through time or season, will influence the impact of the noise source (Kastelein *et al.*, 2008; Tasker *et al.*, 2010). Noise can cause subtle changes in a behaviour or be responsible for more pronounced physiological changes, including death, as has been proven in many laboratory and some field studies, in a variety of species (Table A1.1).

Noise pollution research has mainly centred on marine mammals but it is now a concern that fish and benthic species may be affected in similar ways (Popper *et al.*, 2007; Solan *et al.*, 2016). Previous research has shown evidence of both physiological and behavioural reactions of fish to noise, some of which are reviewed throughout this thesis. Research has covered both intense impulsive anthropogenic noise sources (e.g. pile-driving), and continuous low-frequency drones of chronic noise sources (e.g. vessels), a subset of which include tonal signals. Impulsive noise sources have a rapid rise time – a large change in amplitude over a short time – and are considered extremely damaging to auditory structures if amplitude is high at the listener (Tasker *et al.*, 2010). Anthropogenic noises like these may be of short duration, but can be repeated rapidly over prolonged periods of time. Chronic noise sources are now considered both continuous and intermittent, with a widespread range and potential to impact both physiology and behaviour of marine species. Both types of noise are specified within legislation and each needs research and mitigation.

The ways in which populations cope with increasing frequency and intensity of perturbations of the environment, and how some are able to rebound from these events is only just beginning to be explored (Wingfield *et al.*, 2011a, 2011b). When faced with environmental disturbances such as noise pollution fish have three options: to become more resistant to acute stress; to modulate the actual response to stress for more flexibility; or to be affected but recover quickly and completely once the noise has stopped (Wingfield, 2013). Mechanisms underlying these modulations remain largely unexplored. The most common response when faced with an adverse environment is to undergo physiological changes.

1.4.1 Physiological impacts of noise on fish

Research on physiological damage as a result of vessel noise pollution is still in its infancy, but the number of studies are increasing and some effects have already been observed. There is evidence for a whole range of potential physiological effects resulting from noise pollution (Table A1.1) including instantaneous death (Popper and Hastings, 2009a). A less obvious effect that has been observed in fish is tissue damage (Table 1.1); noise induced tissue damage has not to date been observed to lead directly to death, but it can affect the fitness of the fish, such as prolonging recovery or healing time, which can result in lower chances of survival (Popper and Hastings, 2009a).

Table 1.1. Some examples of tissue damage injuries recorded in fish exposed to anthropogenic noise sources. Images are taken from Halvorsen (2012) and Casper (2012).

Severity of injury	Examples	Available images to demonstrate injuries
Mild	Eye haemorrhage (A) Fin haemorrhage (B & C) Fin haematomas Deflated swim bladder	  
Moderate	Liver haemorrhage (D) Bruised swim bladder (E) Swim bladder haematoma (F) Burst capillaries Muscle haematomas Ovaries haematoma Fat haematoma	

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<tr><td>Mortal</td><td>Kidney haemorrhage (G)
Intestinal haemorrhage (H & I)
Ruptured swim bladder</td><td>


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1.4.1.1 Hearing

Exposure to noise can lead to hearing loss, which is currently used to study impacts of noise pollution on animals (e.g. Scholik and Yan, 2002). In order to use this metric one must first measure the baseline hearing threshold that represents the lowest signal an animal can hear (Kikuchi, 2010). The method for measuring hearing thresholds is known as the auditory evoked potential (AEP) or auditory brainstem response (ABR) recording technique. This technique records synchronous neural activity of eight nerve fibres and brain stem auditory activity evoked by acoustic stimuli (Jewett, 1970). Once this baseline is known, any changes from this threshold can be recorded. Such changes in hearing sensitivity have the potential to lower fitness until hearing recovers (Enger, 1981; Scholik and Yan, 2001; Amoser and Ladich, 2003; Smith, 2004; Popper *et al.*, 2005; Codarin *et al.*, 2009; Cott *et al.*, 2012). Through the use of Auditory Evoked Potential audiograms fish have been observed to detect playback of powerboat noise (running at a top speed of 270 km/h) up to 400 m from the noise source which provides evidence that vessel noise can disturb fish even at a distance (Amoser *et al.*, 2004 - vessel; 103-128 dB re 1 µPa; 100-4000 Hz). Temporary changes in sensitivity are known as temporary threshold shift (TTS) and are treated as a conservative measure of the threshold for injury from noise. This type of injury is considered temporary as the hair cells of fish are normally able to regenerate (Popper *et al.*, 2004). A study by Smith (2004 - white noise; 160-170 dB re 1 µPa; 100-10,000 Hz) investigating TTS saw a 28 dB shift in the hearing threshold of a Goldfish after 24 hours of 160-170 dB re 1 µPa noise exposure (50 dB more than the ambient noise). In this case, hearing thresholds returned to their normal state within 14 days after exposure. Auditory thresholds of 3 species (Red mouthed goby (*Gobius cruentatus*), Mediterranean damselfish (*Chromis chromis*) and the Brown meagre (*Sciaena umbra*)) were tested using 132 dB re 1 µPa playback noise from an 8.5 m cabin cruiser. The cabin cruiser (10 m away from the hydrophone) increased the local ambient noise by 40 dB re 1 µPa which has serious implications for the detection distance of a biologically important signal (Codarin *et al.*, 2009 - vessel; 132-138 dB re 1 µPa; 300-10,000 Hz). The authors predicted that the detection distance of conspecific sounds can be reduced between 10 to more than 100-fold depending on the species. The auditory thresholds of the three fish species increased by up to 10 dB, 20 dB and 35 dB respectively meaning that masking – when the detection of one sound is impaired by the presence of another – is more likely to occur in noisier environments as detection distances of conspecific sounds can significantly decrease (Codarin *et al.,* 2009).

A more serious effect of noise is permanent threshold shifts (PTS), at which point internal organs are damaged, resulting in irreversible damage to sensory hair cells (Popper and Hastings, 2009b). This has been comprehensively studied in mammals (e.g. Saunders *et al.*, 1991; Henderson *et al.,* 2008) including humans, and has been observed in a few fish species (e.g. codfish (*Gadidae*), Oscar (*Astronotus ocellatus*), Flathead minnow (*Pimephales promelas*), Pink snapper (*Pagrus auratus*)) exposed to noise levels of 142-180 dB re 1 µPa for up to 24 hours (Enger, 1981 - source not stated; 180 dB (units not given); 50-400 Hz; Hastings *et al.*, 1996 - white noise; 100-180 dB re 1 µPa; 60-300 Hz; Scholik and Yan, 2001 - white noise; 142 dB re 1 µPa; 300-2000 Hz; McCauley *et al.*, 2003 - air gun; 222.6 dB re 1 µPa peak to peak; 20-1000 Hz). McCauley *et al.*

(2003 - air gun; 222.6 dB re 1 μPa peak to peak; 20-1000 Hz) found that noise caused hair cells to be ripped away, or die, and there was no evidence of repair/replacement of the damaged hair cells up to 2 months post-exposure. As cell damage can take time to manifest, short term studies would not give reliable results (Hastings *et al.*, 1996), and so long-term research is needed to test PTS to truly identify the causes of permanent damage. Hastings *et al.* (1996 - white noise; 180 dB re 1 μPa) exposed five fish to one hour of continuous 100 Hz noise at 180 dB re 1 μPa and suggested that the low levels of hair cell damage observed in their study may have been the result of ending their experiment after only four days, which meant they only recorded the initial stages of cell loss. In addition to hair cells, other structures and organs in fish can also suffer noise damage: gas oscillations induced by high SPLs can potentially cause the swim bladder to tear or rupture, especially when in conjunction with explosions (e.g. Govoni *et al.*, 2003 - explosion; ~2.0 to 8.7 Pa·s; Hz not stated); and blood and tissues can be filled with bubbles from micronuclei in response to noise, which can lead to embolisms and internal bleeding (Halvorsen *et al.*, 2012a – pile-driving; 204-216 dB re 1 $\mu Pa^2 \cdot s$; 0-1200 Hz, 2012b – pile-driving; 204-220 dB re 1 $\mu Pa^2 \cdot s$; 0-1200 Hz).

1.4.1.2 Growth and development

The physiological processes of fish can be adversely affected at vital life stages; when in the presence of noise 15 dB higher than the ambient noise, fish eggs and embryos suffered increased mortality and the surviving fry were observed to undergo slower growth rates (Banner and Hyatt, 1973 - white noise; specific dB not stated; 100-1000 Hz). Evidence of the effect of noise on fish egg and larval development is still limited with few studies published thus far (Simpson *et al.*, 2005, 2016a; Davidson *et al.*, 2009; Jung and Swearer, 2011; Holles *et al.*, 2013; Bruintjes and Radford, 2014; Bolle *et al.*, 2016), but if noise is proven to affect growth and development it could have serious consequences (e.g. being undersized could increase the risk of being eaten by predators). Eggs receive noise via two pathways – through the water column (via sound pressure waves) and the substrate (via particle motion) – and are at a higher risk of damage as they are usually completely sessile, and therefore, unable to avoid the noise source (Solan *et al.*, 2016). After exposure to seismic noise at 3 second intervals, 46 % of 4881 scallop larvae tested showed abnormalities compared to 0 % of the control condition (Soto *et al.*, 2013). Additionally, after 24 hours of exposure to noise significant developmental delays were observed.

Spiga *et al.* (2012b) showed in one study that growth remained uninfluenced by vessel noise, in particular playback of a sport fishing vessel at 175 dB re 1 μPa (Spiga *et al.*, 2012b - vessel; 175

dB re 1 μPa; Hz not stated). However, studies investigating impacts of vessel noise on growth are few; only 15 of the 121 studies listed in Table A1.1.

1.4.1.3 Physiological processes

Noise exposure can induce a biochemical or neuroendocrine response in fish when the stress hormone cortisol is released. This has been observed when fish are subjected to a noise source such as vessels (Wendelaar Bonga, 1997; Smith *et al.*, 2004 - white noise; 160-170 dB re 1 μPa; 100-10,000 Hz; Wysocki *et al.*, 2006 - white noise & vessel; 153-156 dB re 1 μPa; 0-5000 Hz). Data collected has confirmed that vessel noise can stimulate increased cortisol secretion in 3 freshwater species (perch (*Perca fluviatilis*), gudgeon (*Gobio gobio*) and carp (*Cyprinus carpio*)), and 2 marine species (Red drum (*Sciaenops ocellatus*) and Brown trout (*Salmo trutta*)) (Wysocki *et al.*, 2006 - white noise & vessel; 153-156 dB re 1 μPa; 0-5000 Hz; Spiga *et al.*, 2012a - vessel; 175-180 dB re 1 μPa; Hz not stated). However, recovery to normal secretion levels has been recorded 60 minutes after initial noise signal, even if the noise exposure continues (Spiga *et al.*, 2012b). The effects of raised cortisol levels are not fully understood, but it is hypothesised that long term rises in cortisol could be harmful to the individual, as cortisol plays an important role in osmoregulation, growth and reproduction in fish (Mommsen *et al.*, 1999). Elevated cortisol levels have been correlated with other detrimental effects also, including increased susceptibility to infection (Anderson, 2009), decreased growth rates, reduced predator avoidance ability, and increased ventilation rate (McCormick *et al.*, 1998; Woodley and Peterson, 2003). Changes in ventilation rate also correlates with other detrimental effects, including alterations in oxygen consumption, heart rate and plasma cortisol (Wendelaar Bonga, 1997; Simpson *et al.*, 2015). Even though a stress response alone may not be considered a serious effect, it is a precursor for many other health problems.

Heart rates of embryonic clownfish (*Amphiprion ephippium* and *Amphiprion rubrocinctus*) have been monitored and found to increase significantly with vessel noise exposure (Simpson *et al.*, 2005 - white noise; 80-150 re 1 μPa at 1 m; 100-1,200 Hz). Alterations in heart rate have also been observed in adult fish (in particular largemouth bass) which demonstrates that fish experience sub-lethal physiological disturbances in response to the noise propagated by recreational boating activities (Graham and Cooke, 2008 - vessel; dB not stated; Hz not stated). Metabolic rates have also been shown to increase during noise exposure (Simpson *et al.*, 2016a - vessel; dB not stated; Hz not stated).

Physiological responses can differ depending on the gender of the individual; Kight and Swaddle (2011) stated that "noise stress seems to be particularly damaging to females, a relationship that likely stems from sex differences in size, hormone expression and the costs of reproductive investment". Studies in other aquatic animals have shown lower reproductive rates, fewer egg bearing females, decreased food consumption, and faster metabolism (through increased excretion of ammonia and consumption of oxygen), and consequently increased energy costs (Lagardère, 1982; Regnault and Lagardère, 1983).

1.4.2 Behavioural impacts of noise on fish

Physiology is an important aspect for survival; as with any animal, when a fish is physiologically damaged it becomes easy prey, and is less able to hunt. However, if a behavioural reaction of some kind were to remove the fish from harm in the first place, no physiological effects would ever be observed. Behavioural audiograms are used to study impacts of noise on behaviour by looking for behavioural reactions at different noise frequencies and intensities. Behaviour impacts can be immediate (e.g. startle responses (Purser and Radford, 2011)) and substantial through altering an individual's behaviour (e.g. displacement (Engås *et al.*, 1996)) and can have long term consequences on the fitness and ecology of a species (Wysocki *et al.*, 2006). Long term behavioural responses are often due to chronic low level noises, whereas short term behaviours are more of a 'fight or flight' response to acute or intense noise. Little data exists with regard to the effects on long term behaviours as they are much harder to observe than short term behavioural or physiological responses (Popper *et al.*, 2014). Behaviours that affect fish on an individual level could have a 'knock on' effect for the population if sufficient individuals are affected. Therefore, it is important to identify whether underwater noise causes a temporary, and recoverable (such as increased territoriality when exposed to noise), or a more permanent/long-term change in behaviour (such as displacement from breeding grounds).

At the population level, it is the cumulative impact of behavioural changes within and among individuals, that form a serious threat from noise (Tasker *et al.*, 2010). The presence of noise at any stage of the life cycle can have a disadvantageous effect on a population but it is difficult to suggest any particular life stage as the focus of research; harmful effects at the larval settlement and development stage mean fish may never reach adulthood, whereas in the adult life stage reduced foraging or spawning would mean no larvae would be produced in the first place.

1.4.2.1 Avoidance

Avoidance is the behavioural response generally associated with vessel noise. An informative study by Suzuki *et al.* (1980) demonstrating avoidance behaviours of anchovy was one of the first studies of noise pollution impacts on fish behaviour. Fish were exposed to playback of a fishing vessel in a laboratory at levels of 120-140 dB re 1μPa. The results showed that from 130 dB re 1μPa avoidance behaviours, in the form of darting several metres down following exposure, were recorded. It was concluded that avoidance could be observed up to 400 m from a large tanker (Suzuki *et al.*, 1980 - vessel; 120-140 dB re 1 μPa; Hz not stated). Cod and haddock, two important commercial species, have been observed reacting to approaching vessels from a distance of 200 m, including 200 m depth (Ona and Godø, 1990 - vessel; dB not stated; Hz not stated), and Atlantic cod (*Gadus morhua*) has been observed, by Buerkle (1977) (as cited in Handegard *et al.*, 2003), reacting to a trawling vessel at a range of at least 2.5 km, showing that vessel noise has a larger sphere of impact on behaviours. Avoidance has been noticed by fisheries as fish avoid noisy vessels causing catches to diminish; Olsen (1971) noted that vessel-induced avoidance behaviour adversely affected fishing success in Norwegian purse-seine fishing for herring (as cited in Kastelein *et al.*, 2008). More recently, Atlantic cod have shown horizontal and vertical movements away from vessels (Handegard *et al.*, 2003 - vessel; dB not stated; Hz not stated), and tuna schools were less coherent and significantly more likely (significance of $p < 0.05$ reported for study of ~ 100 wild tuna) to disperse when in the presence of vessel noise, than when no noise was present (Sarà *et al.*, 2007 - vessel; 100-135 dB re 1 μPa; 70-20,000 Hz). Diving responses to vessel noise are also reported in the literature. Walleye pollock (*Theragra chalcogramma*) have been observed diving in response to noise, but this displacement has been modest – typically 5 m (De Robertis *et al.*, 2008 - vessel; dB not stated; 0-2,000 Hz). Additionally, individual fish often swim independently towards the surface or the bottom, whilst avoiding the middle layer of their habitat, and show restless behaviour. The degree of avoidance observed can depend on time of day; vessel avoidance behaviour displayed by wintering herring proved highly significant at night, when high densities were found in the upper 100 m, but non-significant during the day when the herring were at greater depths (Vabø *et al.*, 2002 - vessel; dB not stated; Hz not stated) This study, however, did not consider noise emissions specifically.

As with other areas of marine noise pollution research, there are subjective and contradictory views about the impacts of avoidance behaviour. It has been suggested by some that the presence of a moving survey vessel can impact the behaviour of fish, which may in turn influence vessel-based observations and that reactions may be due to the vessel hull itself, not the noise being emitted from it (De Robertis and Handegard, 2013). Gerlotto *et al.* (2004 - vessel; dB not stated; Hz not stated) suggested that the recorded 5 m dive by Peruvian anchovy (*Engraulis ringens*) and Common sardine (*Strangomera bentincki*) was due to avoidance of the vessel hull itself and not the noise being produced. Some studies have reported that fish move towards the vessel as it approaches suggesting there is no adverse avoidance reaction. However, work by Handegard and Tjøstheim (2005) showed that the movements of fish relative to an approaching vessel are more complex than previous research assumed; fish move both away from and towards the vessel as the properties of the sound field around the vessel change. However, ships can produce a non-uniform 'butterfly' pattern of radiated noise in the lateral plane which occurs at higher frequencies, with maximal levels to each side of the vessel because of shading of propeller noise by the hull (Urick, 1983). It was, therefore, suggested that observations of fish approaching ahead of the vessel could be a result of fish moving away from areas of high radiated noise on the sides of the vessel (Misund *et al.*, 1996). For these reasons, noise reduced vessels were designed for conducting surveys. Specialised vessel designs such as diesel-electric propulsion, fixed pitch propellers and quiet hulls have substantially decreased noise levels over a wide frequency range (De Robertis and Handegard, 2013). A noise reduced vessel could provide evidence for whether fish are avoiding vessel noise, or just the presence of the vessel in general. In a comparison study, the noise reduced vessel caught more fish that the normal fishing vessel showing that less avoidance occurred in response to the quieter vessel (Engas, 1991). The radiated noise from the noise-reduced fisheries research vessel (FRV) Johan Hjort was responsible for avoidance behaviour of herring schools at a distance of 540 m when the propeller shaft was operated at a speed of 125 rpm (Mitson and Knudsen, 2003 - vessel; 165 dB re 1 μPa; 18,000, 120,000, 200,000 Hz).

Avoidance of favourable habitats may result in a decline of fish condition and individuals may be forced to perform behaviours usually perceived as risky to survive. For example, if a fish is starving it may be driven to forage in a noisy area, and suffer increased damage from the noise owing to its already deteriorated condition (Sweitzer, 1996). In addition, fish trying to avoid the metabolic and energetic issues of disturbance, may move out of the noisy area altogether. This could result in larger populations in quiet areas creating more intraspecific competition in suboptimal habitats with limited resources (Gill and Sutherland, 2000). Added to which, fishing vessels would focus on the larger fish populations found in the quieter areas thus overexploiting the species fleeing from the noisy environment, whilst bringing more vessel noise into the area. If noise causes avoidance behaviour and fish move away from the area that their prey inhabits then foraging performance would decrease, causing lower survival rates and ultimately a decrease in population. Displacement of fish from usual habitats, or higher noise-induced mortality rates, would mean that higher members of the food chain and their populations might be negatively impacted also, resulting in long-term consequences for the whole ecosystem.

1.4.2.2 Predation

Research has also been carried out on anthropogenic disturbance and predation risk (Frid and Dill, 2002). The idea behind this concept, known as the 'risk-disturbance hypothesis', is that human disturbances take time and energy away from fitness enhancing activities such as foraging or spawning in much the same way as predation risk (Frid and Dill, 2002). When a fish comes into contact with a noise source its response should follow the same economic principles as when prey encounter predators (Gill and Sutherland, 2000). The model predicts that responses to disturbance will be stronger when the perceived risk is greater. Frid & Dill (2002) noticed that escape behaviours increased when the 'disturbance' approached quicker, more directly, and was larger in size. Therefore, anti-predator behaviours may increase when the distance to a safe refuge and the group size of the disturbers are greater, as perceived risk is larger. The resulting trade-off between perceived risk and energy intake can indirectly affect survival and reproduction (Gill and Sutherland, 2000); underestimating danger can result in instant death so individuals would avoid feeding if danger is considered present (Bouskila and Blumstein, 1992). This behaviour results in loss of body condition and reduces both survival and reproductive success (Hik, 1995). Another study showed anti-predator behaviour also diminished in the presence of vessel noise; simulated predators could get closer during noise playback (Simpson *et al.*, 2015 - vessel; 148 dB re 1 μPa; Hz not stated; 2016b; vessel - 70-125 dB re 1 $\mu Pa^2\ Hz^{-1}$; 0-3,000 Hz)). For vocalising species, the predator-prey interaction can be impacted by noise through the masking of signals that would usually aid in location of prey or avoidance of predators.

1.4.2.3 Masking

Masking can result from noise limiting the detection of critical acoustic information, such as the signals made by prey, so making it harder to find food (Siemers *et al.*, 2007). It can also decrease foraging ability or performance through distraction. A narrowing of attention can cause fish to ignore peripheral areas or other stimuli in the area (a behaviour known as stimulus-selective attention or spatially selective attention) with detrimental costs (Dukas, 2002). The resultant behaviour of stimulus selective attention is reduced foraging rate, as more time is needed to forage an area. This reduced foraging efficiency and the lack of vigilant attention to predators means chances of survival decrease (Dukas and Kamil, 2001). Fish have been observed to react more slowly to simulated predator attacks in the presence of noise (Voellmy *et al.*, 2014b - vessel; specific dB not stated; 100-3,000 Hz). The response time seemed to vary among species within the same geographic area, further evidence of the individuality of each species' reaction to noise. Fish suffer predation quicker in noisier conditions with the predator being 82 % more successful (Simpson *et al.*, 2016b - vessel; dB not stated; 0-3,000 Hz). Reasons for the lack of attention to essential tasks may be due to individuals finding it difficult to focus on signals from approaching prey, or by diverting their attention toward the noise source rather than the prey (Purser and Radford, 2011 - white noise; 48-150 dB re 1 μPa; 100-1,000 Hz). The consequences of distraction are: that foraging performance and efficiency decrease; the likelihood of being preyed upon whilst foraging increases; and the overall amount of foraging time spent will be greater in order to gain the necessary food intake. This in turn will force the individual to spend longer in the open predatory environment.

However, as with other behavioural experiments, there is opposing evidence. A contradictory result was that of Purser and Radford (2011 - white noise; 48-150 dB re 1 μPa; 100-1,000 Hz) whose study disagrees with other work on the impacts of noise on foraging (e.g. Bracciali *et al.*, 2012 - vessel; dB not stated; Hz not stated; Spiga *et al.*, 2012a - vessel; 175, 180 dB re 1 μPa; Hz not stated, 2012b - vessel; 175 dB re 1 μPa; Hz not stated). Noise caused startle behaviours in Three-spined sticklebacks (*Gasterosteus aculeatus*) but had no effect on the time spent freezing or hiding compared to a silent control, so there was no significant impact on the total amount of food eaten.

Hearing is the basis of acoustic communication. Fish demonstrate the largest diversity of sound-producing mechanisms found in vertebrates (Ladich and Fine, 2006). Over 800 fish species from more than 109 families are known to vocalise, including some key UK commercial species such as cod and haddock, with many more suspected of doing so (Slabbekoorn *et al.*, 2010; Casaretto *et al.*, 2016a). It is unknown how widespread sound production is amongst fishes, and Hawkins *et al.* (2015) suggested it is likely to be far more extensive than current evidence implies. Fish vocalise for a variety of reasons (Table 1.2). Fish communication signals tend to be <500 Hz (Slabbekoorn *et al.*, 2010), but vary depending on species (Kihslinger and Klimley, 2002; Verzijden *et al.*, 2010), populations (Parmentier *et al.*, 2005), and gender (Ueng *et al.*, 2007). For example, clownfish have been recorded as having differing vocalisations depending on their geographic location; the repertoire of the Madagascar clownfish (*Amphiprion latifasciatus*) consists of chirps, short pops, and long pops, but clownfish in Indonesia do not produce chirps (Parmentier *et al.*, 2005). Variation in pitch and duration has shown correlation with size, season, motivation and age (Myrberg *et al.*, 1993; De Jong *et al.*, 2007; Maruska and Mensinger, 2009; Verzijden *et al.*, 2010).

Masking of vocalisations has been proven to occur in terrestrial animals (e.g. Fletcher, 1940), and it is possible it also occurs in fish. Noise from a source such as shipping can produce noise at frequencies similar to that of fish vocalisations, and in the same critical bandwidths, so vessel noise pollution could directly affect the survival of fish populations by decreasing their ability to hear and respond to biologically important signals. It is hypothesised that noise pollution could affect an individual or population by limiting their ability to communicate. Masking from anthropogenic sources could interfere with vocalisations (such as those used for courtship or coordination), or reduce the distance at which these biologically important vocalisations can be detected by the fish, limiting a fish's ability to find a mate (Holt and Johnston, 2014; Luczkovich *et al.*, 2016). This masking occurs because the introduced noise raises the ambient level and decreases the signal-to-noise ratio, which reduces the signal detection distance and makes identifying signals more difficult (Andersson, 2011). Oyster toadfish (*Opsanus tau*) were observed to stop vocalisations when exposed to vessel noise; calling rates changed from 7 calls/minute in normal conditions to 3.7 calls/minute in noisy conditions (Luczkovich *et al.*, 2016 - vessel; dB not stated; Hz not stated). The lack of signals between conspecifics means that behaviour necessary for a population's survival may not naturally occur.

Table 1.2. Examples of uses of vocalisations.

Use of vocalisation	Example/Reference
to attract mates	damselfish – Parmentier *et al.*, 2006
	blennies – De Jong *et al.*, 2007
	damselfish – Mann, 1995
	drums – Locascio *et al.*, 2012
	cod – Rowe and Hutchings (2006)
	haddock – Casaretto *et al.* (2016)
to establish territory	toadfish – McKibben and Bass, 1998
	minnows – Nicoletto and Linscomb, 2007
whilst foraging	gurnards – Amorim, 2004
	seahorse – Anderson, 2009
when competing for food	cichlids – Lamml and Kramer, 2008
	piranhas – Kastenhuber and Neuhauss, 2011
as a fright response	croakers – Lin *et al.* 2007
to aggregate for spawning and synchronise the release of gametes	catfish – Papes and Ladich, 2011

Wollerman & Wiley (2002) suggested that noisy conditions can interfere with mate selection. Mating calls masked by unnatural noise means that only the loudest individual, or an individual displaying a certain pitch, will be heard, and therefore, mate successfully. This could lead to issues such as a diminished population. Studies on the Blacktail shiner (*Cyprinella venusta*) showed that fish vocalisations were more common when exposed to an increase of 10.2 dB above the control decibel level, and call intensity also increased (Holt and Johnston, 2014 - white noise; 10.9-16.93 dB above ambient; 80 Hz). When heterospecific (between species) vocalisations are masked, foraging is energetically more costly as prey are harder to locate, and predation risks are higher as warning sounds from approaching predators may be masked. Non-vocal species may use the vocalisations of other species as an aid to navigation; sharks monitor the sounds of struggling fish in order to locate and capture them as prey (e.g. Myrberg *et al.*, 1976 observed sharks orientate towards sounds of struggling fish from distances of 125 to 400 m away). Any noise in the environment would, therefore, decrease the shark's chances of successful foraging, and vessels have the potential to mask food patches' sounds from 100 m away (Radford and Montgomery, 2016).

Furthermore, masking can cause problems for parents tending their young as the adults must be able to hear the begging calls of progeny or cues will be missed and the offspring's survival may be compromised (Kilner and Hinde, 2008). Territoriality can also be adversely affected by masking as many fish, such as the Sergeant fish (*Abudefduf saxatilis*), are known to mark and defend their territory using sound (Maruska *et al.*, 2007). The Red Mouthed goby was observed to reduce territoriality in the presence of boat noise, and the resultant increased aggressive contests had a subsequent detrimental effect on reproduction (Sebastianutto *et al.*, 2011 - vessel; 161 dB re 1 μPa; 0-800 Hz). Some marine mammals have simply stopped vocalising altogether in response to anthropogenic noise (Weilgart, 2007a).

Throughout the paleontological record there are examples of species having evolved to overcome the perturbations they faced. Red mouthed goby, Brown meagre and Mediterranean damselfish, for example, all significantly increased their detection threshold levels to hear conspecific signals when exposed to 132 dB re 1 μPa cabin-cruiser noise reproduced in the laboratory (Codarin *et al.*, 2009 - vessel; 132-138 dB re 1 μPa; 300-10,000 Hz). However, as noise in the ocean increases some species may not be able to adapt. If a species is unable to alter its hearing threshold then an alternative solution to overcome masking needs to be found. Currently, over 77 fish species, for which audiograms exist, are known to have hearing thresholds within the same frequency range as the noise produced by vessels (*see* Figure A1.1 and Figure A1.2 for more information). The exact frequency spectrum of the noise alters depending on the type of vessel (*see* Section 1.2 for more information on vessel characteristics and noise emissions). Some individuals may compensate for the increase in vessel noise by changing the amplitude, duration, repetition rate, and / or frequency of the sounds they produce (Foote *et al.*, 2004; Scheifele *et al.*, 2005; Jensen *et al.*, 2009). An effect, which has the potential to overcome the impacts of masking, is known as the Lombard effect – 'the automatic and involuntary change in the intensity of vocalisations in the presence of background noise in order to maintain a constant signal to noise ratio' (Coffey, 2012). To date, the Lombard effect has not been greatly studied in fish, but other animals have shown that this effect can help overcome problems caused by masking (Table 1.3).

1.4.2.4 The Lombard effect

Research to determine whether the Lombard effect occurs in fish has only recently been conducted and only on 3 species (Coffey, 2012 - Blue botia (*Yasuhikotakia Modesta*); Holt and Johnston, 2014 - Blacktail shiner; Luczkovich *et al.*, 2016 - Oyster toadfish). If this phenomenon occurs in all species, then serious concerns regarding the effects of vessel noise on vocalising fish populations may be unfounded. Altering vocalisations, however, may be metabolically expensive, and necessitates that the fish communication range is not already maximised (Jensen *et al.*, 2009). Furthermore, it has been hypothesised that fish vocalisations are dependent on the size of the fish and so some individuals may not physically possess the ability to alter their vocalisations. The occurrence of the Lombard effect may differ according to species or/and individuals, the type of noise source, its frequency and intensity, or with other factors such as season and topography. Studies to determine whether the Lombard effect occurs in fish should seek to assess a range of environmental conditions under which the effect may occur, and also relate these to fish lifecycle stages. Any masking problems encountered during reproductive phases need to be addressed in particular as there could be implications for population survival. Other studies have found alterations in vocalisations occurring such as change in rate of calls in response to noise, and increasing or ceasing vocalisations in noisy periods (Luczkovich *et al.*, 2016). It is also possible that by the fish adjusting its depth in the water column (thereby changing the propagation of the vocalisation) could help it overcome masking and thus counteract the need for the Lombard effect to occur (Forrest, 1994).

If it is proven that some fish species alter the pitch, intensity or duration of their vocalisations in order to remain audible to conspecifics, then behaviour such as aggregated migration, courtship and spawning can occur unhindered in the presence of underwater noise pollution for those species. However, if the Lombard effect is not observed in some species, then the masking of vocalisations by anthropogenic noise should be regarded as a serious threat to vocalising fish species.

Table 1.3. Studies that show evidence of the Lombard effect being used to overcome masking.

Class	Taxa	Observed Effects	Reference
Birds	Free-ranging nightingale (*Luscinia megarhynchos*)	Males were recorded singing at higher levels when in noisier locations	Brumm, 2004
	Ovenbird (*Seiurus aurocapilla*)	Road traffic can affect mate choice and subsequently affect pairing and mating success	Habib *et al.*, 2007
	Great tits (*Parus major*)	Urban great tits at noisy locations sing with a higher minimum frequency	Slabbekoorn and Peet, 2003
Arthropods	Ground-dwelling wolf spider (*Schizocosa ocreata*)	Females, who communicate via vibrations, showed changes in courtship behaviour, receptivity and mating success when exposed to white noise	Gordon and Uetz, 2012
Mammals	St. Lawrence River beluga (*Delphinapterus leucas*)	Increased the intensity of vocalisations in the presence of boat noise	Scheifele *et al.,* 2005
	Killer whales (*Orcinus orca*)	Respond to vessel noise by increasing their vocalisations by 1 dB for every 1 dB increase in background noise	Holt *et al.*, 2009
Amphibians	Brown tree frogs (*Litoria ewingii*)	Altered the pitch of their advertisement calls in the presence of traffic noise	Parris *et al.*, 2009
	Common eastern froglets (*Crinia signifera*)	Frogs decreased their call rate when exposed to aeroplane or motorcycle engine playbacks, and suggested that the frogs were changing their calling behaviour to avoid acoustic masking	Sun and Narins, 2005
Fish	Blue botia (*Yasuhikotakia modesta*)	Increased their vocal amplitude in the presence of 120 dB re 1μPa white noise	Coffey (2012)
	Blacktail shiner (*Cyprinella venusta*)	Increased intensity of vocalization when exposed to noise 10.2 dB above ambient levels	Holt and Johnston (2014)

1.4.2.5 Foraging

Noise pollution has the potential to affect individual energy expenditure through displacement from an area, reducing boldness, or by increasing foraging effort (Purser and Radford, 2011 - white noise; 48-150 dB re 1 μPa; 100-1,000 Hz; Holmes *et al.*, 2017 - vessel; 60-110 dB re 1 $\mu Pa^2 s$; 100-1000 Hz). If noise prevents a fish from foraging then it could decrease that individual's chance of survival. Negative effects on foraging performance and food handling have recently been observed in various fish species (Purser and Radford, 2011 - white noise; 48-150 dB re 1 μPa; 100-1,000 Hz; Sabet *et al.*, 2015 - white noise; 122 dB re 1 μPa; 300-1,500; Nedelec *et al.*, 2017 - vessel; 128 dB re 1 μPa; 0-2,000 Hz). It has been suggested that success averaged over a series of foraging events may be of more importance than the individual results from each single foraging trip or period (Costa, 2012). If true, then continuous noise, such as that produced by vessels, could have implications in or around important feeding grounds as long-term noise could reduce average foraging success (Voellmy *et al.*, 2014 - vessel; specific dB not stated; <5,000 Hz). Even if fish are generally able to forage successfully in the presence of noise, in situations where resources are less abundant or when the individual is in a vulnerable life stage such as larvae or mid-migration, and their ability to forage may already be drastically reduced, the additional presence of noise could have a more devastating effect (Costa, 2012).

1.4.2.6 Larvae

The larval stage is a crucial time for survival in marine animals' lives. As larvae, many organisms are highly susceptible to predation as they drift in the ocean currents (their ability to swim being limited), making them easy prey. Noise pollution from anthropogenic sources can interfere with the larvae's ability to undergo settlement and recruitment processes as they use the ambient ocean noise to orientate and locate desirable habitats – different habitats having distinct acoustic signatures (Montgomery *et al.*, 2006; Radford *et al.*, 2010; Stanley *et al.*, 2012). When exposed to vessel noise larvae were observed to experience greater difficulty finding a suitable settlement habitat, and increased likelihood of mortality (Holles *et al.*, 2013 - vessel; 77.71 dB re 1 μPa; 200-3,000 Hz). If the larvae are unable to locate a site to settle in, they will die or become more susceptible to predation. This would have an impact on the species' population if it happened to multiple individuals in the same year or spawning cycle. Following Cushing's (1975) 'single process' principle, the longer the larvae stay in a stage of high mortality, then the more likely a higher overall population mortality will result. In a series of experiments on coral reef fish, Simpson *et al.* (2016b; vessel - 70-125 dB re 1 μPa^2 Hz^{-1}; 0-3,000 Hz) investigated the impacts of vessel noise on survival and found that in the presence of boat noise only 27 % of 39 larval fish managed to survive the observation period, compared to 79 % of 39 larval fish surviving in the control condition ($p < 0.001$ significance level reported). Although some behavioural responses such as a startle or escape response may be short-term, other behaviours such as reduced foraging efficiency are likely to have a longer-term impact (Nedelec *et al.*, 2015).

In a field experiment, using wild free ranging fish, a powerboat with a frequency of 8,000 Hz and noise levels of 107-111 dB re 1 μPa caused no reactions in 11 larval reef species (Jung and Swearer, 2011; vessel; 107-111 dB re 1 μPa; 8,000 Hz). This contradicts several other studies. The limitation of Jung and Swearer's study was that they used one frequency range, audible only to specialist hearers, and played the noise signal for just 14 seconds. Furthermore, the reef may have absorbed some of the noise before it reached the test area as the noise source was 30 m from the observation area.

1.4.2.7 Migration

Both larvae and adult stage fish are known to migrate over large distances to mating and /or feeding grounds (Stevick *et al.*, 2002). These migration routes can cover vast distances, for example, a species of eel will migrate over 6,000 km from Europe to the Sargasso Sea (Andersson *et al.*, 2012). Mass migrations are known to travel through oceans, seas, and river systems and are usually seasonal or during a particular life stage (Opzeeland and Slabbekoorn, 2012). Much like larval settlement, ambient acoustic cues are used for orientation and coordination during migrations and to adjust to changes or barriers on route. Noisy conditions can cause excess stress in already vulnerable animals. When the eel makes its long migration it undergoes an irreversible physiological transformation in which the eyes and pectoral fins are enlarged, the skin colour changes, and the digestive organs are regressed (Andersson, 2011). The eels stop feeding during the migration phase so they have a limited amount of stored energy at their disposal. If this energy runs out because the length of the migration has increased owing to the avoidance of noisy areas, the eel may starve and never make it to the spawning ground to reproduce. Even if the eels do make their destination, spawning ability could be lowered if the individual has undergone injury, stress or starvation on route. Xie *et al.* (2008 - vessel; dB not stated; Hz not stated) recorded lateral and non-uniform movements when encountering vessels during migrations but at a distance of 7 m from the vessel behaviours and the migration returned to normal.

During migration individuals may need to communicate with others in order to synchronise and coordinate movements, and masking could prevent conspecifics from being heard (Opzeeland and Slabbekoorn, 2012). An example of coordinated herd movement can be seen in the Harp seal who produce trains of low frequency 'thumping' sounds during group migration (Serrano and Miller, 2000). The effect of noise on migration will depend on the level of behavioural change the noise creates; there may be no impact on individuals or the population, or there could be considerable changes affecting the population as a whole (Popper and Hastings, 2009a).

1.4.2.8 Schooling

Schooling can be defined as the use of simple interaction rules, including speed and direction changes, to coordinate movements with near neighbours (Katz *et al.*, 2011; Mann *et al.*, 2013). It is an advantageous behaviour for general survival. Individuals of the herring species, for example, cannot survive alone and anti-predatory benefits such as selfish-herd effects can reduce risk of predation (Hamilton, 1971; Ioannou, 2017). During migrations, schooling has been identified in Bluefin tuna (*Thunnus thynnus*) as a strategy to enhance the accuracy of migration routes (Sarà *et al.*, 2007). As the information needed for successful schooling is gathered through perception using the lateral-line it has been hypothesised that schooling behaviour could be disrupted by noise pollution through masking, distraction or physiological impairment (Partridge and Pitcher, 1980; Faucher *et al.*, 2010). Research on European sea bass (*Dicentrarchus labrax*) has suggested that diminished ability of the lateral line to detect nearest neighbour movements could result in reduced directional and speed correlations between nearest neighbours (Herbert-read *et al.*, 2017 - pile driving; dB not stated; Hz not stated). Further to this, the ability of individuals to process sensory information through multiple sensory systems (such as olfactory systems) may be negatively affected as a consequence of stress and/or distraction. Any deviation from normal schooling behaviour due to noise pollution could impact on the success of a migration or limit survival. Disorientation could occur if fish are dispersing or vertically swimming away from the group. Sarà *et al.* (2007 – vessel; 100-135 dB re 1 μPa; 70-20,000 Hz) found that ferry engine noise elicited significantly different behavioural responses from Bluefin tuna than 'no noise' controls, and concluded that ferries may create a physiological alteration, causing confusion for tuna. Impacts from noise on schooling behaviour could have repercussions for population survival. Schooling can also be used to improve the efficiency of locating food (Pitcher and Parrish, 1993). Feeding rate can be negatively affected by noise; pecking rates of damselfish significantly dropped when exposed to vessel noise (Bracciali *et al.*, 2012 - vessel; dB not stated; Hz not stated).

1.4.2.9 Individual differences

Boat noise can have a negative effect on nesting behaviours. The freshwater Longear sunfish (*Lepomis megalotis*) exhibited altered nesting behaviour in the presence of vessel noise (Mueller, 1980 - vessel; dB unknown; Hz unknown), and playback of a passing boat negatively affected nest digging and defence against predators in Princess of Burundi (*Neolamprologus brichardi*) (Bruintjes and Radford, 2013 - vessel; 127 dB re 1 μPa; Hz not stated). This study also pointed out that the same noise recording can cause different behavioural changes depending on the social context of the fish, and individual differences such as gender; dominant males and females responded differently to the same noise playbacks. When fish were looking after eggs the reaction to noise was greater than when no eggs were present (Radford *et al.*, 2016a - pile-driving; 156 dB re 1 μPa^2·s; 0-1,000 Hz). Additionally, the poorer the condition of the fish the greater the impact of the vessel noise. The Red mouthed goby has also shown diminished defence capabilities when exposed to vessel noise, as well as decreased time spent caring for their nests, and increased time spent in their shelter at sound pressure levels of 142-162 dB re 1 μPa and frequencies ranging between 0-1,033 Hz (Picciulin *et al.*, 2010 - vessel; 162 dB re 1 μPa; 602, 1,033 Hz; Sebastianutto *et al.*, 2011 - vessel; 161 dB re 1 μPa; 0-800 Hz).

1.4.2.10 Uncertainty

Research results have been contradictory, with some studies concluding that noise can have an effect on fish behaviour (e.g. Doksæter *et al.*, 2012; Thomsen *et al.*, 2012; Neo *et al.*, 2014; Voellmy *et al.*, 2014a; Simpson *et al.*, 2015; Magnhagen *et al.*, 2017), and others concluding there are no effects (e.g. Ruggerone *et al.*, 2008; Dokseater, 2009; Andersson *et al.*, 2012; Jørgensen *et al.*, 2012). It is likely that the ways in which fish are affected depends on their species, individual differences, and the exact noises they are exposed to. The uncertainty surrounding which species will be affected and by which source means that many combinations will need to be researched. Currently there is no framework available for suggesting or choosing potential study species, and so Chapter 2 aims to fill this knowledge gap; a guiding framework for prioritising fish species based on a number of criteria is suggested with inbuilt flexibility to allow for any number of differing scenarios/situations. As well as investigating the impacts noise can have on fish, research is also needed to find ways of preventing or limiting their exposure to noise, and to mitigate its effects on fish populations. In Chapters 3, 4 and 5, models are built to address these issues in relation to vessel noise pollution.

1.5 The status of research and understanding

Analysing findings from a literature review is a useful method for identifying trends and gaps in knowledge of an area of research. The literature search conducted looked for terms such as 'anthropogenic', 'noise', 'sound', 'impact', 'underwater', 'marine', 'physiology', 'behaviour', 'effects', and 'audiogram'. Further evaluation was conducted on articles of original research; 121 original research articles focused on fish research were identified (full data in Table A1.1). Other aquatic animals were ignored as not relevant to the thesis. These 121 articles were then systematically characterised according to: whether the experiment was laboratory or field based;

length of the study; the noise stimulus; acoustic metric; the hearing apparatus of the species under study; whether the resulting impact was behavioural or physiological; and if significant impacts were observed. Other parameters noted included the species, sample size, distance of the fish from the source, duration of signal, intensity (dB) and frequency (Hz) of the noise (Table A1.1).

Anthropogenic noise pollution research on fish has increased over recent years as awareness and concern for impacts has grown (Figure 1.2). Although noise pollution research and the impacts on mammals has been studied since the 1950s (Scheville and Lawrence, 1953), research on fish started much later (Banner and Hyatt's 1973 study was the earliest original article returned from the literature search).

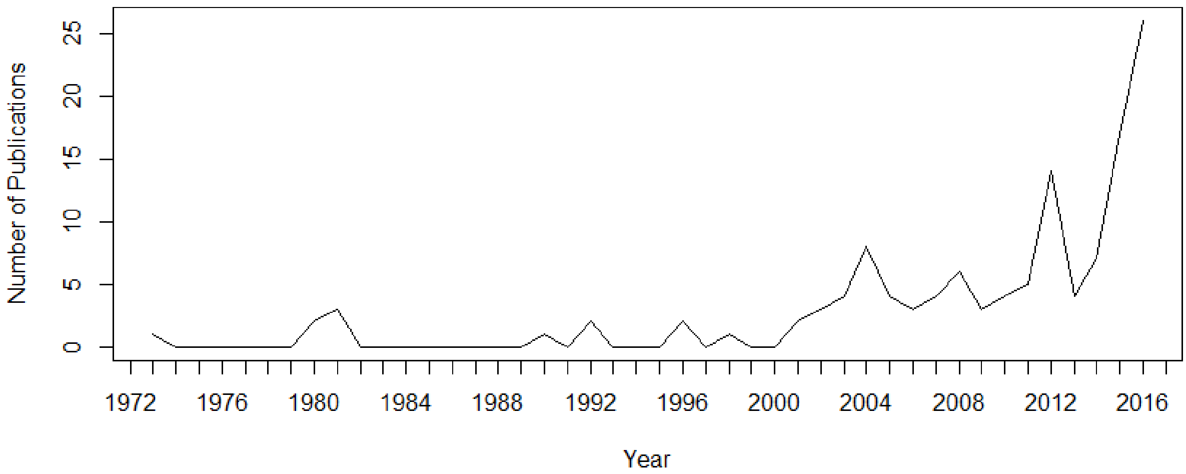


Figure 1.2. Number of original articles on the impacts of noise pollution on fish published per year between 1973 and 2017 found during the literature search, using the search terms listed in Section 1.6, paragraph 1.

Many experiments to date have reported results using different acoustic metrics, varying between sound pressure levels (SPL), sound exposure levels (SEL), received levels, source levels and in some cases no metrics at all. As metrics are not consistent across experiments it can be difficult to compare results. A full list of the studies found during the literature review, the decibel levels and frequencies used, and the metrics used to report the results can be found in Table A1.1. Of the 121 previous studies recorded in Table A1.1, only 78 % reported acoustic metrics and the metric varied between them (Figure 1.3). It is interesting that 22 % of studies mentioned no decibel levels at all (Figure 1.4), and 37 % of studies did not mention the frequency of the noise being used (Figure 1.4). As repeats of experiments are necessary to provide strong insights and to generalise results the metrics and measurement methods should be documented in a consistent manner (Braun, 2015).

One aspect of the noise source which was reported in every study was the source type. The chosen studies cover 11 different sources, with boat noise, pile-driving and white noise as the three most popular (Figure 1.5).

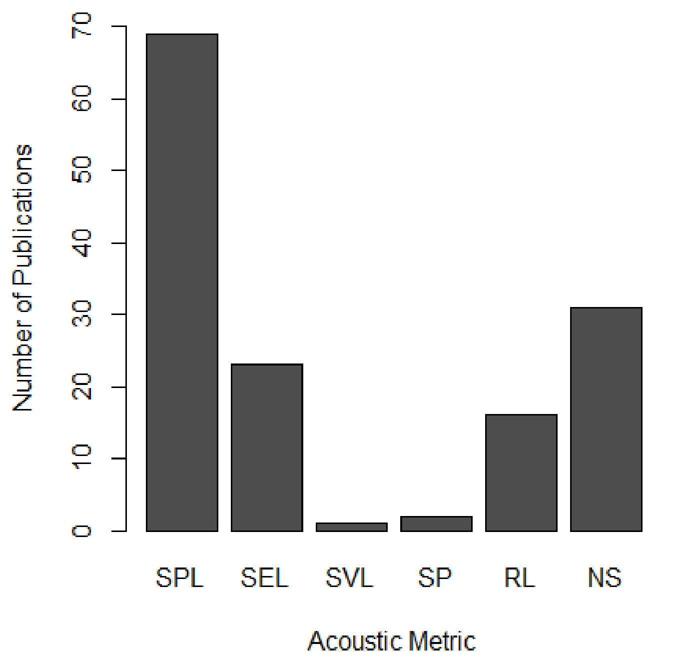


Figure 1.3. Number of publications analysed during the literature search reporting each acoustic metric, out of a total of 121 studies published between 1973 and 2017. Search terms listed in Section 1.6, paragraph 1. Acoustic metrics reported were Sound Pressure Level (SPL), Sound Exposure Level (SEL), Sound Velocity Level (SVL), Sound Pressure (SP), and Received Level (RL). Thirty-one studies did not report the acoustic metrics used (NS).

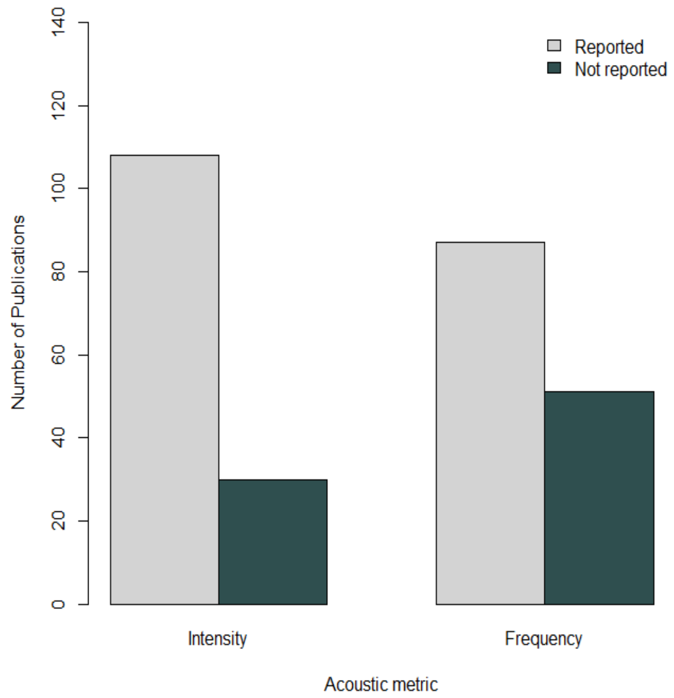


Figure 1.4. Number of publications analysed during the literature search that stated the intensity and frequency of the noise source under study, out of a total of 121 studies published between 1973 and 2017. Search terms listed in Section 1.6, paragraph 1.

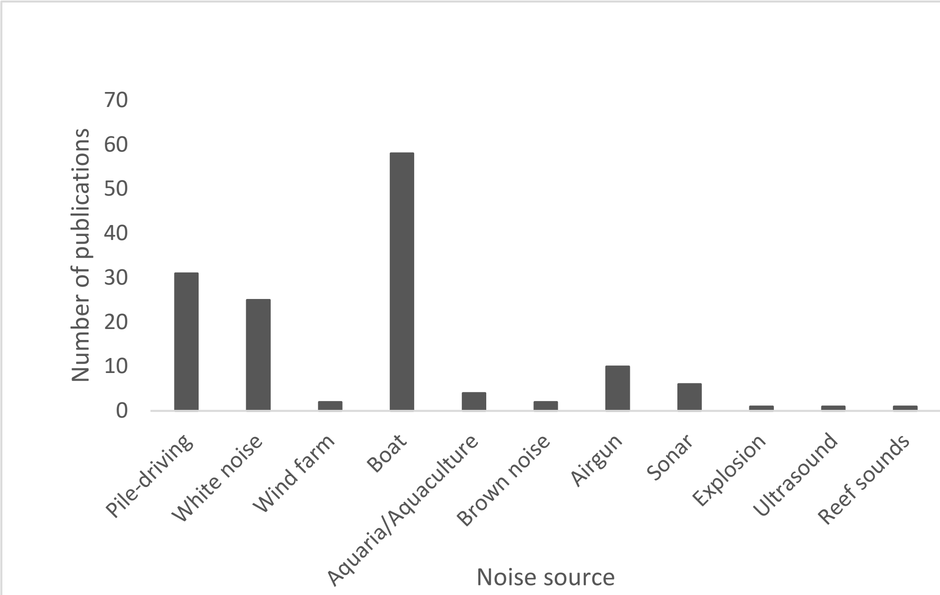


Figure 1.5. Number of publications examining each of the 11 noise sources.

Even though the body of research is growing, the number of species studied is still relatively few considering there are more than 32,000 species in the oceans (Popper *et al.*, 2014). The studies recorded here investigate noise impacts on only 60 fish species (Figure 1.6), of which 53 have specialised hearing apparatus (a swim bladder), and therefore considered more susceptible to noise impact.

Interestingly, proportionally the number of significant impacts reported is similar for those with (63.01 %) and without (62.50 %) a swim bladder. Fish with swim bladders are not impacted significantly more than those without a swim bladder (χ^2_1 = 0.001, p = 0.977, Figure 1.7). However, the sample size of fish without a swim bladder is small.

In general, more studies have reported evidence for significant impacts from various noise sources over non-significant impacts, which in itself is significant (χ^2_1 = 19.922, p = 0.00, Figure 1.8). A major challenge in understanding how anthropogenic-induced environmental changes affect organisms is establishing cause and effect relationships.

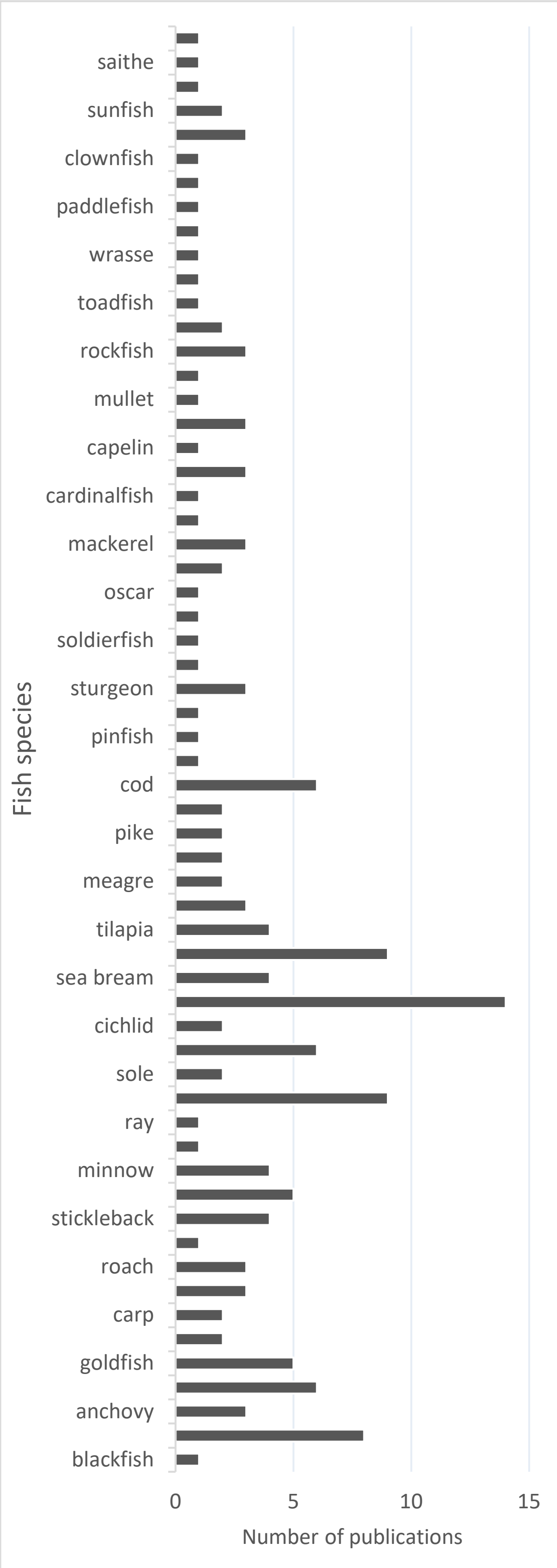


Figure 1.6. 60 species were investigated during the 121 studies dating from 1973 to 2017 that were analysed in the meta-analysis.

Whether a study analysed behavioural or physiological impacts was also recorded, showing that the two experimental designs are fairly evenly split, giving no indication that research focuses on one and lacks for the other (Figure 1.9). More behavioural studies showed significant effects than physiological studies (Figure 1.10). This may be due to physiological effects only resulting from higher intensity noise exposure – higher than those tested during the studies – and from long-term experiments of which there are few. As behavioural affects have been seen to occur from as little as 77 dB (Holles *et al.*, 2013) they will likely be observed in most studies. The experimental design of the study may also influence whether or not certain impacts are observed. For example, in laboratory experiments, fish are unable to undertake behaviours such as avoidance to prevent physiological damage making the risk of damage greater.

Laboratory studies are considered a successful way to analyse noise pollution impacts but the debate of their usefulness is still ongoing (Sabet *et al.*, 2015; Simpson *et al.*, 2015; Purser *et al.*, 2016; Rogers *et al.*, 2016; Slabbekoorn, 2016). On first look at the literature search data, there is not a large difference in the number of laboratory studies compared to those in the field (Figure 1.11). However, when the field studies are broken down into wild and controlled field studies there is a much more obvious difference between the number of laboratory and field studies (Figure 1.12). Re-categorising the studies into controlled and wild experiment designs, rather that laboratory or field, shows that studies on free-ranging fish in their habitats are limited (Figure 1.13).

Noise exposure in tanks is not necessarily representative of exposure in a fishes' natural environment (Rogers *et al.*, 2016). Besides the acoustic field of a small tank being considerably different from the acoustic field that occurs in the animal's natural environment, the acoustic wavelength being emitted from the speaker is often much larger than the tank dimensions. The speed of sound in water is around 1,500 m/s, meaning the acoustic wavelength ranges from ~1.5 to 30 m; tanks of such magnitude are rare (Rogers *et al.*, 2016). Owing to tanks being smaller in size than the wavelength produced from the speaker, even a short-duration sound could result in reverberation – defined as the persistence of sound in an enclosed space as a result of multiple reflections after sound generation has stopped (Yost, 1994) – making it difficult to separate the original sound from reflected sound. Particle motion levels are much higher relative to sound pressure in small tanks compared with open field conditions (Slabbekoorn, 2016). To prevent such issues associated with small tanks arising, studies should be conducted in tanks larger than the acoustic wavelength (which could be larger than 30 m) so that reverberation is limited, or the natural environment should be used. Small tank studies cannot be used to compare between species impacts or provide accurate data on hearing thresholds (Rogers *et al.*, 2016). Wild field experiments are needed to provide more accurate information on impacts and hearing thresholds. Noise exposure experiments in the field are difficult to conduct, and so most researchers attempt laboratory experiments and generalise the results to real-life environments. As a middle-ground, open water experiments using caged fish have become more common (Popper *et al.*, 2005, 2016; Houghton *et al.*, 2010; Debusschere *et al.*, 2014). Although closer to the real conditions of the ocean (acoustically speaking) such experiments still do not give a true indicator of wild behaviour, and may show physiological impacts when, if not caged, the fish could escape the area and not be impacted physiologically at all.

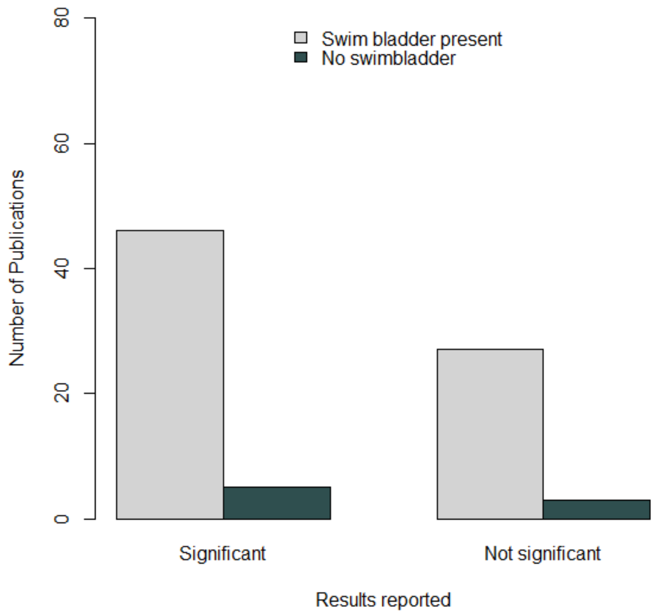


Figure 1.7. Whether significant impacts were reported for species with or without a swim bladder.

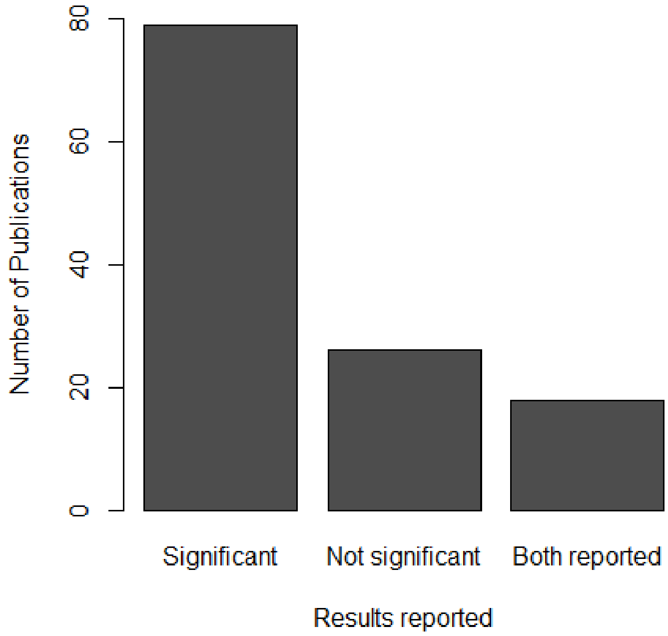


Figure 1.8. Whether significant impacts were reported for species or not. Some studies looked at more than one impact so may have reported some significant impacts and some non-significant impacts.

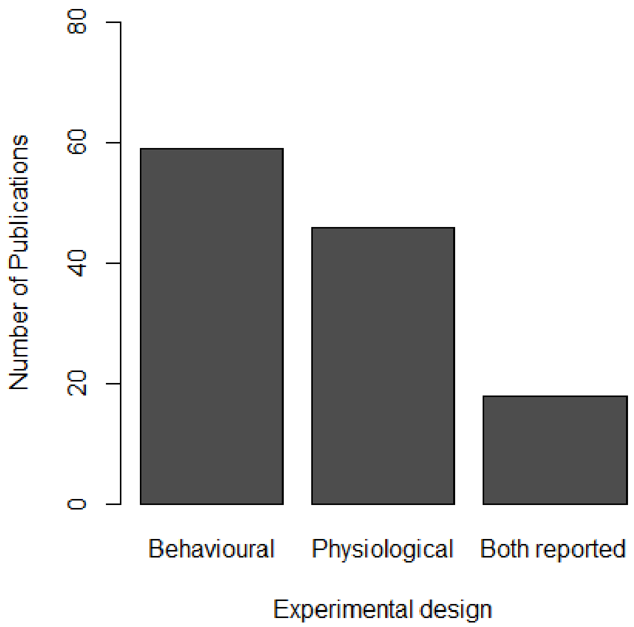


Figure 1.9. The number of studies investigating noise impacts on behaviour and physiology. More recent studies look at the impacts of both.

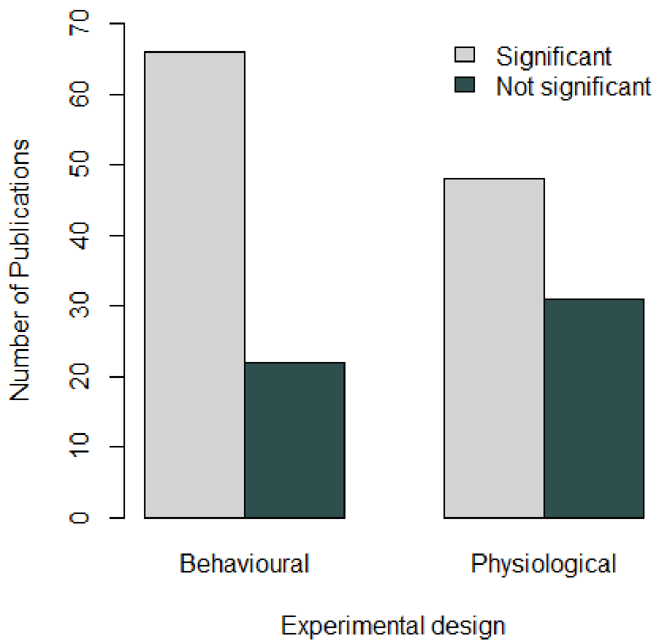


Figure 1.10. The results reported for behavioural and physiological studies.

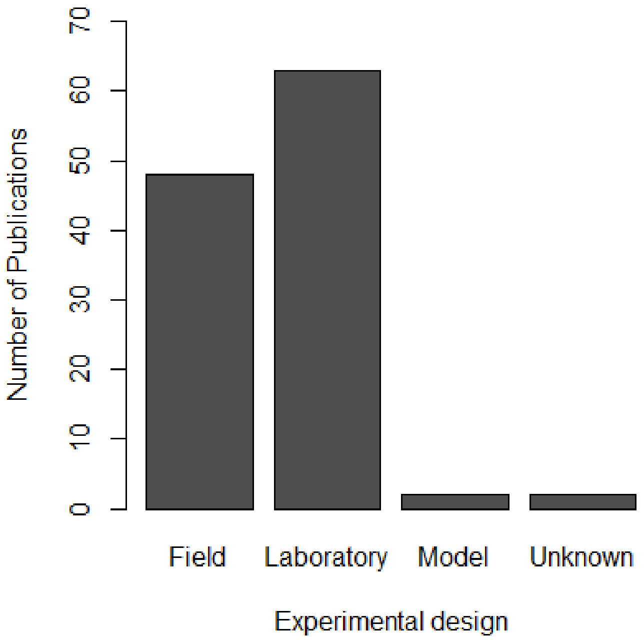


Figure 1.11. The numbers of field and laboratory studies. Two studies predicted impacts based on models and two studies remain unknown as original papers could not be found and data was taken from a more recent review.

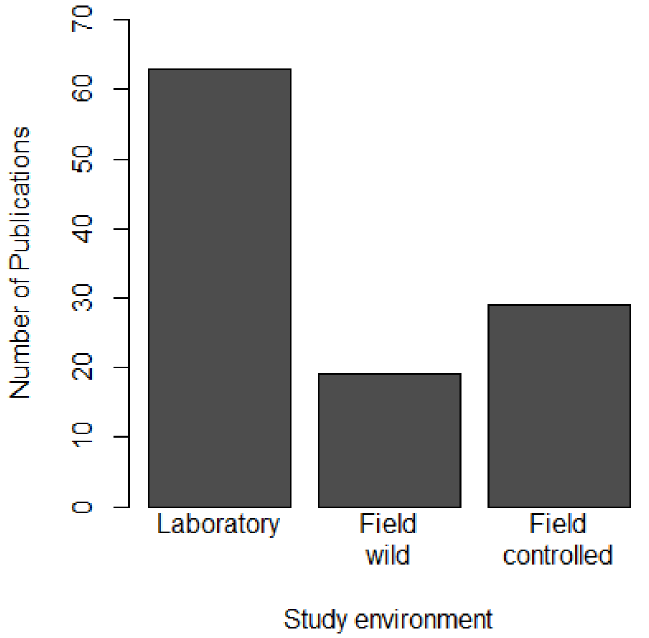


Figure 1.12. The numbers of laboratory and field studies, with field studies categorised into wild or controlled. Controlled studies are those that cage or enclose the fish or limit the movement of fish to one reef patch.

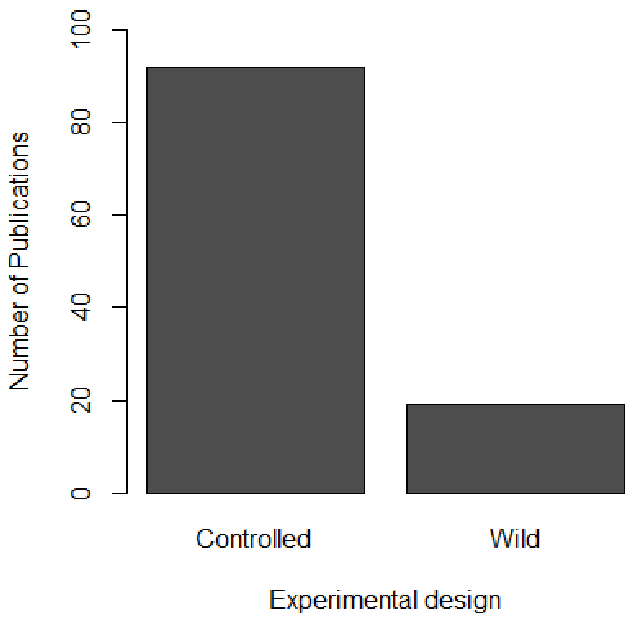


Figure 1.13. The number of controlled and wild studies. Wild is defined as free-ranging fish that can escape the study area. Controlled studies confine the movement of the focal fish.

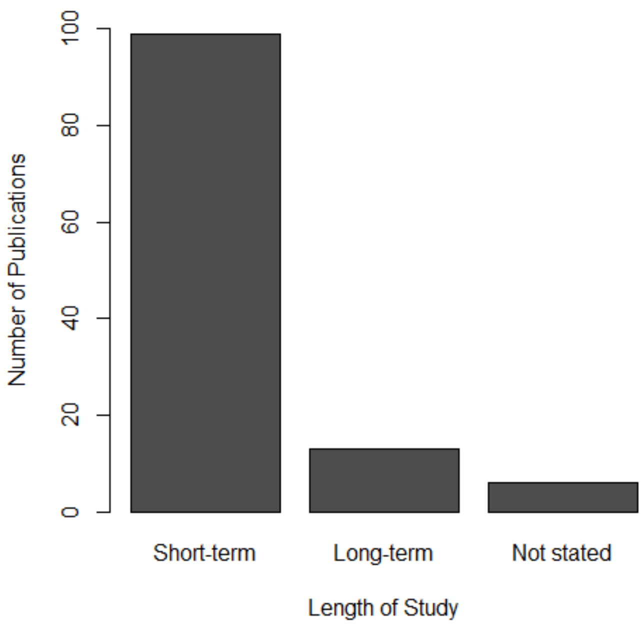


Figure 1.14. The number of publications that conducted long- or short-term studies. Two publications could not be categorised as data was taken from a more recent review article.

Another limitation of laboratory studies is that they are usually short-term. There is very little known about the long-term effects of noise (Kunc *et al.*, 2016); to understand the long-term effects of noise pollution, repeated or long-term exposure of the same individuals to noise is necessary. It would be difficult to look at the long-term impacts on an individual in the field without the use of expensive tagging equipment, but observations on populations would be feasible. Long-term studies could be easily achieved and repeated in laboratory settings and would inform on which species, if any, can habituate to noise over time or whether damage increases over time.

Of the 121 studies in this review, it was found that significantly more short-term impact studies were conducted than studies investigating long-term, or irreversible (e.g. PTS), impacts (χ^2_1 = 0.014, $P < 0.001$, Figure 1.14). This may be due to long-term research being costlier and more time intensive than short-term studies. Of the research found during the in-depth literature search only 2 % of studies investigated long-term effects of marine noise pollution, with the longest study being 9 months (Wysocki *et al.*, 2007 - aquaculture noise; 150 dB re 1 μPa; 2-20,000 Hz). This is a surprisingly low number of studies and emphasises the need for more long-term investigations to be carried out. However, as long-term investigations can be resource intensive, and field experiments difficult to manage in the natural environment, there is potential for noise impact modelling to help provide answers.

1.6 Mitigation and management

Legislation has been put in place in an attempt to minimise the impact on the natural environment and to offer some level of protection for marine systems that provide important services to humanity (Spromberg and Birge, 2005). The Heard Island Feasibility Test (HIFT) and the Acoustic Thermometry of Ocean Climate (ATOC) projects generated interest and concern about the effects of anthropogenic noise on marine mammals (Richardson *et al.*, 1995). Legislation regarding noise pollution, however, is still limited in the UK. The HIFT and ATOC experiments, designed to investigate climate change and ocean temperatures, involved sending noises through the ocean. Some individuals viewed the experiments as harmful to marine mammals, but the accusations proved to be groundless and based on misinformation (Au *et al.*, 1997; Popper *et al.*, 2004). However, since they did engender concern, the experiments raised public awareness of underwater noise pollution.

The ocean is full of varying noise sources which propagate over large distances and are trans-boundary in terms of legislation. Cumulative exposure, therefore, can only be managed effectively through incorporation within a larger management framework (Petruny *et al.*, 2016).

The two European-wide policies written to date relating to marine ecosystems are: the Water Framework Directive 2000/60/EC (WFD) which aims to achieve 'good ecological status' (GEcS) in transitional and coastal waters (1 nm out in England, Wales and Northern Ireland and 3 nm out in Scotland) by 2015, and the European Marine Strategy Framework Directive 2008/56/EC (MSFD) which requires EU Member States to achieve 'good environmental status' (GES) of their seas (all territorial waters) by 2020. GEcS focuses on the ecological structure of the environment, i.e. the abundance, presence, and cover of species at a given time (Heiskanen *et al.*, 2004; Borja *et al.*, 2013). GES considers the structure, function and processes of the ecosystem, taking their resilience and the impacts and activities carried out in the area of concern into account (European Commission, 2008). This means that there is a legislative framework to protect marine species and habitats, and to better manage impacts from anthropogenic sources. The MSFD contains 11 descriptors of GES covering both the status of species and habitats, as well as pressures and impacts. Descriptor 11 relates to noise pollution and consists of indicators addressing two main issues: the distribution in time and place of loud, low and mid frequency impulsive sound that is mainly introduced by offshore construction using pile driving (e.g., for offshore wind farms) and seismic surveys; and the trend of continuous low frequency sound indicated mainly by shipping activity (Breen *et al.*, 2012). The Directive required member states to provide an assessment of the current state of their seas by July 2012, to develop a set of indicators and targets for GES by July 2012 that would better define what is meant by 'good', then to establish how this might be measured and monitored by July 2014, and to establish a programme of management measures by 2016 that could be implemented to meet those targets, if not already being achieved. The MSFD has made significant progress in establishing protected areas in European seas but work is still ongoing for member states to meet the 2020 target of GES. The latest Commission Decision document states that GES Member States should apply the criteria, methodological standards, specifications and standardised methods for monitoring and assessment, and that assessment should be based on the best available science, but scientific and technical progress is still required (European Commission, 2017).

As wider policies and frameworks are limited to the two mentioned above, local efforts are also used to protect the underwater acoustic environment, for example, Glacier Bay National Park implements marine vessel quotas, speed regulations, and routing restrictions in biologically important areas (McKenna *et al.*, 2017). Whether localised attempts to prevent or lessen noise impacts are successful has not been studied. Owing to the large propagation distances of low frequency sounds and the difficulties of enforcing laws in the ocean it is hard to imagine that localised policy can have a major impact on mitigation.

Another localised method of working towards GES is the use of Impact Assessments. In the United Kingdom it is a legal requirement that new developments in the sea are supported by an Environmental Impact Assessment (EIA) (Harvey and Clarke, 2012). The strategic planning, assessment, and licensing of a marine development will contribute towards the achievement and maintenance of GES by avoiding unintentional and irrevocable consequences of any proposed new schemes and minimising potential adverse effects for the environment (Defra, 2015). An EIA is a legislation/management tool which identifies and evaluates the potential impacts (effects) of a proposed development and makes sure that mitigation methods are properly understood by the relevant authority before development decisions are made (Canter, 1977; Department for Communities and Local Government, 2000). Additionally, EIA regulations require that cumulative effects also be considered through a Cumulative Impact Assessment (CIA) (AMEC Environment & Infrastructure Limited, 2013). CIAs investigate the implications from the incremental impact of actions when added to other past, present or reasonable future actions. Cumulative impacts can result from individually minor actions that become collectively significant over a period of time (Eccleston, 2011). The CIA process analyses the potential impacts and risks of proposed developments in the context of the potential effects of other human activities and natural environmental and social external drivers. Some environmental and social attributes considered to be important in assessing risks in CIAs are: habitats, wildlife populations (e.g., biodiversity), and ecosystem services (Cardinale *et al.*, 2013). To date, not all noise sources and their effects at the level of individuals, populations, habitats, or ecosystems have been incorporated into management decisions (Andersson, 2011). However, assessing underwater energy release(s) (including noise) is now frequently a requirement of the EIA (where a risk is deemed to be present based on known noise source levels and sensitivity of species present). Through EIAs, developers may be obliged to monitor and mitigate the effects of the energy input from their activities to comply with government legislation. Whether an impact is deemed "significant", however, often relies on professional judgement owing to lack of quantitative tools (Western Australian Government, 2010). With limited research on noise pollution impacts on fish, especially fish populations, assessments are likely to include subjective or generalised results and recommendations.

Models are often used within the EIA process to predict noise levels at the development site and to map the predicted exposure to the supposed noise exposure thresholds for particular species to estimate whether certain physiological or behavioural responses are likely to occur (Farcas *et al.*, 2016). However, such models are often simplistic, with limited environmental data, and without field measurements to ground-truth model predictions; errors and uncertainties in noise modelling can lead to significant pitfalls in the EIA process (Farcas *et al.*, 2016). If noise exposure is overestimated then developments (or other noise producing activities) could be disallowed when in actual fact there may be no negative impact on the ecosystem. Conversely, underestimating may allow activities to occur when there is higher risk to fish than expected resulting in (potentially irreversible) damage to individuals, populations or even the ecosystem as a whole. For this reason, noise impact assessments need to be precautionary, taking into account any uncertainty and worst-case scenario. As a result, mitigation measures that are a condition of the licence are potentially more stringent than might be deemed necessary to take account of this uncertainty, particularly when protected species are likely to be affected. This is known as the precautionary principle (Scotland and Northern Ireland Forum for Environmental Research, 2006).

The implementation of the MSFD and awareness that the occurrence and use of potentially harmful noise sources are likely to increase in the coming years, raises the question of how to mitigate for any harmful effects; another important area of future study (Azzellino *et al.*, 2011). Especially on a larger scale. Many organisations are working towards creating quieter vessels to reduce noise pollution. Noise reduced vessels were designed for conducting surveys and specialised vessel designs such as diesel-electric propulsion, fixed pitch propellers and quiet hulls have substantially decreased noise levels over a wide frequency range (De Robertis and Handegard, 2013). Reduced vessel noise could provide evidence for whether fish are avoiding vessel noise, or just the presence of the vessel in general. In a comparison study, the noise reduced vessel caught more fish that the normal fishing vessel showing that less avoidance occurred in response to the quieter vessel (Engas, 1991).

In 2014, the International Maritime Organization (IMO) adopted guidelines for the reduction of underwater noise from commercial shipping (Papastavrou, 2014). The International Council for the Exploration of the Sea Cooperative Research Report No.209 summarises that there is overwhelming evidence for avoidance reactions to vessels when levels exceed hearing thresholds by 30 dB. It recommends that vessel 1/3 octave band underwater radiated noise should not exceed this level (Mitson, 1995), and has made recommendations for research and fishing vessels to stay within 30 dB of a species hearing threshold. A caveat to this, however, is that not all species hearing thresholds are currently known.

There is a need for scientific research on the impacts of noise on wildlife to translate into management actions or recommendations (Tabarelli and Gascon, 2005). To implement the correct policies and regulations a consensus is needed as results so far have often been contradictory. More research is needed to provide definitive evidence of oceanic noise pollution, and to develop assessment metrics that will be universally accepted as a way of measuring the impact of noise from new developments or shipping lanes on marine life. Guidance criteria covering noise exposure thresholds for impacts on marine fish are not available for many anthropogenic sources owing to significant gaps in knowledge (Popper *et al.*, 2014). Currently guidance criteria exist only for pile-driving noise emissions, and none are yet available on emissions from vessels. Future research on noise pollution and its impact on the marine environment will aid in the conservation of all fish species, and help provide a sound basis for future legislative decisions and noise criteria guidelines.

1.7 Ocean noise mapping

As a result of the increasing number of scientific and management efforts, there has been a developing focus on the much larger scale and longer term chronic effects of increases in ocean noise and changes in underwater soundscapes (Gedamke *et al.*, 2016a). Conducting studies at the population, community or ecosystem level are not impossible, but can be challenging (Boyd *et al.*, 2011a; Williams *et al.*, 2015b), and so a more preferred method may be to predict impacts through the use of models. Recent research has highlighted the importance of soundscape characterisation, modelling, and mapping (Boyd *et al.*, 2011b; Dekeling *et al.*, 2016). Models are often simplified representations of complex theories, analogies of systems, or summaries of data (Wainwright and Mulligan, 2004). Modelling allows environments and situations to be simulated using mathematical equations, to predict and analyse trends and behaviour (Yorkor, 2013). It is a powerful tool for developing and testing theories by way of causal explanation, prediction, and description (Shmueli, 2010). The model-based approach to research represents a more powerful way to evaluate noise levels (Rossi *et al.*, 2016).

Consequently, models and noise maps are seen as a way of obtaining more accurate results at lower costs (Dekeling *et al.*, 2016). It is believed that the use of models, combining data from measurements and noise sources with predictions of propagation loss, will contribute directly to effective ambient noise monitoring and noise assessment. Dekeling *et al.* (2016) have suggested several uses for noise models in the future:

(1) The creation of noise maps that facilitate more cost-effective trend estimation;

(2) The identification of trends for different source types through identifying the cause of any fluctuations, and consequently facilitating mitigation action;

(3) The removal of selected sources if they do not cause a departure from good environmental status (e.g., rain and lightning); and

(4) The presentation of a better overview of the actual levels and distribution of levels across their sea area, allowing identification of departures from good environmental status.

The authors state that the outputs could contribute to a greater understanding of the likely impacts of noise in the future by predicting possible effects of future changes, and allow predictions about the efficacy of alternative mitigation actions (Dekeling *et al.*, 2016).

A recent review article made a call for "more scientific programmes that monitor trends in soundscapes through the acquisition of long-term data sets with immediate emphasis in areas of future change and/or critical habitat" and the "development of models predicting the degree of masking of particular sounds by different man-made sounds under varying conditions in the sea" (Hawkins *et al.*, 2015). The applications of models are vast, from development licencing, prioritising research, and predicting outcomes, to planning mitigation.

Monitoring and modelling ocean noise can be used as a method for determining ocean noise level baselines; it is not possible to establish policies for noise pollution without a baseline, thus the continued examination of soundscapes is critical to conservation and management efforts (Haver *et al.*, 2017). Noise modelling for EIAs can be used to predict noise exposure of a particular activity by modelling the received noise level at a given point, allowing inferences to be drawn about whether nearby species will be harmed or not. Archival or real-time recordings can be used to monitor soundscapes over temporal and spatial scales, to aid understanding of noise patterns and trends and the implications changes in noise levels can have (Hatch *et al.*, 2016; Haver *et al.*, 2017) Predictive modelling for mitigation can identify the most efficient way to reduce noise impacts in an area without impacting business-as-usual. For example, models based on planned increases in vessel movements in the Moray Firth may be able to forecast associated increases in noise exposure (Lusseau *et al.*, 2011; New *et al.*, 2013). There is need for a mechanistic, integrative approach to make predictions about scenarios that are (logistically) impossible or difficult to obtain in the field (Graham and Cooke, 2008; Popper and Hastings, 2009b; Williams *et al.*, 2015b; Slabbekoorn, 2016).

For models, as with laboratory studies, there is often a species of interest at the heart of the research. With over 32,000 species, all with differing hearing thresholds and potential impacts it is difficult to know where research should be focused. Modelling can aid in the prioritisation of research. Currently, there are no prioritisation models specifically associated with noise pollution impacts on fish. Risk and vulnerability models can help identify species likely to be threatened by noise pollution (*see* Chapter 2). Modelling of vessel traffic and vessel source levels can help identify areas where fish populations will face increased vessel noise exposure (*see* Chapter 3). Predicting vessel noise exposure levels from modelling alterative vessel traffic scenarios will help identify and prioritise beneficial mitigation measures (*see* Chapter 4). Modelling, combined with field research and biological information, will help in the identification of noise-sensitive species and pinpoint the locations where they are prone to suffer most from noise exposure. Mitigation measures, for example, temporal or geographic area closures, can then be designed for activities that generate high levels of noise (Simmonds *et al.*, 2014) (*see* Chapter 4 and Chapter 5).

Predicting the efficiency of mitigation measures using models is beneficial as strategies may cost money and effort to implement. For example, technological innovations (such as quieter ship propellers) would require manufacturers to alter designs and materials and adjust for consequent costs; modifications to standard operations (such as slower ship speeds or creating restricted areas) could impact economic efficiency and would need added effort to enforce. Such large-scale mitigation efforts must be founded on strong evidence. Models and simulations provide a cost-effective method of obtaining such evidence (Chapter 4). In addition, the benefits of mitigation on fish are not fully understood (Shannon *et al.*, 2015); it is harder to predict long term outcomes of mitigation without the use of models.

1.8 Literature review synopsis

There is evidence to support that a wide variety of known effects occur in fish as a result of anthropogenic noise pollution (Table 1.4), and more impacts may yet be recorded as research continues to grow and new techniques are pioneered and developed. If noise pollution is allowed to continue unabated the overall effects on fish populations could have serious repercussions both for fish and mankind. Planned mitigation measures, successfully implemented, would help protect endangered fish species from further harm, and alleviate the considerable social and economic implications that any significant reduction in fish populations could have. Hawkins *et* *al.* (2015) stated that there are two kinds of mitigation, "The first involves the use of biological information to minimize effects. The second involves changes to the sound source to minimize effects". Ideally, both types of mitigation, biological and changing the sound source, should be implemented to minimize noise impacts on marine fish species.

The review of the literature recognises the areas that have gained research attention, identifies those lacking it, and highlights the concerns expressed surrounding marine noise pollution and its impacts on marine fish. There is limited knowledge of exactly how noise impacts will manifest long-term but clearly there is a need for more research on the long-term impacts of noise on fish rather than simply extrapolating results of short-term studies to predict long-term consequence. Relatively little research has considered the effects of extended exposure to chronic noise; extended exposures may result in habituation or sensitisation, and thus changes in response (Nedelec *et al.*, 2016b). The effects of continuous noise on fish behaviour such as that produced by vessels, is less intense but considered both a constant drone and a random intermittent noise, and is known to elicit behavioural responses in several species (e.g. Amoser *et al.*, 2004; Bracciali *et al.*, 2012; Bruintjes and Radford, 2014; Bruintjes *et al.*, 2016; Celi *et al.*, 2016; Luczkovich *et al.*, 2016). Many researchers claim that ocean noise will continue to increase but, actually, the trend of ocean noise is still unclear. If ocean noise does continue to increase, species will need a propensity to adapt quickly, or human intervention will be necessary in terms of mitigation to prevent population decline or possible extinction of species. Vessels are considered by some as the greatest threat to marine species as vessel noise is the most widespread anthropogenic noise source in the oceans. Nedelec et al (2017a) claimed that to their knowledge, all studies but one to date has focused on the responses of individual species, which suggests population level effects have been largely ignored thus far. Studies have shown that non-injurious effects can still accumulate to the extent of population-level impacts mediated through detrimental physiological changes and behavioural alterations, so large-scale population studies need to gain research attention. Whether vessel noise causes long term avoidance strategies in fish is currently unknown but there could be economic consequences if it is shown to effect commercial species within breeding or fishing grounds. The benefits of long-term studies conducted over broad spatial scales are that they may offer a more complete understanding of the population-level and interacting effects of noise on wildlife (Shannon *et al.*, 2015). This is especially true for fish species.

The effects of vessel noise are not only an issue for the fish, but also for the researchers trying to assimilate meaningful data. Behavioural disturbances caused by survey vessels themselves can introduce a substantial bias in abundance estimates of commercially important stocks (Hjellvik *et al.*, 2008). Field research on the impacts of vessel noise has proved hard to conduct owing to the lack of control over the environment and limited direct observation of the fish. There is also doubt around whether fish are reacting to visual stimuli of the vessel or the noise itself (Gerlotto *et al.*, 2004).

Contradictory results of noise pollution impact research show just how unique each species is in its reaction to noise, both behaviourally and physiologically. Researching every combination of species, noise and context is near impossible, but filling in key taxonomic gaps will allow predictions to be made through comparative studies or mechanistic models (Williams *et al.*, 2015b). A discussion panel at the 3rd International Conference for the Effects of Noise on Aquatic Life produced a summary of key questions which included identifying: priority species and groups, such as fishery species and priority life stages (e.g. larval, Smolt and juvenile); priority habitats, especially estuarine, inlet and spawning areas, deep sea areas and Marine Protected Areas (Lewandowski *et al.*, 2016). The prioritisation of fish species and habitats will allow key groups to be identified and research focused on the areas where it is most needed.

Owing to the disadvantages of long-term field studies discussed in this chapter, modelling is a more appropriate method to guide and justify mitigation. Long-term, and large geographic studies can be conducted at relatively little cost, without the need to change legislation when there is still uncertainty of success. The predictive capabilities of models have been underutilised in noise pollution research thus far. With plans for more energy farms to be constructed offshore, increased use of motorways of the sea, the alteration of shipping lanes, and improved vessels becoming larger and faster, future noise levels are not easy to predict. A reliable method of identifying possible future scenarios and predicting the potential impacts on, and possible mitigation for, marine life would be beneficial and help build confidence on future noise level estimates.

Vessel noise pollution is always present in the oceans and is the most widespread anthropogenic noise pollution source (Celi *et al.*, 2016). Vessel noise research has increased over the past two decades; the meta-analysis shows publications concerning vessel noise impacts on fish has more than doubled since 2011, compared to the number of studies published from 2000 to 2010. Vessel noise could pose a serious threat to both individual animals and entire populations (Weilgart, 2007b; Clark *et al.*, 2009; Slabbekoorn *et al.*, 2010). Predicting future trends of vessel noise pollution and identifying efficient mitigation measures could help to reduce noise pollution impacts on fish in future years.

Table 1.4. A summary of known effects of noise on fish showing potential impacts of various noise sources.

Effect	Impacts observed	Evidence reference	Noise source	dB levels (metric)
Temporary Auditory Threshold Shift	• Severe damage to sensory hair cells of the saccule of the ear • Damage increased for at least 58 days post exposure • Auditory thresholds did return to normal after 18 or 24 hrs depending on species	Hastings *et al.* (1996); Amoser & Ladich (2003); McCauley *et al.* (2003); Cott *et al.* (2012)	Air gun, white noise	180 (SPL) 158 (SPL) 222.6 (SL) 205 (SPL)
Permanent Auditory Threshold Shift	• Elevated auditory threshold • Longer noise exposure had more impact • No recovery seen to those exposed for 24 hours	Scholik & Yan (2001)	white noise	142 (SPL)
Damage to non-auditory body tissue	• Bruised organs • Haemorrhaging of tissues	Casper *et al.* (2012)	Pile driving	210 (SEL)
Embolism	• Mild injuries and substantial physiological costs e.g. barotrauma, embolism • Substantial injuries could not be recovered from	Halvorsen *et al.* (2011)	Pile driving	215 (SEL)
Stress-related responses (cortisol secretion, heart rate, breathing rate etc.)	• Elevated plasma cortisol levels • Increased cardiac output, increased heart rate and decreased stroke volume	Smith *et al.* (2004); Wysocki *et al.* (2006); Graham & Cooke (2008); Spiga *et al.* (2012a)	White noise, boat	160 (SPL) 153 (SPL) Not stated 180 (SPL)
Increased vulnerability to disease	• Increased stress due to noise can lead to increased vulnerability to disease	Anderson *et al.* (2011)	Aquarium noise	123.3 (SPL)
Decrease in reproductive rate	• Egg viability significantly reduced • Growth rates of fry decrease in louder exposure	Banner & Hyatt (1973)	white noise	Not stated
Interference with sensory vision	• Allowed a simulated predator to approach closer before they hid • Masked an approaching predator's sound • Reallocated attention and distraction • Distraction made worse when noise and lights used together • Increased vulnerability to predation	Chan *et al.* (2010)	Boat	98.1 (SPL)
Masking of communication	• Vocalisations less common when exposed to noise • Altered auditory sensitivity due to masking	Ramcharitar & Popper (2004); Luczkovich *et al.* (2012)	Boat, white noise	124 (SL) Not stated
Change in foraging efficiency	• Both decreased and increased foraging observed depending on noise source • Increased foraging errors • Increased food vs non-food errors • Negative impact on foraging activity	Bracciali *et al.* (2012); Purser & Radford (2011)	Boat, white noise	Not stated 50-150 (SPL)
Reduced habitat availability	• Ability to use ambient sounds to aid settlement on reefs decreased • Larvae settled later than normal	Simpson et al (2016)	Ambient reef sounds	136.1 (RL)
Increased energy needs	• Protein to energy ratio decreased • Higher feed conversion ratio values	Spiga *et al.* (2012b)	Boat	175 (SPL)
Increased disturbance	• Change in swimming behaviour • Alarm behaviour exhibited • Startle responses and 'C' starts	Pearson *et al.* (1992); Wardle *et al.* (2001); Amoser *et al.* (2004)	Air gun, boat	161 (SPL) 206 (SPL) 103 (SPL)

	• Vertical movement and freezing • Involuntary sideways movements			
Adverse impacts on nursing	• Decrease in time spent caring for nests	Picciulin *et al.* (2010)	Boat	162.4 (SPL)
Disorientation and decreased schooling	• Increased vertical movements • Dispersion of school • Restless, speed changes and turning behaviour • Lateral movements away from source • Scattering and non-uniform swimming	Sara *et al.* (2007); Xie *et al.* (2008)	Boat	100-135 (SL) Not stated
Displacement or avoidance of area	• Catches in loud areas decreased, with catches just outside the loud area increasing • Strong, downward avoidance reactions • Fish avoidance occurs between the surface and 200-m depth	Engas *et al.* (1996); Slotte *et al.* (2004); Suzuki *et al.* (1980); Mitson & Knudsen (2003); Ona & Godo (1990)	Air gun, boat,	248.7 (SPL) Not stated 130 (SPL) 165 (SL) Not stated
Ineffective territorial behaviours	• Decreased nest digging • Ability to defend home successfully decreased	Sebatianutto *et al.* (2011); Bruintjes & Radford (2013)	Boat	161 (SEL) 127 (SPL)
Compromised viability of individuals	• Influenced fat stores, growth and several reproductive indices • Increased cardiac output	Meier & Horseman (1977); Graham & Cooke (2008)	Broadband, boat	140 (Not stated) Not stated

1.9 Thesis aims and objectives

Although previously limited, research on the impacts of underwater noise on marine life is steadily increasing. Key questions raised about the impacts of noise pollution on fish include: which species and habitats should be prioritised first; where should research, mitigation efforts and legislation be focused; how can researchers overcome challenges in noise mapping and apply the models to direct mitigation (Lewandowski *et al.*, 2016).

The overall aim of this thesis is to address the key questions raised above through creative modelling and then demonstrating their potential for use in future noise impact research and mitigation. The models created and methods used are either completely novel, or add innovation to previously reported work, to develop, expand and improve their effectiveness as modelling tools.

1.9.1 Chapter 1: Literature review

Aim: to undertake a comprehensive literature review to identify gaps in knowledge and pinpoint areas for targeted research.

Objectives:

1a: collect and collate all relevant literature pertaining to the impacts of anthropogenic noise emissions on fish.

1b: conduct a meta-analysis on all previous studies collected with focus on significance of impacts, types of noise source, frequency, duration, and intensity of the source to identify gaps in knowledge and look for potential areas of further research to reduce marine fish species risk and vulnerability to noise impacts.

1c: create and define the aims and objectives for thesis research, in line with the needs identified by industry.

1.9.2 Chapter 2: Establishing priorities for noise pollution impact research on marine fish: a value- and susceptibility-based framework

Aim: to create a method for prioritising fish species vulnerable to noise pollution impacts to guide research and mitigation.

Objectives:

2a: identify indicators for assessing a species vulnerability to threats from noise.

2b: assess likelihood of exposure to noise and severity of harm should exposure occur through the collection of available audiogram data and previous research results on impact thresholds.

2c: create a framework using a risk-based approach for prioritising species of fish that could potentially be vulnerable to noise impacts.

2d: discuss the applications of the method for use in mitigation, research and as an online tool.

1.9.3 Chapter 3: Creation of an ocean noise map: using AIS data to model shipping noise emissions

Aim: to create a method for modelling vessel source level noise with a visual output (mapping) of vessel emissions.

Objectives:

3a: build a robust model that estimates source levels of vessels using AIS data and calculates the noise exposure level resulting from the vessel source levels.

3b: create a user-friendly visual output of the noise exposure levels in UK waters.

3c: assess the noise exposure levels output from the model in terms of a case study using historical AIS data as the basis for predicting future trends.

1.9.4 Chapter 4: Using AIS data modelling of noise emissions as a predictive tool

Aim: to show the applications of an AIS-based noise exposure model for predicting the outcomes of mitigation scenarios.

Objectives:

4a: identify five potential future scenarios to test the AIS-based model.

4b: assess each scenario using predictive modelling and determine whether the suggested measure can be practically applied and recommended as a suitable mitigation measure.

4c: discuss the usefulness of the predictive model method for use in mitigation.

1.9.5 Chapter 5: Combining prioritisation, historical and predictive modelling as a method for mitigation

Aim: to combine prioritisation, biological and source level modelling to predict impacts on species migration.

Objectives:

5a: assess the potential for two species to be impacted by anthropogenic noise during migrations.

5b: analyse the species migration data alongside the model created in Chapter 3 to identify high risk areas in need of mitigation and present the data visually (through mapping).

5c: discuss the usefulness of combinations of models in mitigation.

1.9.6 Chapter 6: Thesis discussion

Aim: to discuss the relevance and part played by each method and model designed and explained in previous chapters and its potential role in noise mitigation

Objectives:

6a: discuss the usefulness of each type of mitigation.

6b: discuss the applications and results of the models created during the previous chapters.

6c: discuss whether each model provides beneficial applications for research and decision making in regard to the mitigation of noise on marine fish species and populations.

2. Chapter Two: Establishing priorities for noise pollution impact research on marine fish: a value- and susceptibility-based framework

This chapter focuses on acquiring the biological information needed for efficient mitigation and prioritisation. Fish species differ greatly in anatomy and behaviour, and reactions to noise exposure can vary both within and between species due to individual differences, specific social context, and/or geographic location (Ladich, 1999; Parmentier *et al.*, 2005; Popper and Hastings, 2009b; Bruintjes and Radford, 2013; De Robertis and Handegard, 2013). This means that no single standard model of response to noise would prove generic for all fish species. Species specific research on the impacts of noise would have to be considered, but extensive species-appropriate research would likely be unrealistic owing to time and budget constraints. Currently, no structure for prioritising noise impact research on fish exists. The aim of this chapter is to devise a method of identifying and prioritising those species most in need of further research and mitigation by assessing their value to the ecosystem and likely level of susceptibility to noise pollution.

A framework based on key considerations pertinent to the assessment of risks regarding fish and noise pollution in the marine environment was created. It assigns scores to quantify the overall value of a species to the ecosystem, and reflect the likelihood of harm from noise pollution that the species may encounter. The two primary indicators: value and susceptibility, and their contributing factors are then combined to produce a risk-based tool for directing future noise impact research, by identifying priority species, and suitable areas of focus for mitigation. By utilising multiple indicators, the framework helps identify vulnerable species and habitats, and provides added detail on where mitigation may be most beneficial for individual species. This new approach to ranking of species provides a useful low cost risk-based tool for decision-makers to make more informed decisions through greater understanding of noise pollution impacts.

2.1 Introduction

2.1.1 Priorities

Anthropogenic threats need to be managed effectively to prevent further decline of marine species. Research on noise in the marine ecosystem has been recognised as a means of improving environmental management (Popper and Fewtrell, 2003; Popper and Hastings, 2009a, 2009b;

Slabbekoorn *et al.*, 2010). However, more research is still needed to provide improved knowledge of marine noise pollution impacts, and to develop assessment metrics capable of more accurately measuring the impact of anthropogenic noise on marine life. To help achieve and maintain sustainable fish populations, targeted research on the impacts of marine noise on fish species needs to be undertaken.

Marine and estuarine ecosystems are subject to anthropogenic noise from a number of sources, such as shipping, the use of sonar, the construction of port infrastructure, renewable energy devices, prospecting for oil and gas, and aggregate extraction. Particularly in the northern hemisphere, anthropogenic noise in the marine environment can be loud, frequent and persistent (McDonald *et al.*, 2008; Andrew *et al.*, 2011). The impacts of noise on fish is of growing concern, as evidenced by the increasing amount of research and public awareness on the topic (Figure 1.2). This extraneous noise in the oceans has the potential to affect individual organisms, populations, food chains and whole ecosystems.

It has been suggested that research addressing the actual status and future trend of underwater noise pollution should be conducted in high priority areas (i.e. areas with biologically important communities) in order to inform plans for mitigation (Codarin and Picciulin, 2015). However, it is difficult to predict the likely impact that a noise source will have on any one fish species. Almost a third of the 8,479 fish species listed on the IUCN Red List are designated as "threatened" or "endangered" (International Union for Conservation of Nature, 2014), and yet relatively little is known about the vulnerability of fish to noise pollution. At least 58 fish species have been studied in terms of noise pollution impacts to date, covering 11 different noise sources (from an analysis of 121 articles), but this covers a minimal number of the 32,000 known fish species. Most studies do not give reasons for the choice of study species (only 3 of the 122 studies analysed in Chapter 1 provided a reason). Kastelein *et al.* (2017) selected European sea bass based on its economic importance, and ease of maintenance and availability. Casper *et al.* (2012) chose to study Atlantic salmon owing to their threatened/endangered status on the US West Coast. Hastings *et al.* (1996) studied the physiological impacts of noise on Oscar as a representative of fish without specialised hearing, and suggested results may be extrapolated, with caution, for similar morphological species such as tuna and salmon. Extrapolation is unlikely to be accurate from one species to another, and the choice of Oscar over other more commercial species was not clearly stated by the authors. It is unknown whether the 58 species studied to date were the highest priority species to protect from noise pollution impacts, or whether they were chosen for other reasons, such as convenience (e.g. Tavolga, 1974; Kastelein *et al.*, 2017).

Impacts have been shown to vary both between and even within species; the same noise playback can cause different behavioural changes depending on the social context of the fish, and individual differences such as gender (Bruintjes and Radford, 2013). More research is needed to move towards a consensus for the impacts noise can have on various species variants. A way of prioritising or targeting certain species would provide a logical starting point for planning research (Landrø and Amundsen, 2015).

2.1.2 The need for a novel prioritisation method

Understanding the ways in which particular threats affect ecosystems can aid in prioritisation of the most important or most manageable threats (Halpern *et al.*, 2007). This same logic can be applied to species; understanding the impact of noise on particular populations would allow for the most affected to be protected as a first priority. Halpern *et al.* (2007) raised the questions of how to decide which threats, or ecosystems, should be focused on as a priority, and how to maximise the return on conservation investment. Mitigating multiple threats to a species is a daunting task, particularly when funding constraints limit the number of threats that can be addressed at any one time. Threats are typically assessed and prioritised via expert opinion workshops that often leave no written evidence of the rationale for decisions, making it difficult to update recommendations as new information comes to light. A quantitative, replicable, and evidence-based method is needed for determining the impact of any given threat on a particular ecosystem. This would ensure that information about the process is preserved and thus allow for evaluation and revision of resulting decisions as new information becomes available. For noise pollution research and mitigation prioritisation, it makes sense to use a fish's sensitivity to noise as a method of prioritising focus; the more sensitive the species, the more urgent the need for research. Unfortunately, there is a high level of uncertainty associated with fish species' hearing and noise-induced reactions.

A major limitation of noise pollution research in fish is that even closely related species can have differing hearing capabilities and display different reactions to noise, in addition to the individual or population level differences found within species (Popper and Hastings, 2009b). Auditory thresholds can be measured in various ways. A common method is the auditory evoked potential method, which involves playing an auditory stimulus (noise) to the fish and recording very small electrical voltage potentials that occur in the brain in response, using electrodes placed on the head. Ladich & Fay's (2013) review on the auditory evoked potential audiometry in fish provided evidence of hearing abilities varying within species. Differences were also observed in humans as a result of varying ages (Pedersen *et al.*, 1989). Audiogram results for humans have also been affected when there is ear damage present (often not recorded in fish audiogram studies as it requires dissection of the ear), or by environmental conditions such as hypoxia (Henry *et al.*, 2003; McAnally *et al.*, 2003). If the small sample of fish used are all from one area or habitat, the fish may have already been exposed to damaging noise levels and so their audiogram results would already exhibit a higher hearing threshold than a fish not yet exposed to such noise (Steinmetz *et al.*, 2009). It may, therefore, be erroneous to generalise for multiple species, even those inhabiting the same area and exposed to the same noise sources. In 2006, a study measured the impact of construction work on the migration of salmon (*Salmo salar*) but owing to a lack of available wild salmon had to generalise results from another species (Nedwell *et al.*, 2006). Farmed brown trout (*Salmo trutta*) were considered a close relative to salmon and so it was thought they would react similarly. However, at noise levels where salmon were expected to react strongly, the brown trout showed little reaction. The results of this study emphasise how generalisations should be avoided and even species considered similar or closely related should still be considered independently.

Additionally, there are concerns about the accuracy and usefulness of fish audiograms. Many audiograms have been conducted in poorly monitored acoustic environments, and may not always reflect the true hearing capabilities of the fish (Popper *et al.*, 2014). Some fish audiogram studies have used a very small sample size (e.g. Amoser and Ladich, 2003 (6 fish); Ramcharitar *et al.*, 2006 (10 fish); Wysocki *et al.*, 2009 (6 fish of each species); Gutscher *et al.*, 2011 (6 fish)). Audiograms have been conducted on humans for several decades, and indications are that information on hearing sensitivity using a small sample size provides incomplete information. Results of an audiogram can differ depending on the method used, the specific equipment involved, and the individual experimenter's technique (ISVR Consulting, 2004; Barlow *et al.*, 2014). Barlow (2014) showed that there is a high level of variability both between different audiometers and within the same audiometer, with variation of Sound Pressure Levels (SPL) at the ear for a specific frequency changing by up to 21 dB (Barlow *et al.*, 2014). There is, therefore, potential for error in audiogram testing.

As a consequence of all these factors, to allow accurate mitigation development, a larger sample size is needed to provide meaningful audiogram data. As an example, the study on humans by Davis (1995) involved selection of 48,313 participants from 4 different cities across the UK;

from this sample only 2,679 were considered acceptable audiograms. As audiogram methods improve, fish species will need to be tested (or re-tested) to ascertain a more accurate hearing threshold, which can then be used to prioritise the most noise sensitive species.

For an accurate impact assessment or for prioritisation of species mitigation using noise sensitivity from audiograms, all species present at a specific site would have to be tested independently. However, with approximately 31,900 known fish species worldwide it would be impractical to research noise impacts on each individual species. With many species to research, and suggested sample sizes being significantly larger than current studies use, an alternative method of prioritising research would prove beneficial.

2.1.3 The risk-based approach

It has been recognised that a focus limited to perturbations and stressors is insufficient for understanding the impacts on, and responses of, the affected system or its components (Kasperson *et al.*, 2003). Environmental management decisions are increasingly taking into consideration the risks to populations from anthropogenic stressors (potentially harmful stimuli). Risk assessments are a useful tool in conservation and environmental management because they determine the value of a risk relative to a threat. Risk is defined as the probability that loss or impact will occur multiplied by the severity of that loss or impact, and can be evaluated using a risk matrix (Calow, 1998). Ecological risk assessment is a way of examining risks so that they may be better avoided, reduced, or otherwise managed (Wilson and Crouch, 1987), and has been used effectively in many fields as an aid in decision making (Lackey, 1997). The goal of ecological risk assessments is to estimate the likelihood of the adverse effects of stressors on populations and ecosystems (Calow *et al.*, 1997). Risk assessment tools provide insight to help reallocate research and mitigation efforts from low-risk areas to high-risk areas. Conservation organisations are developing global prioritisation models for conservation action in marine systems that rely on ranking the impact of threats and assessing the risk the species face (e.g. Olson and Dinerstein, 1998).

Vulnerability assessments – the process of identifying, quantifying, and prioritising (or ranking) the vulnerabilities in a system – support environmental management by defining targets and developing scenarios (De Lange *et al.*, 2010). Vulnerability is the state of being open to injury (or extinction) and has been defined in environmental research as the potential for loss (Cutter, 1996). Vulnerability assessments are similar to those of risk, but have an added dimension that includes the surrounding environment as well as the actual impact on the object itself (Turner *et* *al.*, 2003). Ecological and ecosystem vulnerability assessments can be expressed with a score or level of potential impact related to a certain stressor in a given environment. A system such as this can aid in the protection of natural and high-quality ecosystems against potential impact.

Combining these two assessment tools (risk and vulnerability) would enable the degree of risk to which species are exposed to be quantified or ranked, allowing conservation efforts to be focused on a particular area. This could help reduce the decline in population density, and regulations could be introduced to protect those ecosystems under greatest threat. It is clear from previous studies that good ecosystem management requires a reliable standard of measurement, and assessing risk and vulnerability is crucial when working towards mitigation and good environmental status for a species (Micheli *et al.*, 2013; Tett *et al.*, 2013).

2.1.4 Chapter aim

Conservation and management efforts should, ideally, prioritise the allocation of resources on mitigating impacts of human activities on the environment (Teck *et al.*, 2010), and endeavour to protect the most vulnerable species. Understanding the ways in which important or manageable threats may affect ecosystems can assist in this prioritisation of mitigation. Key areas of concern previously identified include priority species and groups, such as fishery species and priority life stages (e.g. larval, smolt and juvenile), and priority habitats, especially estuarine, inlet and spawning areas, deep-sea areas and Marine Protected Areas (Lewandowski *et al.*, 2016). Munns *et al.* (2006) state that basic research to develop new theories for assessing risk is needed to address population concerns. However, the decision on which threats, ecosystems, or species should be focused on as a priority remains challenging (Halpern *et al.*, 2007).

The aim of this study is to create a reliable method of ranking marine fish species in order of priority for research and mitigation focus in relation to anthropogenic noise impacts. It is based on two primary indicators: their value to society/the ecosystem they inhabit, and their susceptibility to impact from noise. In this thesis, 'susceptibility' means having the potential to be influenced or harmed by a particular stressor – in this case noise.

The first primary indicator, which forms the basis of the framework, estimates the value of a species to society / the ecosystem using three influencing factors: population condition, commercial importance and ecosystem services provision. Population condition identifies the current state of the species' populations using their conservation status, and population trend, as measures of conservation importance. Species of least concern that have an increasing population trend will exhibit a lower overall risk of impact from noise pollution than endangered species with decreasing population levels. However, it is not sufficient to focus solely on the structure of the ecosystem in terms of the number of species present or their population condition. The prioritisation framework must also reflect that some species are considered of greater commercial importance than others, and that any significant decline in commercially viable populations could have an economic impact on society. Quantifying the ecosystem services that species provide to human well-being will identify species on whom the ecosystem may depend. Marine ecosystems provide numerous goods and services which can be regarded as a link between the functioning of ecosystems and their contribution to human well-being (Daily *et al.*, 1997), including supporting, provisioning, regulating and cultural services (Costanza *et al.*, 1997). Ecologists recognise that some species (keystone species and ecosystem engineers) are essential to the integrity of the ecosystem owing to their key roles and ability to create a more complex habitat either morphologically or behaviourally (Coleman and Williams, 2002; Garibaldi and Turner, 2004). Species' engineering activities influence both biological diversity and ecosystem function and so their decline would have marked ecosystem-level impacts through supporting services. Therefore, knowledge of the ecosystem service role of marine fish species has been included within the ecosystem services provision factor.

The second primary indicator, the susceptibility indicator, examines the scale of influence of a threat and is devised using two influencing factors: the likelihood of exposure to the noise threat, and the severity of impact if exposure occurs. The likelihood of exposure has been included in many ecosystem risk assessments as it determines the degree of contact a fish has with a threat (De Lange *et al.*, 2010). The severity of an impact, should it occur, is important to consider alongside the exposure potential. A common noise threat having little impact would have a different outcome to a rare noise threat causing serious injury or death. The term "risk" represents both the likelihood and severity of exposure to a threat that a species may face (Calow, 1998), and so a risk-based approach in terms of a risk matrix, is used to quantify the relative susceptibility of a species.

Given the importance of conservation measures in the marine environment and the uncertainties of existing approaches to guide and inform research direction, there is a need for robust tooling, such as the framework presented here, to underpin processes, and to identify, quantify and prioritise concerns regarding the potential impact of noise on marine fish. A prioritisation assessment tool showing the likelihood of different fish species' being harmed by noise pollution would benefit researchers and decision-makers by directing research toward the most deserving species. It would show those at higher risk of noise pollution impacts, and those whose diminishing presence would have a bigger impact on the ecosystem and/or commercial fisheries. This framework uses a combination of risk and vulnerability assessment methods based on key considerations pertaining to the effects of noise impacts on fish in the marine environment, to provide a repeatable semi-quantitative framework for prioritising species for use in future noise pollution research.

2.2 Method

A Prioritisation Index score (overall species' priority score) was calculated by multiplying the scores of two primary indicators: value and susceptibility (Figure 2.1). The value indicator comprised of three influencing factors: population condition, commercial importance, and ecosystem service provision. The susceptibility - the potential to be influenced or harmed by a particular threat (IUCN, 2014) - indicator was determined using a risk-based matrix approach, comprised of two influencing factors: likelihood of exposure to noise, and the severity of impact if exposed. The Prioritisation Index (PI) ranked species according to their overall priority score allowing the most vulnerable species to be identified for future research and mitigation.

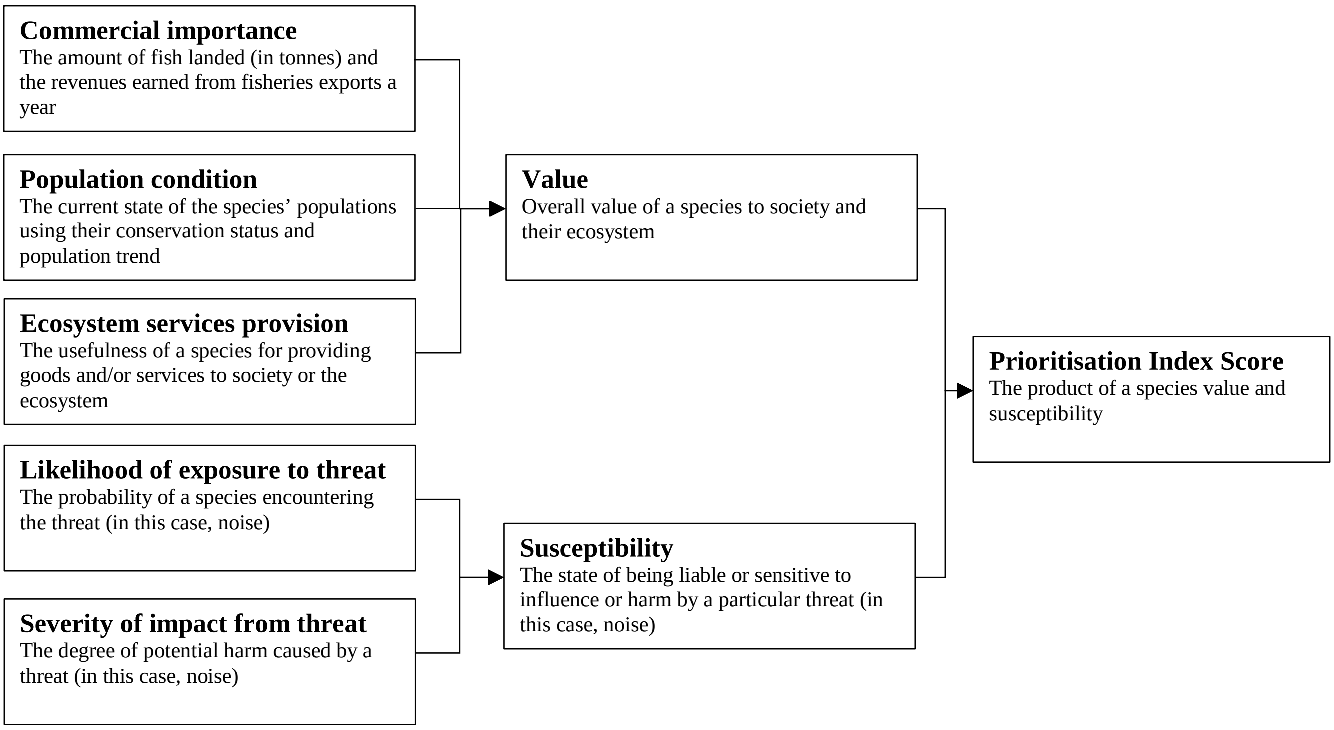


Figure 2.1. Conceptual diagram summarising the Prioritisation Index framework wherein a ranking of species provides a tool for decision makers, and influences the direction and focus of future research and evaluation. Information from the IUCN and peer-reviewed sources were used to predict the value of a species and its susceptibility to noise pollution (sources used are listed in Table 2.2, Table 2.3, Table 2.5, Table 2.7 and Table 2.8, Figure A2.2). The ranking produced indicates those species in greatest need of research and mitigation efforts.

2.2.1 Study species

The study focused on UK marine fish species of conservation, commercial and/or ecological importance. In total, 52 species were chosen as: they are either fished in UK waters or by the UK fishing fleet elsewhere (as recorded by the Food and Agriculture Organization (FAO)); are considered a keystone species providing useful services to the ecosystem; or have populations identified as facing a higher risk of extinction (Table A2.1 provides the full list of species considered). To keep the number of species at a manageable level for this case study, all 52 species were analysed for population condition, and those species who were considered to have low conservation importance (that is species who are of least concern on the IUCN Red List or near threatened on the IUCN Red List but with increasing population trends) were omitted from further study (Table A2.1). The 27 remaining species were then subjected to a comprehensive analysis using the full Prioritisation Index framework.

2.2.2 Noise sources

Underwater noise pollution from shipping, military and civil sonar, air guns, dredgers, pile drivers, ocean science studies, explosions, fishing and aquaculture equipment (including trawlers, pingers and acoustic deterrent devices), and from noises associated with oil and gas production (such as construction of rigs and turbines) are of concern (OSPAR, 2009; Department of Energy and Climate Change, 2011). As such, these sources were included in the framework (Table 2.1). Source levels (SL) and frequencies in Hertz (Hz) of the noise sources used here were collated from peer-reviewed publications. Minimum and maximum SL were taken from several publications (Table 2.2) and an average was calculated (Table 2.1). The SL relates to the sound pressure level 1 m from a hypothetical monopole source with sound power output equivalent to the actual source and is expressed in terms of dB re 1 μPa at 1 m. It should be noted that the SL is typically estimated based on far field measurements taken at ranges much greater than 1 m from the source, and then back projected to estimate the level one would expect 1 m from the idealised monopole source. There are limitations to using this method of calculating SL (*see* Robinson and Lepper, 2013) but it does represent the most widely accepted method of characterising the acoustic output of a source. It should be noted that the SL does not represent the sound pressure level experienced by the fish, which will typically be at some unknown range from the source. The noise sources considered here are characterised by their sound pressure levels only. Particle motion is not covered owing to the limited data available, but would make a useful addition to the model in the future as some species are more impacted by sound pressure and others by particle velocity (Horodysky *et al.*, 2008; Ladich and Fay, 2013).

Table 2.1. The noise source levels used in this study. Data are taken from the average of peak source level identified from previous works (for list of references see Table 2.2). Intermittent sources are grouped into long-term or short-term activities depending on their likely duration in one area.

Activity	Type of noise	Source level (dB re 1 µPa at 1 m)
Wind farm operation	Continuous	127
Fishing vessels	Continuous	153
Recreational craft	Continuous	154
Passenger vessels	Continuous	162
Dredgers	Continuous	167
Cargo vessels	Continuous	179
Echo-sounders	Intermittent (LT)	210
Pile driving	Intermittent (LT)	217
Low-frequency sonar	Intermittent (ST)	223
Air Gun	Intermittent (ST)	243
Explosions	Intermittent (ST)	258

Table 2.2. The data used to calculate average source levels and relevant references, where $_{(p)}$ refers to peak level, $_{(p-p)}$ refers to peak-to-peak, and $_{(z-p)}$ refers to zero-to-peak. Source level data is limited due to difficulties in determining the source level as source levels are always inferred from measurements at greater distances that are influenced by a situation specific propagation (Ainslie *et al.*, 2009).

(a) Activity	Minimum SL reported	Maximum SL reported	Metric	Reference
Air gun	na	223	dB re 1 μPa$_{(p)}$ at 1 m	McCauley *et al.* (2003)
Air gun	233	263	dB re 1 μPa at 1 m	Bowles *et al.* (1994)
Air gun	215	224	dB re 1 μPa at 1 m	Thompson *et al.* (1998)
Air gun	na	242	dB re 1 μPa$_{(z-p)}$ at 1 m	Guan *et al.* (2015)
Air gun	235	240	dB re 1 μPa at 1 m	Hatch and Wright (2007)
Air gun	222	265	dB re 1 μPa$_{(p-p)}$ at 1 m	Kongsberg *et al.* (2010)
Air gun	258	261	dB re 1 μPa at 1 m	Miller *et al.* (2009)
Air gun	260	262	dB re 1 μPa at 1 m	OSPAR Commission (2009b)
Cargo vessel	175	198	dB re 1 μPa at 1 m	Marine Management Organisation (2015)
Cargo vessel	178	192	dB re 1 μPa at 1 m	Marine Management Organisation (2015)
Cargo vessel	na	192	dB re 1 μPa at 1 m	Marine Management Organisation (2015)
Cargo vessel	160	180	dB re 1 μPa at 1 m	Richardson *et al.* (1995)
Cargo vessel	169	198	dB re 1 μPa at 1 m	Marine Management Organisation (2015)
Cargo vessel	112	187	dB re 1 μPa at 1 m	Jones *et al.* (2017)
Cargo vessel	na	190	dB re 1 μPa at 1 m	Richardson *et al.* (1995)
Cargo vessel	185	190	dB re 1 μPa at 1 m	Thiele (1982)
Dredger	150	162	dB re 1 μPa at 1 m	Marine Management Organisation (2015)
Dredger	na	160	dB re 1 μPa at 1 m	Marine Management Organisation (2015)
Dredger	168	186	dB re 1 μPa at 1 m	OSPAR Commission (2009b)
Dredger	172	188	dB re 1 μPa^{2}m^2	Ainslie *et al.*, (2009)
Dredger	123	191	dB re 1 μPa at 1 m	Jones *et al.* (2017)
Echo-sounder	200	230	dB re 1 μPa at 1 m	National Science Foundation (U.S) (2008)
Echo-sounder	na	201	dB re 1 μPa at 1 m	National Research Council (2003)
Low frequency sonar	215	240	dB re 1 μPa at 1 m	Marine Management Organisation (2015)
Low frequency sonar	na	215	dB re 1 μPa at 1 m	Hatch and Wright (2007)
Passenger vessel	168	168	dB re 1 μPa at 1m	Marine Management Organisation (2015)
Passenger vessel	171	171	dB re 1 μPa at 1m	Marine Management Organisation (2015)
Passenger vessel	145	169	dB re 1 μPa at 1m	Erbe (2002)
Passenger vessel	115	181	dB re 1 μPa at 1m	Jones *et al.* (2017)
Passenger vessel	160	170	dB re 1 μPa at 1m	Richardson *et al.* (1995)
Pile driving	226	250	dB re 1 μPa^{2}s	Bailey *et al.* (2010)
Pile driving	205	220	dB re 1 μPa^{2}s	Ainslie *et al.* (2012)
Pile driving	na	200	dB re 1 μPa^{2}s	Dahne *et al.* (2013)
Pile driving	177	202	dB re 1 μPa^{2}s	Hawkins (2006)
Pile driving	na	208.2	dB re 1 μPa^{2}s	Yang *et al.* (2015)

Pile driving	205	214	dB re 1 μPa^{2}s	Robinson *et al.* (2007)
Pile driving	215	220	dB re 1 μPa^{2}s	De Jong and Ainslie (2008)
Pile driving	243	257	dB re 1 μPa$_{(p)}$ at 1m	OSPAR Commission (2009b)
Recreational vessel	150	175	dB re 1 μPa at 1m	Marine Management Organisation (2015)
Recreational vessel	105	152	dB re 1 μPa at 1m	Marine Management Organisation (2015)
Recreational vessel	160	175	dB re 1 μPa at 1 m	OSPAR Commission (2009b)
Recreational vessel	113	205	dB re 1 μPa at 1 m	Jones *et al.* (2017)
Fishing vessel	na	140	dB re 1 μPa at 1 m	National Research Council (2003)
Fishing vessel	147	158	dB re 1 μPa at 1 m	Marine Management Organisation (2015)
Fishing vessel	na	151	dB re 1 μPa at 1 m	Malme *et al.* (1989)
Fishing vessel	na	158	dB re 1 μPa at 1 m	Greene (1985)
Fishing vessel	113	202	dB re 1 μPa at 1 m	Jones *et al.* (2017)
Wind farm operation	73	153	dB re 1 μPa at 1 m	Marine Management Organisation (2015b)
Wind farm operation	126	142	dB re 1 μPa at 1 m	Andersson *et al.* (2012)
Wind farm operation	na	142	dB re 1 μPa$_{(rms)}$ at 1 m	OSPAR Commission (2009b)
Explosion	272	287	dB re 1 μPa$_{(p)}$ at 1 m	OSPAR Commission (2009b)
Explosion	205	304	dB re 1 μPa at 1 m	Hildebrand (2009)
Explosion	231	251	dB re 1 μPa^{2}m^{2}s	Ainslie (2010)

2.2.3 Scoring

The influencing factors included in the Prioritisation Index (PI) were selected according to their contribution to Good Environmental Status as stated in the MSFD (2008/56/EC). Population condition, commercial importance, and ecosystem services provision together provide each species with a score of their relative value to society/ecosystems (Eq. 1 *see* Section 2.2.6 for equations). Each species' level of susceptibility, or risk, considers the likelihood of exposure to noise pollution and the severity of impact if exposure occurs (Eq. 2).

The factors themselves consist of either one or two variables. Where two variables were used, the two scores were added together to produce an overall factor score. The sum of the variables was used instead of the product to avoid value scores of zero occurring for species with a stable population trend (which scores zero (*see* Table 2.4 for detail on scoring)).

The range of scores varied between factors owing to the classification of data for each variable (categorical, ordinal or nominal); all final scores were normalised giving each species a final factor score between 0 and 1, providing equal weighting to each before the final prioritisation calculation. The Prioritisation Index score was the product of the scores of species' value and susceptibility to noise pollution. Larger positive numbers denote higher priority species (with up to a maximum possible score of 3), suggesting a greater need for focused research efforts.

2.2.4 Species value

The value score represented a species overall population condition, commercial importance and ecosystem services provision. To calculate an overall value score for each species, their three variable scores were normalised to provide equal weighting to each category and then the normalised scores were added together (Eq. 1).

2.2.4.1 Population condition

Population condition was based on two variables: population status (Table 2.3), and trend, as identified by the International Union for Conservation of Nature Red List (IUCN, 2014). The number and labels of each category corresponded with those used by the IUCN (2001). A linear scale (1: least concern to 6: possibly extinct) was employed to match the five criteria (Figure A2.1) used by the IUCN to evaluate and assign a category (Table 2.4a). Trend was scored 1 (decreasing), 0 (stable) or -1 (increasing) (Table 2.4b). For this study, the population trend as stated by the IUCN was used, but regional population status and trends should be used if available and if more appropriate to a specific study. The IUCN criteria use the total estimated number of mature individuals of a species to record increasing, decreasing or constant trends. Scores for species listed with unknown population trends, and those not yet assessed by the IUCN, were estimated and evidenced using other sources (Table 2.5). The overall population condition for each species was calculated as the sum of the two assigned scores.

Table 2.3. IUCN Red List conservation status categories and definitions (International Union for Conservation of Nature, 2001).

Status	Description
Least concern (LC)	A taxon is Least Concern when it has been evaluated against the criteria and does not qualify for Critically Endangered, Endangered, Vulnerable or Near Threatened. Widespread and abundant taxa are included in this category.
Near threatened (NT)	A taxon is Near Threatened when it has been evaluated against the criteria but does not qualify for Critically Endangered, Endangered or Vulnerable now, but is close to qualifying for or is likely to qualify for a threatened category in the near future.
Vulnerable (VU)	A taxon is Vulnerable when the best available evidence indicates that it meets any of the criteria A to E for Vulnerable, and it is therefore considered to be facing a high risk of extinction in the wild.
Endangered (EN)	A taxon is Endangered when the best available evidence indicates that it meets any of the criteria A to E for Endangered, and it is therefore considered to be facing a very high risk of extinction in the wild.
Critically endangered (CR)	A taxon is Critically Endangered when the best available evidence indicates that it meets any of the criteria A to E for Critically Endangered, and it is therefore considered to be facing an extremely high risk of extinction in the wild.
Extinct (EX)	A taxon is Extinct when there is no reasonable doubt that the last individual has died. A taxon is presumed Extinct when exhaustive surveys in known and/or expected habitat, at appropriate times (diurnal, seasonal, annual), and throughout its historic range have failed to record an individual. Surveys should be over a time frame appropriate to the taxon's life cycles and life form.

Table 2.4. (a) Allocation of population condition scores using IUCN Red List. (b) Allocation of population trend scores using IUCN Red List. Decreasing population trends receives the highest score as declining populations face higher levels of extinction risk.

a	IUCN Red List status	Assigned score
	Least concern	1
	Near threatened	2
	Vulnerable	3
	Endangered	4
	Critically Endangered	5
	Possibly extinct	6

b	IUCN Red List status	Assigned score
	Increasing	-1
	Stable	0
	Decreasing	1

Table 2.5. Sources of information used to determine the population condition of fish species if species was not evaluated or data deficient on the IUCN Red List. Scoresbased on the information derived from these sources were ranked in the same way as the information from the IUCN Red List shown in Table 2.4. Information used to calculate scores for all other species not listed here was taken from the IUCN Red List.

Population status	**Evidence source**
sandeel (*Hyperoplus lanceolatus*)	Joint Nature Conservation Committee (2010)
hake (*Merluccius merluccius*)	Fundazioa (2011)
garfish (*Belone belone*)	Samsun *et al.* (2006)
sole (*Solea solea*)	Morat *et al.* (2014)
sandy ray (*Raja cirularis*)	Baxter *et al.* (2011)
meagre (*Sciaena umbra*)	Malak *et al.* (2011)
seahorse (*Hippocampus hippocampus*)	OSPAR Commission (2013)
whiting (*Merlangius merlangus*)	Seafish (2013)
catfish (*Anarhichas lupus*)	Federal Register (2009)
seabream (*Sparus aurata*)	Brown *et al.* (2005)
ling (*Molva dypterygia*)	The Norwegian Ministry of Trade Industry and Fisheries (2013)

Population trend	**Evidence source**
sandeel (*Hyperoplus lanceolatus*)	Royal Society for the Protection of Birds (2013)
hake (*Merluccius merluccius*)	National Oceanic and Atmospheric Administration (2013a)
lamprey (*Petromyzon marinus*)	Great Lakes Fishery Comission (2012)
seabass (*Dicentrarchus labrax*)	National Oceanic and Atmospheric Administration (2013b)
salmon (*Salmo salar*)	National Oceanic and Atmospheric Administration (2015)
cod (*Gadus morhua*)	National Oceanic and Atmospheric Administration (2013c)
pollock (*Pollachius virens*)	National Oceanic and Atmospheric Administration (2013d)
ling (*Molva dypterygia*)	Poulsen *et al.* (2007)
haddock (*Melanogrammus aeglefinus*)	Seafish (2013b)
catfish (*Anarhichas lupus*)	Catfish Conservation Group (2015)
halibut (*Hippoglossus hippoglossus*)	Bedford Institute of Oceanography (2013)

2.2.4.2 Commercial importance

Commercial importance represents a species' economic worth according to export value in one year in US dollars, and tonnes landed overall recorded by British fleets in one year. By including both these variables the commercial importance score (and therefore Prioritisation Index) took into account each species importance from both their relative economic value and their exploitation levels. Landing tonnes measured the catches landed by UK fleets, whilst export value included fish landed by both the UK and foreign fleets that provide financial benefits to the UK. Owing to the lack of available catch data each variable only included data for 2011 (Food and Agriculture Organization, 2014, 2015). No refactoring or scaling was used here as the data was already numerical. The export value and tonnes for each species were added together and normalised to provide the score for commercial importance.

2.2.4.3 Ecosystem services provision

Eleven goods/benefits were analysed in this case study (Table 2.6), with each species being allocated a score to reflect their contribution to each ecosystem service, following the method used by Potts *et al.* (2014), from 0 ("no contribution") to 4 ("significant contribution"). Wherever possible, scores were taken directly from Potts *et al.* (2014); other species scores were allocated based on available literature (*see* Table A2.2 for full scores).

Table 2.6. List of the ecosystem services used in the index. Services provide provisioning, regulating and cultural goods or benefits. The framework in this case assumes all goods/benefits are equally important and are given equal weighting.

Ecosystem goods/benefits
Fish feed (wild, farmed, bait)
Ornaments and aquaria
Medicines and blue biotechnology
Healthy climate
Prevention of coastal erosion
Sea defence
Waste burial/removal/neutralisation
Tourism and nature watching
Spiritual and cultural well-being
Aesthetic benefits
Education

The individual scores for a species contribution to each of the 11 ecosystem services were then totalled. The overall scores fell between 0 and 44, with a species scoring 44 if it significantly contributed to all 11 ecosystem services, or 0 if it did not contribute to any. As the Prioritisation Index already incorporates a value for commercial importance, the ecosystem service 'fishing for human use and consumption' was omitted from ecosystem services provision so as not to be included twice.

2.2.5 Species susceptibility to noise

The habitat types (Table 2.7) were assessed using a risk matrix approach according to the likelihood of noise exposure from several different noise sources (e.g. shipping, pile driving, sonar), and the severity of impact from those noise sources (Figure 2.2b). Risk scores from 1 to 15 were allocated to each noise source depending on the intensity and duration of the noise emitted (Figure 2.2a). Each habitat was then given an overall risk score constituting the sum of risk scores for all noise sources that can occur in that specific habitat (Eq. 2) (Figure 2.3). Each species was also allocated a susceptibility score based on its habitat's overall risk. Some species, such as European eel (*Anguilla anguilla*) and other diadromous species, could be allocated different scores for different life stages, but in this study, the score was based on the main habitat where they spend the majority of their adult life. As with other scores, susceptibility was normalised to produce equal weighting.

Table 2.7. Descriptions of habitat types used in the matrix. Descriptions taken from Federal Government Data Comittee (2012).

Habitat types	Description
Estuarine	The transition area between freshwater and ocean. Maximum depth is usually less than 30m.
Intertidal	Area of the shore that lies between the highest normal high tide and the lowest normal low tide. Maximum depth of ~15m.
Near-shore	Ocean habitat extending from landward limit to the 30m depth contour.
Off-shore	Extends from 30m depth contour to the continental shelf break.
Continental Shelf	The gently sloping plain covered by relatively shallow water. Maximum depth of ~200m.
Deep water	The area beyond the continental shelf where the continental slope begins to fall away. Minimum depth of ~200m.

2.2.5.1 Likelihood of exposure

The likelihood of exposure is higher if noise is continuously present so the duration of the noise source, whether intermittent (short- or long-term) or continuous, was considered. Short-term exposure was allocated the lowest risk score, and continuous exposure allocated highest (Figure 2.2 a).

2.2.5.2 Severity of impact if exposed

The severity of impact was quantified using impact data observed in previous peer-reviewed studies (Table 2.8). The greater the intensity of the noise source, the more likely the effects would be observed. For example, a noise emission under 80 dB re 1 μPa at 1 m is unlikely to cause any adverse effects, whilst a noise emission of 130 dB re 1 μPa at 1 m has the potential to cause four detrimental effects and a noise emission of 210 dB re 1 μPa at 1 m could potentially cause 13 separate effects (Figure 2.2 a). If multiple effects were more likely to be observed, then a higher risk score was allocated. The susceptibility factor also took into account a species' sensitivity to a particular threat wherever possible. Whenever audiogram data were available auditory thresholds for the species were considered, and their habitat risk score accurately reflected only the noise sources falling within their reported hearing range. This assessment was achievable for 27 of the original 52 species considered. All remaining species were treated as though they could hear all noise sources excluding high frequency sonar and echo-sounders owing to this type of specialist hearing being extremely rare (shown only in 2 of 78 species with published audiograms).

Table 2.8. Minimum decibel levels where effect has been observed. SPL refers to sound pressure level in dB re 1 μPa at 1 m and is indicative of the average amount of sound at one location (Ainslie *et al.*, 2009). SPL was used over Sound Exposure Level (SEL) owing to studies reporting SPL being available for all observed effects listed. Studies on intermittent sources tend to focus only on SEL and can be erroneous as it relies on the estimated distance of the fish from the noise source.

Effect observed	Minimum dB level effect observed at	Metric	Reference
Altered swimming behaviour	80	SPL	Andersson *et al.* (2007)
Decreased antipredator	108	SPL	Simpson *et al.* (2015)
Increased cortisol secretion	123	SPL	Celi *et al.* (2016)
Diminished reproductive behaviour	127	SPL	Bruintjes and Radford (2013)
Avoidance behaviour	130	SPL	Suzuki *et al.* (1980)
Temporary Threshold Shift	132	SPL	Codarin *et al.* (2009)
Habitat and settlement choice impacted	136	SPL	Simpson *et al.* (2016)
Increase ventilation rate	139	SPL	Bruintjes *et al.* (2016)
Altered shelter-use behaviour	142	SPL	Picciulin *et al.* (2010)
Reduced foraging efficiency	150	SPL	Purser and Radford (2011)
Defence and aggression affected	161	SPL	Sebastianutto *et al.* (2011)
Altered schooling and dispersion	168	SPL	Doksæter *et al.* (2012)
Long term hair cell damage	222	SPL	McCauley *et al.* (2003)

a.		Type of exposure		
dB re 1 μPa	**Effect observed**	**Intermittent (ST)**	**Intermittent (LT)**	**Continuous**
80	**Altered swimming behaviour**	1	2	3
108	**Decreased antipredator behaviour**	2	3	4
123	**Increased cortisol secretion**	3	5	5
127	**Diminished reproductive behaviour**	4	4	6
130	**Avoidance behaviour**	5	6	7
132	**Temporary Threshold Shift**	6	7	8
136	**Habitat and settlement choice impacted**	7	8	9
139	**Increased ventilation rate**	8	9	10
142	**Altered shelter-use behaviour**	9	10	11
150	**Reduced foraging efficiency**	10	11	12
161	**Defence and aggression effected**	11	12	13
168	**Altered schooling and dispersion**	12	13	14
222	**Permanent ear hair damage**	13	14	15

b.	Pile driving	Wind farm operation	Air guns	Explosions	Dredgers	Recreational craft	Passenger vessels	Cargo vessels	Fishing vessels	Low Frequency Sonar	Echo-sounders	Susceptibility score (inclusive of echo-sounder)
Estuarine	0	0	0	0	13	12	13	0	0	0	13	38 (51)
Intertidal	0	0	0	0	13	12	0	0	12	0	13	37 (50)
Near shore	14	0	13	0	13	12	13	14	12	14	13	105 (118)
Off shore	14	5	13	13	13	12	13	14	12	14	13	123 (136)
Shelf	14	5	13	13	13	12	13	14	12	14	13	123 (136)
Deep water	0	0	13	13	0	12	13	14	12	14	13	91 (104)

Figure 2.2. Risk matrix approach used to calculate species susceptibility scores. **(a)** To quantify a species susceptibility to noise pollution a risk matrix with both the likelihood of exposure and severity of impact if exposed was used. Risk scores range from 1 – 15 following a general risk matrix method of using consecutive numbers as exposure/severity increases. The longer the duration of the noise source, the greater the risk of a fish being exposed to the noise. The higher the intensity, the higher the number of effects likely to be observed. The Sound Pressure Levels (dB re 1 μPa at 1 m) used to represent severity of impact were taken from peer-reviewed publications (Table 2.8). **(b)** Each noise source included in the study was given a score using the risk matrix (Figure 2.2a) depending on the average source level and type of exposure collected from the literature (Table 2.1). The scores of sources potentially present in each environment were summed to provide an overall score for the habitat. Each species was allocated a susceptibility score based on the habitat it inhabits. Only two species listed in the index can hear the high frequencies (12,000 – 710,000 Hz) produced by echo-sounders and so this noise source was only included in the habitat score for Cod (*Gadus morhua*) and Herring (*Clupea harengus*).

$$\begin{aligned} Estuarine\ suceptibilty\ score &= Dredging + Recreational\ craft + Passenger\ vessels \\ &= 12 + 11 + 13 \\ &= 36 \end{aligned}$$

Figure 2.3. Example of susceptibility score being calculated for an estuarine habitat where dredging, recreational craft and passenger vessels occur. Dredging is a continuous noise source emitting an average of 157 dB re 1 μPa at 1 m; recreational craft are a continuous noise source emitting an average of 146 dB re 1 μPa at 1 m; passenger vessels are a continuous noise source emitting 163 dB re 1 μPa at 1 m.

2.2.6 Assessing priority

The Prioritisation Index score for a species was calculated from the product of the normalised value and susceptibility scores (Eq. 3). This amalgamated score then allowed each species to be listed in priority order from most vulnerable (highest score) to least vulnerable (lowest score closest to 0): the higher the score, the more potential for significant impacts and the greater the need to focus future noise pollution research efforts.

$$V = \left(\left(\frac{P_s+P_t}{P_m}\right) + \left(\frac{C_e+C_i}{C_m}\right) + \frac{E}{E_m}\right) \qquad \text{(Eq. 1)}$$

$$S = \sum R_n \qquad \text{(Eq. 2)}$$

$$I = V * S \qquad \text{(Eq. 3)}$$

Prioritisation Index is calculated by the Value multiplied by the Susceptibility where V is the value of a given species, P_s is the IUCN population status, P_t is the Population trend, P_m is the maximum value of $P_s + P_t$ across all species, C_e is the commercial export value, C_i is the commercial landing tonnes, C_m is the maximum value of $C_e + C_i$ across all species, E is the ecosystem services provision, E_m is the maximum value of E across all species, S is the susceptibility to noise pollution, R_n is risk score for the species based on the likelihood of exposure within its habitat and the severity of effects that could be observed, and I is the Prioritisation Index (overall priority score for a species).

2.2.7 Relative Weight Analysis

To ensure the correct weightings were used when assessing the priority of species Sensitivity Analysis and Relative Weight Analysis (Tonidandel and Lebreton, 2015) were run on the data in R 3.3.2. This tested whether one variable has more influence on the final priority score than the other variables and allows corrections to be made to the equation to ensure all variables carry equal weight.

2.3 Results

The Relative Weight Analysis showed that the variables' relative weights did not differ significantly from each other (for population condition, commercial importance and ecosystem services provision the raw weights were 0.41, 0.35 and 0.24 respectively) as the confidence intervals did not include zero. Therefore, no extra weightings were required for the overall priority calculation.

The full possible range of prioritisation scores was from 0 to 3, but the PI results showed overall scores ranging from 0.107 to 1.242, indicating that all the species assessed experience some risk of harm from noise pollution (Figure 2.4). With the exception of the two highest priority species, the overall priority scores were a continuum. Atlantic cod (*Gadus morhua*), the species ranked highest with a score of 1.242, is far more at risk than Mackerel (*Scomber scombrus*), prioritised 2nd with a score of 0.970, and all the other species with scores of 0.757 and below, based on this assessment. No one species was considered highest priority in all contributing factors, although some species were consistently high scorers in more than one category. Atlantic cod ranked top for both primary indicators: value and susceptibility.

Atlantic cod, listed in the IUCN Red List as vulnerable with an unknown population trend, was the highest scorer in the commercial importance category, and it ranked joint 6th for ecosystem services provision. Combined, this allowed the species to rank top in overall value to society (Table 2.10).

Although it is species priority that is of main interest, individual factor scores provide added information, such as, indicating whether species' loss will affect ecosystems or fisheries, and in determining where risk of noise impacts are highest (Table 2.9). The highest scoring species within each of the value categories, overall value, and susceptibility category are shown in Table 2.10.

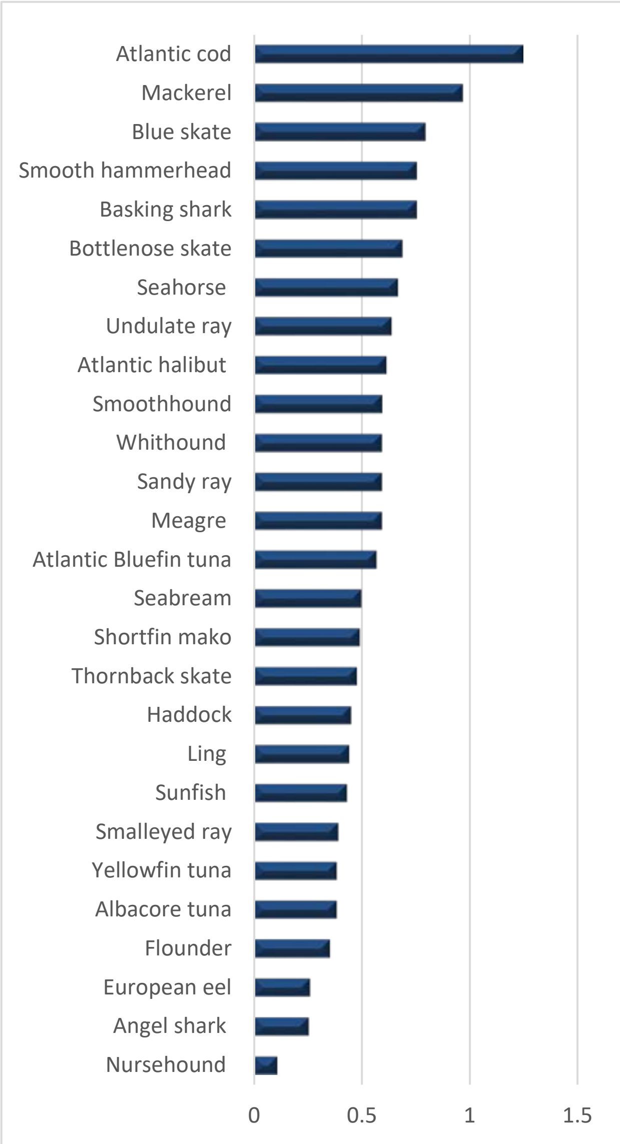


Figure 2.4. Prioritisation Index for 27 UK marine fish species. Species are ranked in order of priority for future focus relating to noise pollution. Zero denotes low priority species, less likely to be heavily impacted by noise and so indicates lower urgency in terms of conservation or management efforts. Larger positive numbers denote higher priority species (with up to a maximum possible score of 3), suggesting a greater need for focused research conservation and effective management.

Table 2.9. Species' scores for each factor/indicator and overall prioritisation score. The following species were removed after assigning population condition as currently considered not significantly 'at risk': Atlantic pollock, Bullet tuna, catfish, dab, Frigate tuna, garfish, goby, hake, herring, lamprey, mullet, plaice, pout, runner, salmon, Sand eel, sardine, anchovy, Sea bass, Skipjack tuna, sole, solenette, sprat, stickleback, whiting, wrasse.

Species	Population condition	Commercial importance	Ecosystem services provision	Overall species value	Susceptibility	Prioritisation score
Albacore tuna	0.429	0.000	0.159	0.588	0.662	0.389
Angel shark	0.857	0.000	0.091	0.948	0.279	0.265
Atlantic Bluefin tuna	0.714	0.000	0.159	0.873	0.662	0.578
Atlantic cod	0.429	0.564	0.250	1.242	1.000	1.242
Atlantic halibut	0.571	0.001	0.114	0.686	0.897	0.615
Basking shark	0.571	0.000	0.273	0.844	0.897	0.757
Blue skate	0.857	0.000	0.205	1.062	0.765	0.812
Bottlenose skate	0.714	0.000	0.205	0.919	0.765	0.703
European eel	0.857	0.001	0.273	1.131	0.272	0.308
Flounder	0.286	0.000	0.182	0.468	0.765	0.358
Haddock	0.429	0.150	0.114	0.692	0.662	0.458
Ling	0.429	0.000	0.250	0.679	0.662	0.449
Mackerel	0.286	0.500	0.295	1.081	0.897	0.970
Meagre	0.571	0.000	0.091	0.662	0.897	0.594
Nursehound	0.286	0.000	0.114	0.399	0.279	0.112
Sandy ray	0.571	0.000	0.091	0.662	0.897	0.594
Sea bream	0.571	0.001	0.091	0.663	0.765	0.507
Seahorse	0.571	0.000	0.318	0.890	0.765	0.680
Shortfin mako	0.571	0.000	0.182	0.753	0.662	0.499
Smalleyed ray	0.429	0.001	0.091	0.520	0.765	0.398
Smooth hammerhead	0.571	0.000	0.273	0.844	0.897	0.757
Smoothhound	0.571	0.001	0.091	0.663	0.897	0.595
Sunfish	0.571	0.000	0.091	0.662	0.662	0.438
Thornback skate	0.429	0.003	0.205	0.636	0.765	0.486
Undulate ray	0.714	0.000	0.136	0.851	0.765	0.650
Whithound	0.571	0.000	0.091	0.662	0.897	0.594
Yellowfin tuna	0.429	0.001	0.159	0.588	0.662	0.389

Table 2.10. Normalised results of the Prioritisation Index categories showing the species who scored the highest score in each influencing factor. All normalised scores lie between 0 and 1. As the value score is the sum of the population condition, commercial importance and ecosystem services score its maximum score is 3, but all other categories have a maximum score of 1. Species scores differed amongst categories and so the ranking was different for each.

Category	Species	Score
Population condition	Angel shark, *Squatina squatina*	0.857
	European eel, *Anguilla anguilla*	0.857
	Blue skate, *Dipturus batis*	0.857
Commercial importance	Atlantic cod, *Gadus morhua*	0.564
Ecosystem services	Seahorse, *Hippocampus hippocampus*	0.318
Overall value	Atlantic cod, *Gadus morhua*	1.242
Susceptibility	Atlantic cod, *Gadus morhua*	1.00

2.4 Reasons for chosen approach

Several variances on the model were conducted before choosing the final method stated in this chapter:

Population trend score from 1-3 (1 – increasing, 2 – stable, 3 – decreasing):

The population trend scores were chosen to be -1 to 1 for the final framework because having all positive scores (1-3) meant that if a species population was increasing, it's score also increased giving it a higher overall score closer to the maximum (indicating higher vulnerability). Using a minus score meant that positive trends resulted in population condition scores decreasing thus more accurately reflecting lower vulnerability.

Multiplication of factors to provide overall indicator scores:

The decision to use the sum of factor scores over the product was made due to some species in the commercial importance factor being allocated a score of zero. Some fish species are not commercialised at all, and so had no recorded catches by the UK fishing fleet. This meant that a score of zero was allocated, which, when multiplied against the two other factors to produce an overall value score resulted in the species having no value. Seahorse ranked 1st in ecosystem services and joint 7th in population condition out of the 27 species listed but due to there being no recorded commercial catches it would have been allocated a value score of zero which would have drastically reduced its overall prioritisation score and position in the Index.

Keeping full list of species:

The index starts with 52 species, but uses the first factor of population condition to reduce this number. This was done to keep the list a reasonable size and focus on those species whose population is unstable, in decline or already at dangerously low levels. There is no reason not to include more species if needed as there is no size limit to the Prioritisation Index method. This chapter uses the UK as a case study to show the potential use and application of the PI and so it was not essential to keep every species included in the list. The flexibility of the framework allows the user to make an informed decision on how many species to include, whether to include more than one life stage, or even add in more categories specific to the assessment needed.

Normalising factors using maximum-recorded score:

Normalising scores using the maximum-recorded (the highest score allocated to any one species) score allowed one species to score the highest possible score, with all other species scores then shown relative to that highest scorer. Using this method allowed relative scores to be calculated, but not absolute scores that could then be compared to scores from other Prioritisation Indices from different locations, times or with different species included. Using the maximum possible value to normalise the scores does allow absolute scores to be calculated and comparisons made between indices. The maximum possible score could not be used for the commercial importance factor as there is no upper limit available on landing tonnes or export value. Even though fishing quotas are available, the catch numbers could rise above the limit due to errors in reporting or by-catch. For these reasons, commercial importance was normalised using the maximum-recorded method for both alternative approaches.

These alternative methods produce very different results to the method chosen (for example, Figure 2.5 shows the results of the PI if data was normalised differently). The Index scores are higher when using the maximum-recorded normalising approach, reaching highs of 1.850 compared to the top scorer of 1.242 in the maximum-possible approach. The priority order of the species also differ between the two approaches. Seahorse, for example, has risen up the priority list as its ecosystem services score has more weight when using the maximum-recorded approach; instead of having 44 possibilities the maximum is 23 (Figure A2.2). However, the chosen method was identified as the best option as it allows absolute scores to be calculated which can be used to compare priorities from multiple PI studies. Using the maximum-recorded value, for example, would only provide relative priority of species within one study as the maximum-recorded score would likely change between studies depending on the species included.

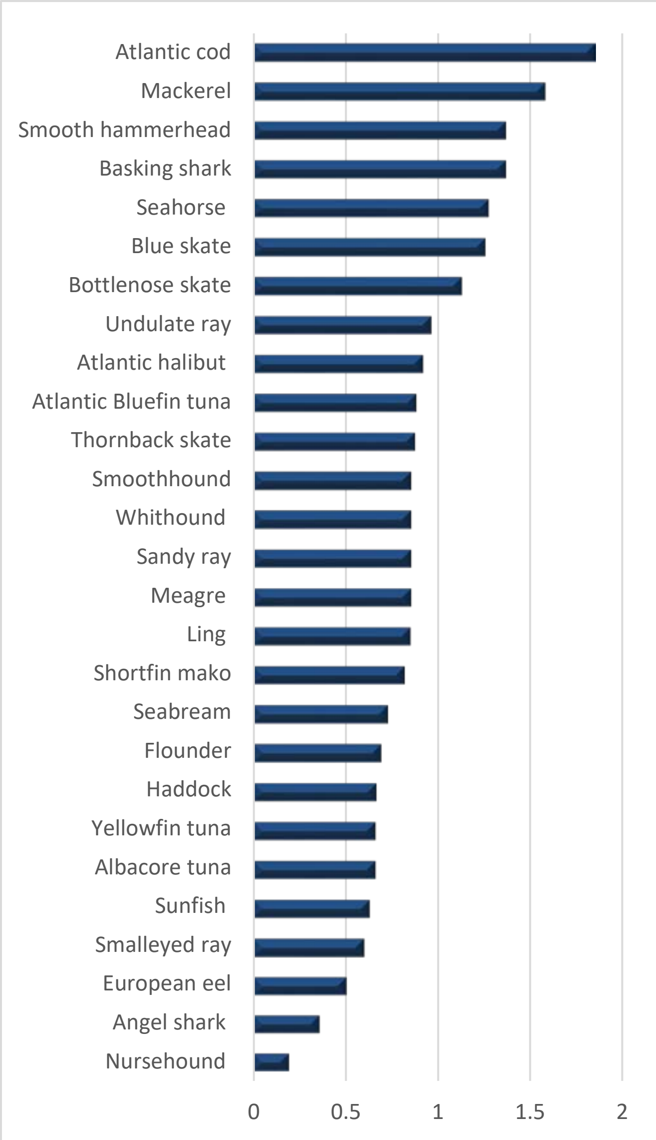


Figure 2.5. Results of PI when normalising factors using the maximum-recorded score instead of maximum-possible score.

2.5 Discussion

The Prioritisation Index (PI), as derived in this study, permits the ranking and prioritisation of UK marine fish in terms of their value to society and the ecosystem, combined with their potential susceptibility to noise pollution. The Prioritisation Index was designed as a means to aid mitigation efforts by providing a better understanding of the potential vulnerabilities of species and ecosystems to noise impacts. The higher the species scores on the Prioritisation Index, the greater the potential for direct impact of noise pollution on the species (i.e. physical injury) and indirect impacts (i.e. decline in ecosystem engineering) on their surrounding ecosystems. The work is based on the assumption that all fish hear noise and react negatively to it, and although this may not always be the case, such frameworks need to be precautionary to take account of any uncertainty and orientate around worst case scenarios (Scotland and Northern Ireland Forum for Environmental Research, 2006). The precautionary principle is applied during impact assessments when there is good reason to believe harmful effects may occur to organisms or the environment, or when uncertainties exist that prevent risk being assessed with enough confidence to inform decision-making. This study by no means generates a wholly definitive list, but offers a way of directing research or attention towards species most in need of mitigation measures, and remains flexible enough to be of use in other scenarios.

The results of this study indicate that for UK waters Atlantic cod should be the priority in terms of noise impact research (Figure 2.4). Their high value score, coupled with the likelihood that they encounter multiple noise sources and have specialist hearing apparatus that allows them to hear a wider range of frequencies, means Atlantic cod should be highly prioritised for noise pollution research. A marked fall in the numbers of Atlantic cod would have both a serious impact on the fishing industry and the general ecosystem through the loss of their contribution to ecosystem services. The meta-analysis in Chapter 1 showed only 6 of 121 studies have focused on cod so far, all of which saw significant impacts of noise on behaviour and / or physiology (Enger, 1981; Ona and Godø, 1990; Engås *et al.*, 1996; Kastelein *et al.*, 2008; Thomsen *et al.*, 2012; Nedelec *et al.*, 2015). These studies covered 4 different sound sources in various environments showing that cod are vulnerable to a wide range of sources, frequencies and intensities (*see* Table A1.1 for full study details).

Uncertainties surrounding the Prioritisation Index can be alleviated by collecting data only from peer-reviewed journals and avoiding other reports that have not been rigorously critiqued. When information (such as IUCN status) was lacking in the public databases, the data had to be collected from published works, which is a time-consuming method. This emphasises the need for public databases and publication of raw data. Scientific data capture and re-use are occurring in the 'big science' fields (i.e. physics and genomics) but the smaller areas of science, especially those using technology to record data such as marine noise, are less likely to have digital libraries available to share data (Wallis *et al.*, 2010). Freedom of access to information would aid scientific research, and the creation of desk-based models such as this one, in terms of time, accuracy, and cost. The information on noise sources used in the susceptibility category was taken from peer reviewed journals investigating the impacts of certain noise sources on marine species, but for greater accuracy, when an index such as this is used for a specific location, CIA or EIA, then the noise sources in the area should be recorded in the field to make the category scores more relevant to the report. To allow ocean noise data to be more accessible, a public library of noise sources and their specifications would prove beneficial. Such digital libraries are starting to be built (e.g. SONIC Ship Underwater Radiates Noise Database), but they are still in the early stages of development and need more data input before they can be used as an evidence base for research and impact studies.

Many fish species differ in hearing sensitivity within different bandwidths, and so differing hearing capabilities need to be taken into account in any future noise impact prioritisation frameworks. This would involve conducting audiograms for each of the species appearing in the index and allocating a hearing capability score. However, the variation between individuals in hearing sensitivity, particularly at the margins, means that extrapolating from an average to predict the effect on an individual, or determining an average from too few measurements, are potential hazards for the researcher using this approach (Leighton, 2016, 2017). The likelihood of impact from the noise sources would alter accordingly, as fish with poor hearing in a particular frequency band would be less affected by the noise than those with better hearing in the same band. Fish species with a swim bladder (and other gas chambers) have a greater potential to suffer from physiological trauma than those without such physiology, as sudden pressure changes and motions of small bubbles from noise has the potential to cause rapid movement of the walls of these chambers resulting in damage to nearby tissues, such as the kidney and gonads, and the circulatory system (Popper *et al.*, 2014). Audiogram information was included in this study but only for 27 of the 52 species considered in the PI as hearing thresholds of the remaining species have yet to be tested; only 78 fish species in total appear to have been tested to produce audiograms (with published results in peer-reviewed journals). Species included without available audiograms were assumed to hear between 0 – 100 Hz as the majority of species with available audiograms were able to hear within this bandwidth. As previously stated, all species and individuals can vary in hearing ability, and so audiogram data can never provide exact measurements of hearing sensitivity, but they should still be conducted so that more species thresholds can be included in risk and vulnerability assessment tools. Also, as technology and acoustic experimental design improves the concerns surrounding audiogram data will begin to decrease. Methodological advancements are ongoing and as more accurate audiograms are generated and further species studied, more accurate thresholds can be included. This is also true for the impact thresholds (Table 2.8); as future research provides evidence of the levels of noise responsible for physiological and behavioural changes so the PI will better reflect species overall susceptibility.

Propagation of noise varies depending on the frequency of the noise source and the characteristics of the environment, such as bathymetry, sediment type and the sound speed profile. Incorporating propagation into the risk matrix in the future would provide greater accuracy for the likelihood of exposure of a species to the source. For example, as high frequency noise propagates further in shallow water than low frequency noise, the likelihood of exposure to vessel noise is lower than the likelihood of exposure to sonar in shallow areas (Amoser and Ladich, 2005).

Future fisheries prioritisation indices could be improved by adding tonnage data from a number of years as it would provide vital information with regard to lost commercial species. For example, the number of Basking sharks caught in Norway in 1967 was 8,800 whereas in 2011 only 2 were caught (Food and Agriculture Organization, 2015) so the species scored low on commercial importance purely because overfishing in the past has severely depleted the population resulting in legislative protection (OSPAR Commission, 2009a). Although the current population condition was accounted for within the Index it would be advantageous to incorporate past population data to reflect the depletion of numbers due to overfishing. Many species may still be considered commercial and actively fished but the small landing numbers would not represent this, and their status could be misrepresented in the Index.

It has been suggested that juvenile fish and larvae are often more vulnerable to harm than adults (Holtby *et al.*, 1992; Radford *et al.*, 2010). To overcome the varying sensitivity of different life stages, each life stage of fish can be incorporated into the matrix separately if there are enough data available. This may not accurately assess the underlying reality; if juveniles are harmed the survival of the future population is threatened, resulting in the conclusion that juveniles and larvae should be prioritised higher so that they reach reproductive maturity. This is currently not reflected in the Index. Furthermore, fish audiograms have rarely been conducted on juvenile fish, so the juvenile hearing sensitivity of many species is still unknown. Juveniles may have different auditory thresholds to mature adults (as has been observed in marine mammals), and size-related differences in auditory thresholds have been observed in fish (Egner and Mann, 2005; Greenhow *et al.*, 2014). Once audiograms of juveniles have been conducted and are freely available, the Index can incorporate the information to good effect.

Although legislators and those working towards successful mitigation are starting to include the concept of ecosystem services in their strategies, it is still overlooked by some as there is no simple or established way of integrating the ecosystem services concept into decision-making processes (Hauck *et al.*, 2013). It is important to include ecosystem services provision in prioritisation for mitigation as it conveys the benefits of ecosystem conservation to diverse stakeholder groups by providing an anthropocentric justification for conserving species and ecosystems, based on our dependence on the goods and services they provide (Reid, 2006; Lamarque *et al.*, 2011). Appealing to the human benefits of mitigation could help promote incentive. The Prioritisation Index provides a method of valuing and including ecosystem services in impact assessments and prioritisation decisions. Ecosystem services in this index included keystone species which were weighted equally with all the other ecosystem services. In some ecosystems keystone species play a large role and so it may be beneficial to weight them more heavily than other ecosystem services in some cases. This ability to change and alter the parameters used ensures that this index will prove a useful flexible tool in the future for researchers and decision makers.

The framework used here eliminates any misleading conclusion that may occur when only a single factor is considered. Whilst it is undeniably important to focus research on endangered species vulnerable to noise pollution, it is also necessary to direct research and protection towards species of significant commercial and economic importance. Multiple factors must be considered and accounted for if an accurate picture of the vulnerabilities of a species to noise pollution is to be determined. Estimating the potential vulnerability of a species to noise using just one factor ignores that a species might be more affected in a different but equally weighted factor (Table 2.9). For example, if choosing to focus on species that are only of commercial value, then seahorses, a highly protected UK species, would not be considered at all. Similarly, mackerel, a species considered of least concern on the IUCN Red List that many would not consider as needing any population management, was highly ranked on the Prioritisation Index, as it could have serious repercussions on the ecosystem and fishing industry if it suffered a population decline. If a future index included benthic species then it would be of use to consider not only the population condition and direct impact of noise on the species, but also the contribution made by a given species to providing a healthy environment in which other species might flourish, as there is evidence that key processes in mediating benthic nutrient cycling can be impacted by noise (Solan *et al.*, 2016). The overall priority ranking is useful to identify the need for understanding noise pollution impacts of certain species, however, by just analysing the overall results of all factors can prevent subtle outcomes of the analysis to be overlooked. Two species with almost identical scores could be ranked overall in the same position, but the reason for their ranking may be very different; one species may have scored higher on the susceptibility indicator and, therefore, mitigation should be focused on minimising exposure to noise, whereas the other may have scored higher on the value indicator and so mitigation would need to focus on reducing further population decline. The framework used to derive the Prioritisation Index allows individual factors to be considered as it produces ranked lists of each factor during the process, but additionally lets the user look across all influencing factors. Using a risk assessment alone would show the likely impact of noise, but would fail to take account of the value of the species to both the ecosystem and society. A combination of categories provides for a more comprehensive analysis.

The PI provides both a relative and absolute method of prioritising species. A species rank, or prioritisation score can be used, depending on whether absolute or relative terms are needed. The ranking provides a priority position relative to the other species in the study. To compare priorities between studies or locations the prioritisation scores would need to be used as an absolute score. Reporting the full table of scores from all factors allows easier analysis and comparisons to be made. The Prioritisation Index developed here is an ideal tool for such a purpose as it is flexible and capable of handling any number of categories with provision for weighting individual factors according to the needs of the research. These factors allow the tool to provide meaningful results that are useful in decision-making processes.

A major obstacle in noise pollution research is that results cannot be generalised between species because of the uniqueness of each species in terms of hearing apparatus, sensitivity thresholds and response to noise. As no two species shared the same scores for all categories (Table 2.9), this study emphasises how unique each fish species is in all aspects, including their degree of vulnerability. Individual species scores and those of the underlying influencing factors pinpoint the most likely threats faced by a species and indicate where conservation efforts should be directed for maximum benefit for that species. A high score at any level within the framework should be investigated to help prevent any further damage to the wellbeing of the species. When multiple threats in varying degrees are present, the PI helps to identify the factors and relationships that combine to form the overall risks from the impacts of noise pollution and thus allow more meaningful assessments to be produced. Access to this type of data is particularly useful in the development of conservation programmes, and to enable more informed policy and legislative decisions to be made. For example, if a species' highest score was shown to be habitat susceptibility, then focusing conservation efforts on improving the habitat would likely prove most beneficial. Likewise, if commercial importance plays a large part in the overall vulnerability score, then fishing quotas could be altered or introduced to improve the chances of survival and to enhance population density. Atlantic Bluefin tuna (ranked 14th) scored low in all categories for this UK case study, but if the Index was European wide instead of UK specific, then Atlantic

Bluefin tuna would have scored highest commercially ranking them second overall. As multiple threats are often present in varying degrees, the index helps identify the factors and relationships that combine to form the overall risks from the impacts of noise pollution. This allows for more meaningful impact assessments to be produced.

This study focused on commercial species caught by the UK fishing fleets along with well-known keystone species and some endangered species. However, there are many more species in UK waters that are likely to be at risk from noise pollution and which could be included in any future matrices. The principles within this framework can be adapted and applied either in a narrower or in a broader sense: at different spatial scales, i.e. for regional conservation efforts; for species-specific projects; as the basis for species case studies to identify risk at different life stages; to cover non-marine species; or any other given specific reason for assessment. The data input into the Index will vary depending on the specific question or assessment the framework is being used to address. Category scores may be considered independently, or the overall score could provide a wider picture. Developing a case-by-case understanding of how a threat will respond to different conservation actions will allow planners to anticipate both the positive and negative consequences of each action, thereby making conservation planning more effective (Wilson *et al.*, 2005).

It is important to note that this framework does not replace the need for expert opinion. In this case study all goods/benefits listed under the ecosystem services provision factor were assumed to be of equal importance and therefore equally weighted. However, if in terms of the specific species or location under evaluation, one ecosystem service is considered of more value than another, the weighting can be adjusted. Expert opinion will be needed to apply such weightings. The influencing factors in this Index are all weighted equally, implying that no one factor has more influence than any other, but this may not always be considered appropriate. The weightings can be easily altered to better reflect reality, but this can lead to bias; a conservationist may apply different weightings to a construction manager in order to influence the species ranking in favour of their own project or viewpoint. Equal weighting negates such bias. Relative weight analysis was used to ensure equal weightings were appropriate for this case study. Such statistical methods should be used by other researchers following the framework to evidence their chosen weightings and remove any bias they may accidentally introduce.

Noise pollution research has, to date, primarily focused on a narrow range of relatively few species. Seabass, for instance, has been the subject of several studies of noise pollution research (e.g. Kastelein *et al.*, 2008; Everley, 2013; Debusschere *et al.*, 2014; Neo *et al.*, 2014, 2015, 2016;

Bruintjes *et al.*, 2016; Radford *et al.*, 2016a; Herbert-read *et al.*, 2017), as has herring (e.g. Vabø *et al.*, 2002; Mitson and Knudsen, 2003; Slotte *et al.*, 2004; Hjellvik *et al.*, 2008; Kastelein *et al.*, 2008; Dokseater, 2009; Doksæter *et al.*, 2012; Sivle *et al.*, 2012, 2013; Handegard *et al.*, 2016). However, based on the outcomes of the present study, these species' potential vulnerability from the impacts of noise pollution are in fact rather low (Figure 2.4). Of the studies analysed during Chapter 1, no study used both value and susceptibility as reasons for choice. This demonstrates the value of the PI in providing a semi-quantitative assessment that effectively highlights why some species should be chosen over others as the focus for research and conservation efforts. It legitimises species choice. Priority species in the geographic area, or the life cycle of interest can be focused on. The prioritisation of species is likely to change and differ depending on the various factors under consideration. Ideally, the index would be expanded to include more considerations, such as the seasonality and spatial extent of spawning areas that are important factors known to influence population levels. However, such information is still limited for many species and it would add a further level of complexity to any assessment tool.

The proposed method of prioritisation assessment via a risk-based matrix can be thought of as a useful additional approach to ecological vulnerability and impact assessments. The use of this modelling tool for prioritisation analysis has several advantages: all information input into the PI was obtained from literature and public databases; results are ecologically meaningful as all information used was based on wild fish and direct noise recordings; and no laboratory fish conditioned to captivity and unrealistic environments or estimations of fish communities were involved. It improves upon previous approaches by including more variables (such as the commercial importance of the species, their environment, and degree of exposure to risk) to calculate the potential vulnerability to threat. The broader ability of this framework to combine multiple factors will add value to Cumulative Impact Assessments.

2.6 Conclusions

Good ecosystem management requires a reliable standard of measurement, and conservation at all trophic levels may indeed benefit from a systematic, reproducible and well-defined method of ranking threats to marine ecosystems (Halpern *et al.*, 2007; Micheli *et al.*, 2013; Tett *et al.*, 2013). The Prioritisation Index framework does this in relation to noise pollution impacts. This study demonstrates how a risk-based matrix could be used as a valuable tool in vulnerability assessment and ecosystem risk management, and how it could provide relevant knowledge for use in impact assessments, conservation actions, and in the development of mitigation. There are still gaps in our knowledge regarding the understanding of marine ecosystems and their responses to human activities, but the Prioritisation Index results helps to pinpoint where more information and research is needed in relation to noise pollution impacts. By considering a wide range of criteria during the process it ensures a more complete and targeted approach to decision-making and can help organisations prioritise how to allocate their limited time, money and resources for the most beneficial results.

A systematic, replicable and well-defined method of ranking threats to marine ecosystems has been demonstrated in this chapter through the creation of a prioritisation method that can be used to rank species vulnerability to noise impacts based on two main indicators and their influencing factors. Species are assessed on their likelihood of exposure to noise and severity of harm should they become exposed. The framework presented provides a repeatable and semi-quantitative method to demonstrate why one species should be chosen over another for mitigation. Previous works have chosen species either for convenience or to test one known factor. Cumulative factors or overall vulnerabilities have not been considered, and the results of independent experiments, although useful, cannot be applied meaningfully across a wider range of species. It is hoped that this framework will change perceptions on species choice, and that the initial selection of focal species will receive more thought and reasoning than before when embarking on research.

3. Chapter Three: Creation of an ocean noise map: using AIS data to model vessel noise emissions

The increasing number of research and management efforts to combat marine noise pollution has led to a greater awareness of the long-term chronic effects of ocean noise, and the larger scale changes in underwater soundscapes (Gedamke *et al.*, 2016b). Underwater noise from shipping is increasingly recognised as a significant and pervasive pollutant with the potential to impact marine ecosystems on a global scale (Williams *et al.*, 2015a). The highest intensities of vessel emissions often fall within frequencies ranging from 100 – 1,000 Hz (McDonald *et al.*, 2014), which is a frequency range of greatest sensitivity for a number of fish (Popper and Fewtrell, 2003). As previously discussed in Chapter 1 (Section 1.4.2), decreased foraging efficiency, changes in vocalisations, and altered schooling behaviours have all been observed in response to vessel noise. Such behavioural changes have the potential to affect fish populations. Knowledge regarding the impacts of noise on marine fish has greatly increased over the past decades, but the overall picture remains incomplete, especially population level consequences of noise exposure (McGregor *et al.*, 2013). Research recommendations advise conducting studies to determine the actual status and trend of underwater noise pollution in high priority areas, defined as those areas having biologically important communities (Codarin and Picciulin, 2015). These priority areas can be identified by frameworks such as the Prioritisation Index presented in Chapter 2 as, for example, that particular index showed the UK continental shelf as a priority area for Atlantic cod. The prioritisation framework information can be used to plan for the management and conservation of marine fish species. More recent research has highlighted the importance of soundscape characterisation, modelling, and mapping (Boyd *et al.*, 2011b). Models and noise maps are seen as valuable low cost tools for generating comprehensive results, and here we offer a model-based approach to research and evaluate vessel noise source levels. To mitigate against vessel noise it is useful to know the source levels of ships and how the noise emissions can propagate and impact on fish species. Modelling, as a method for mitigation, provides information that can be used to make recommendations for future mitigation measures. For example, it is suggested that models based on planned increases in vessel movements may be able to forecast associated increases in noise exposure through identifying trends (Lusseau *et al.*, 2011; New *et al.*, 2013). Predictions of future trends in noise exposure can provide decision-makers with information to underpin damage limitation strategies which can be implemented to combat potential harm to marine fish species. In this chapter, an innovative model to evaluate vessel noise pollution in waters surrounding the UK at a large temporal (yearly) and spatial scale (hundreds of kilometres) using Automated Identification System (AIS) and online vessel data is presented.

3.1 Ocean noise models

Concern about noise impacts on fish has primarily focused on the potential acute effects of noise exposure leading to immediate consequences, such as direct physical harm. Studies showing short-term behavioural and physiological impacts of noise are numerous (Nedelec *et al.*, 2017b), with at least 68 studies reporting short-term effects in the literature since 2000 (e.g. Nichols *et al.*, 2015; Johansson *et al.*, 2016; Holmes *et al.*, 2017; Magnhagen *et al.*, 2017). In Hawkins *et al.'s* (2015) publication on 'Information gaps in understanding the effects of noise on fishes and invertebrates', the authors identified 'scientific programmes that monitor trends in soundscapes through the acquisition of long-term data sets with immediate emphasis in areas of future change and/or critical habitat' as a gap in the knowledge-base. Other studies have called for areas where there is a high prevalence of shipping traffic to be monitored so that any impacts occurring on marine species can be addressed and mitigated for (Merchant *et al.*, 2012; Williams *et al.*, 2015b). There is, therefore, a need to identify areas of high exposure risk within the marine environment and develop methods to assess long-term noise exposure and trends (Erbe *et al.*, 2012, 2014; Merchant *et al.*, 2012). Vessels are the most widespread source of anthropogenic noise in the oceans, contributing to the accumulation of low frequency noise (Arveson and Vendittis, 2000). Large commercial vessels are the main culprits for vessel noise emissions, with source levels ranging from about 150 dB to over 190 dB re 1 μPa at 1 m (Arveson and Vendittis, 2000; Hildebrand, 2009; McKenna *et al.*, 2012), with the highest intensities usually falling within frequencies ranging from 100 to 1,000 Hz (McDonald *et al.*, 2014). The majority of audiograms of marine fish species indicate that their greatest sensitivity to noise falls within this range (Popper *et al.* 2003), suggesting vessel noise exposure could have a negative impact on a variety of fish species. Exposure to shipping noise is responsible for impacts such as alterations in heart rate, increased cortisol secretion, unusual swimming behaviours and reduced foraging behaviour to name a few (Sarà *et al.*, 2007; Spiga *et al.*, 2012b; De Robertis and Handegard, 2013; Magnhagen *et al.*, 2017). However, there has been a developing focus on the much larger scale and longer term chronic effects of increases in ocean noise and the subsequent changes in underwater soundscapes (Gedamke *et al.*, 2016b). Longer-term experiments conducted over broader spatial scales may offer a more complete understanding of the population-level and interacting effects of noise on wildlife (Shannon *et al.*, 2015), but few studies to date have attempted to explore the effects of noise at a large geographic or long temporal scale, with the exception of Erbe *et al.* (2012) who studied the coast of Seattle for one year. As mentioned previously, large scale research can be difficult to conduct owing to constraints on time and financial resources. The importance of soundscape characterisation, modelling, and mapping as a means of identifying the potential long-term effects of noise over large spatial scales has been highlighted in recent years (Boyd *et al.*, 2011a).

Pollutant trends can be analysed using map-based tools; mapping the density, concentration and dispersal of pollutants over time or geographic location can identify trends. As part of the Marine Strategy Framework Directive (MSFD), 2008/56/EC Member States must achieve and maintain Good Environmental Status (GES) in their Seas. There are two indicators for underwater noise used to describe the GES, one of which (Indicator 11.2.1) focuses on the issue of chronic exposure to low frequency noise, with the main contributor given as commercial shipping noise (van der Graaf *et al.*, 2012). The criteria to assess low frequency continuous noise is to monitor the ambient noise level trend within the 1/3 octave bands 63 Hz and 125 Hz (centre frequency), which should not exceed 100 dB re 1 µPa rms (average noise level in these octave bands over a year) (HM Government, 2012). This indicator is intended for monitoring trends in noise levels; unfortunately, historical measurements of underwater sea ambient noise are available for only some of the European waters to date (Codarin and Picciulin, 2015). Modelling of underwater noise levels using Global Positioning System (GPS) data from tracked vessels has been proposed as a way to map noise exposure from shipping to facilitate targeted mitigation measures (Erbe *et al.*, 2012). Such data, known as Automatic Identification Systems (AIS) data, is transmitted from ships and stored in databases by a number of companies. AIS-based modelling means estimations of vessel source levels, and exposure maps, can be produced for past years covering any number of AIS transmitted locations. Modelling using historical data highlights any knowledge gaps and shows past trends. Decision makers can use this information to extrapolate the data to aid future mitigation measures.

3.2 Tracking ships to analyse pollution

Automatic Identification Systems (AIS) provides an electronic means for ships to broadcast ship data at regular intervals including: vessel identification, Global Positioning System (GPS) position, course, and speed. Information is transmitted continuously, providing a comprehensive and detailed data set for individual vessels which can be used to estimate and allocate emissions based on improved traffic pattern data (Perez *et al.*, 2009). It provides a spatial representation of vessel movements within the receiving range of transmissions. Under the International Maritime Organization's (IMO) mandates, all ocean-going commercial vessels of over 300 gross tonnes or carrying more than 165 passengers, as well as all tug/tows, are required to carry AIS transmitters (Federal Register, 2003; International Association of Marine Aids to Navigation and Lighthouse Authorities, 2004). Also, many vessels not matching the IMO criteria, voluntarily use AIS transceivers in case of accidents. AIS transmissions can be received via terrestrial or satellite receivers. For further information on AIS systems see Neenan *et al.* (2016) and Shelmerdine (2015).

One of the primary requirements in assessments of potential impacts of noise on marine life is the estimation of received levels at different locations where the targeted species are of concern (Spiga, 2015). Modelling of underwater noise levels using AIS data has been proposed as a way to map received level noise exposure from shipping to facilitate targeted mitigation measures (Erbe *et al.*, 2012b).

3.2.1 Terrestrial AIS

A network of terrestrial receivers is run and maintained throughout the UK providing continual listening and observation of vessel traffic. The transmission range of the receivers can vary from as little as 20nm to 350nm depending on the atmospheric conditions (Associated British Ports Marine Environmental Research, 2014).

3.2.2 Satellite AIS

Satellites in Low Earth Orbit, at an altitude of between 650 and 850 km above the earth, are capable of detecting AIS signals (Associated British Ports Marine Environmental Research, 2014). Satellites are able to collect AIS data from ships further from shore, whose transmissions would be out of the range of terrestrial AIS receivers. However, as the Marine Scotland (2014) report stated, there are still limitations to satellite receivers:

- A satellite's coverage range has a limit and so multiple satellites are needed to cover an entire vessel transit line; each satellite has a maximum listening time of 12 minutes before a ship moves out of its range. Therefore, it is possible for a vessel to not have satellite coverage for the whole of its voyage.

- Owing to the nature of the data transmitted, satellite AIS provides a greater number of broken transit lines when a vessel manoeuvres around landmasses (for example around islands). Gaps in the transit line can be predicted but can lead to inaccurate assumptions.

- Satellite receivers are prone to data collision problems due to the large amounts of data they receive at any one time. When data collision occurs a proportion of the transmitted messages are lost.

- Large volumes of data messages must be stored on-board the satellites until the ground link stations come into range of the satellites orbit and the data can be re-transmitted. As the memory for data has a limited capacity some data can be lost by the system before it can be re-transmitted.

- The reception field of view directly underneath the satellite is compromised resulting in a small hole in the middle of the 5,000 km field of view.

- Terrain can significantly restrict the signal path. High terrain is likely to block the signal path and prevent the satellite receiving the message, for instance, when vessels are travelling along a fjord coastline.

Using satellite and terrestrial AIS data together will increase the area of ocean included in the dataset and provide more accurate transmissions both close to land and in the open ocean.

There are two types of Automatic Identification Systems found on vessels. AIS-A provides characterisation of commercial shipping but misses the bulk of non-AIS vessels, including: commercial vessels below 300 Gross Tonnes, recreational vessels, fishing vessels, and Military/Government vessels whilst on deployment. AIS-B is a non-mandatory form of AIS typically used by small commercial craft, fishing vessels and recreational vessels. AIS-B includes Vessel Monitoring System (VMS) used primarily by fishing vessels.

3.3 AIS modelling

The use of physical or numerical models is a growing area in noise pollution research, with models having been developed for several common noise sources, including pile driving (e.g. Fricke and Rolfes, 2015), seismic airguns (e.g. MacGillivray, 2006), and shipping (e.g. Wales and Heitmeyer, 2002). A combination of modelling and map-based tools can be used to study the impacts of wide-spread pollutants such as vessel noise through long term measuring of noise or through Automatic Identification System (AIS) models.

It is suggested that models based on planned increases in vessel movements may be able to forecast associated increases in noise exposure through identifying trends (Lusseau *et al.*, 2011; New *et al.*, 2013), providing a promising indication that AIS-based noise mapping could be successfully applied to target vessel noise mitigation efforts (Merchant *et al.*, 2014a). Such maps (e,g, Erbe *et al.*, 2012, 2014) would help to identify areas of greatest concern for the conservation of acoustically sensitive species, provided they can be adequately validated (Merchant *et al.*, 2015).

There is doubt about the efficacy of such an approach as only certain vessels carry operational AIS transmitters (Merchant *et al.*, 2016). Yet, it has been demonstrated (in the Sutors, Moray Firth, Scotland) that noise emissions generated by AIS-carrying vessels were generally greater than those produced by non-AIS vessels for frequency bands 100–10,000 Hz, and most noise emissions were attributable to the vessels operating with AIS transceivers (Merchant *et al.*, 2016). Williams et al (2015b) claimed that AIS tracked vessels effectively contributed all of the noise exposure in the frequency range 20 – 1,000 Hz. Noise models using AIS data should account, therefore, for most of the noise emissions present, assuming that the source levels and propagation models used are sufficiently accurate and provide accurate emission predictions. Coomber *et al.* (2016) and Shelmerdine (2015) have demonstrated that AIS data processing times are not time exhaustive and that the benefits of the highly detailed data are incredibly useful in shipping management, and so vessel noise mitigation management could also benefit from AIS modelling methods. Such models can be applied to predict shipping noise levels under various scenarios and suggest areas in need of mitigation.

3.4 Previous models

Previous studies that have used AIS data to analyse marine noise pollution (e.g. Associated British Ports Marine Environmental Research, 2014; Erbe *et al.*, 2014; Merchant *et al.*, 2014b; MMO, 2014, 2015b; Viola *et al.*, 2017) have concluded that it is a useful mechanism for predicting noise pollution. However, studies that use AIS data to analyse ship noise emissions to date have been short-term (covering just a few months) and over a small geographic area (e.g. Viola *et al.*, 2017). Merchant *et al.* (2014b) used AIS data to identify ship noise impacts in the Moray Firth over a four-month period; Erbe *et al.* (2014) used AIS data to assess ship noise hotspots in British Columbia's waters from June to September in 2008. The Marine Scotland (2014) study modelled a 7-day period but recognised that this was too short a time to provide reliable information on patterns of ocean use over a month. Another complication of vessel noise mapping models is the accuracy of the source level. The most important factor to reduce uncertainty in noise exposure predictions is the sound level of the noise source (Farcas *et al.*, 2016). Previous authors have recorded other caveats of AIS mapping such as low ship location and noise emission accuracy. Other past AIS-based vessel noise models have used a density grid – the average number of ships present in a map grid cell over a specific period - to estimate vessel positions and noise emissions (e.g. Associated British Ports Marine Environmental Research, 2014; MMO, 2014, 2015 used 2 km by 2 km density grids). The density grid method assumes that noise sources within a cell occur independently of one another. However, using the original AIS vessel transit line information rather than the density grid would provide the exact timing of vessel noise sources. Using precise location data provides a more accurate map and considers the cumulative impact of two or more noise sources. The Marine Management Organisation (MMO) (2015) used source levels taken from previous literature; the specific source levels of the individual ships present in the AIS data were not used, reducing the integrity of the noise map. Having built their AIS-based vessel noise map in Geographical Information System (GIS) software, the MMO (2015) then expressed the view that until variation in source levels can be modelled accurately and predicted, researchers should not expend the necessary significant time and effort to refine the use of GIS tools in ocean noise mapping.

3.5 Study aim

One of the primary requirements in assessments of potential impacts of noise on marine life is the estimation of received levels at different locations where the targeted species are of concern (Spiga, 2015), so a method of predicting received noise levels from vessels is needed. Also required, are scientific programmes that monitor trends in soundscapes through the acquisition of long-term data sets with immediate emphasis in areas of future change and/or critical habitat (Hawkins *et al.*, 2015). The aim of this research is to build an accurate ocean vessel noise exposure map using Automatic Information Systems (AIS) and online vessel data to quantify specific source level noise emissions from shipping in waters surrounding the UK, and provide useful visual outputs of the noise emissions over larger geographic and temporal scales than previously attempted. This study acknowledges the caveats mentioned in previous works and has broadened the study by designing a method that can analyse an entire year of AIS data to map cumulative vessel noise exposure throughout that whole year. This means that no predictions or generalisations in terms of vessel positions need to be made, and seasonal variations in both vessel movements and noise exposure levels are accounted for in the data. Instead of a local study site, the data used here encapsulates the whole UK, and provides maps of the North Atlantic, North Sea and Baltic Sea. Farcas *et al.* (2016) stated that the most important factor to reduce uncertainty in noise exposure predictions is the sound level of the noise source, and so this model calculates the Source Level (SL), rather than estimating SLs based on ship type as previous studies have done (e.g. MMO, 2015). This work addresses the question of unknown or estimated source levels through the application of an equation based on work by Wales and

Heitmeyer (2002) and the SONIC Report (Colin *et al.*, 2015) that can predict the source level from an individual ship at any transmission point during its voyage. The model will also improve on previous AIS-based map propagation methods by calculating propagation from each point using more complex methods accounting for bathymetry and sediment of the ocean floor, resulting in a noise exposure level for each 1 km grid cell throughout the study area.

The improved methods and techniques mentioned above are only beneficial if high quality data is used for the study. This chapter uses data purchased from Orbcomm (Rochelle Park, New Jersey, United States). The Confidence Criteria cited in Marine Scotland's (2014) report (Table 3.1) was applied to the dataset and the overall confidence of the data can be rated as high (Table 3.2).

Table 3.1. Confidence scores taken from Marine Scotland's (2014) report

Score	Description
0	Unable to assess from the information provided
1	Low confidence in the information provided
2	Moderate confidence in the information provided
3	High confidence in the information provided

Table 3.2. Confidence of the data used in this chapter based on the criteria from Marine Scotland (2014). Confidence scores are provided in Table 3.1.

Confidence Criteria	Confidence Notes	Level of confidence
Methodology	Data processing and analyses methods are well documented and are adapted from previously published studies (Associated British Ports Marine Environmental Research, 2014; Marine Management Organisation, 2014; Coello *et al.*, 2015). The more advance methodology used will improve understanding of vessel movements and their noise emissions.	3
Timeliness	The data covers an entire year from 01/01/2013 – 31/12/2013, and the area and time period covered is the same for both the AIS-A and AIS-B and satellite and terrestrial data.	3
Spatial	The combined use of terrestrial and satellite AIS data allows for more accurate positions, and all areas of the ocean (Figure A3.1) to be recorded.	3
Completeness	Data consists of 12 months of data covering the entire study area. However, it is not possible to be completely confident in the completeness of the data as there is no way to know if every single vessel transmitted every single journey in 2013. Confidence level was lowered to account for any possible technical failures of the transmitters or receivers.	2

3.6 Method

3.6.1 Conceptual model

Noise emissions from marine shipping were calculated by applying a bottom-up activity-based methodology using AIS data to derive vessel activity and noise emissions. The model used Java programming language and the PostgreSQL database management system to produce a cumulative map of ocean shipping noise emissions (Figure 3.1). Technical build information can be found in Appendix 3 (Section 10.1). The data was processed in several steps, from raw AIS data to visual map outputs (Figure 3.2).

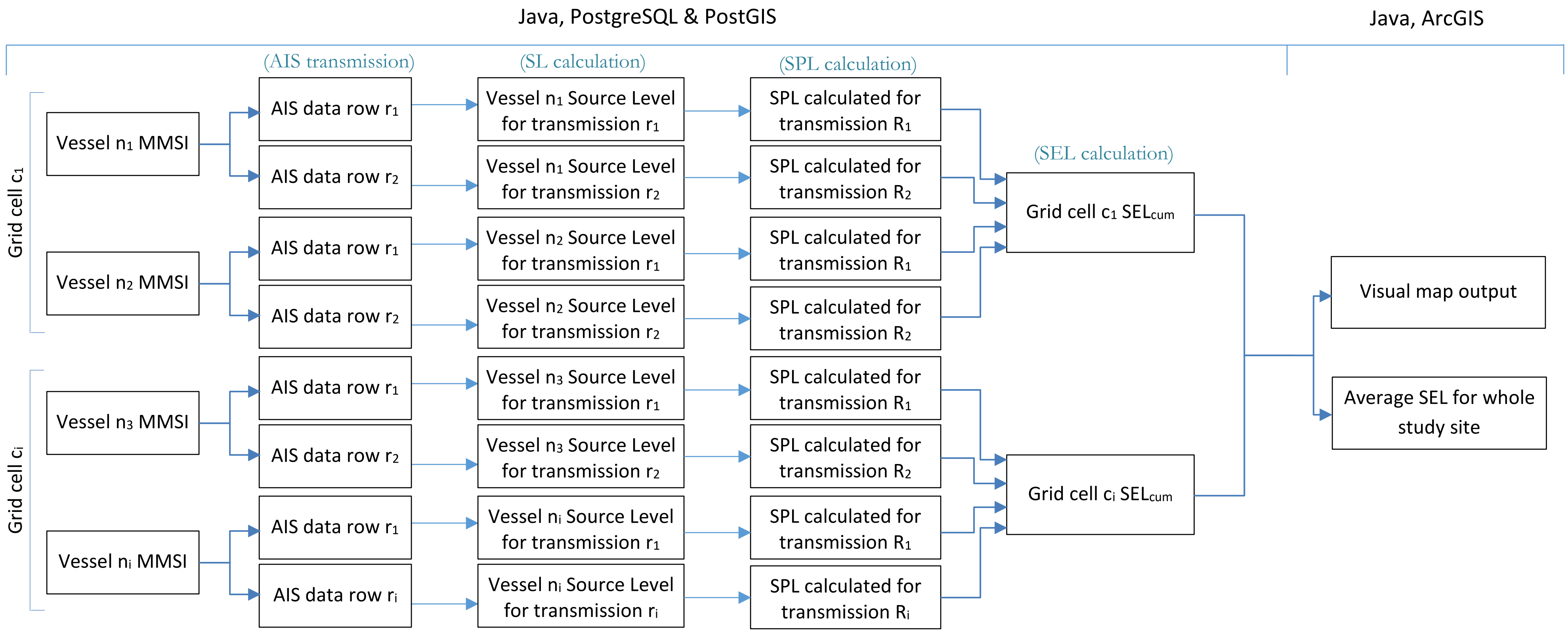


Figure 3.1. Conceptual diagram of the automatic identification system (AIS) modelling approach for estimating vessel Source Level (SL) emissions and cumulative Sound Exposure Levels (SELcum). Each vessel is associated with Maritime Mobile Service Identity (MMSI) number that is a unique identifier transmitted within the AIS data, and transmits an AIS message approximately every 12 minutes throughout its journey. A vessel can be present in only one 1 km by 1 km grid cell at a time, but can travel though many different cells over the course of the journey. Each AIS data row represents one transmission from the vessel. It is possible to transmit multiple AIS data rows in one grid cell, if the vessel is present in the cell for more than 12 minutes. Source Levels were estimated for each transmission stating the calculated SL of the vessel at that given point in time. The Sound Pressure Level (SPL) was then calculated for each transmission using the SL and propagation loss, which takes into account depth and sediment. The overall monthly grid cell exposure level (SELcum) was then calculated to provide one cumulative Sound Exposure Level for each 1km by 1 km grid cell. The dataset consisting of all cells SELcum values was then imported into ArcMap to produce a more user friendly visual output. The average SELcum for the whole month was also calculated using Java, as was the average SPL (SPLav) for the whole of 2013. PostgreSQL and local file systems were used to store the data during processing.

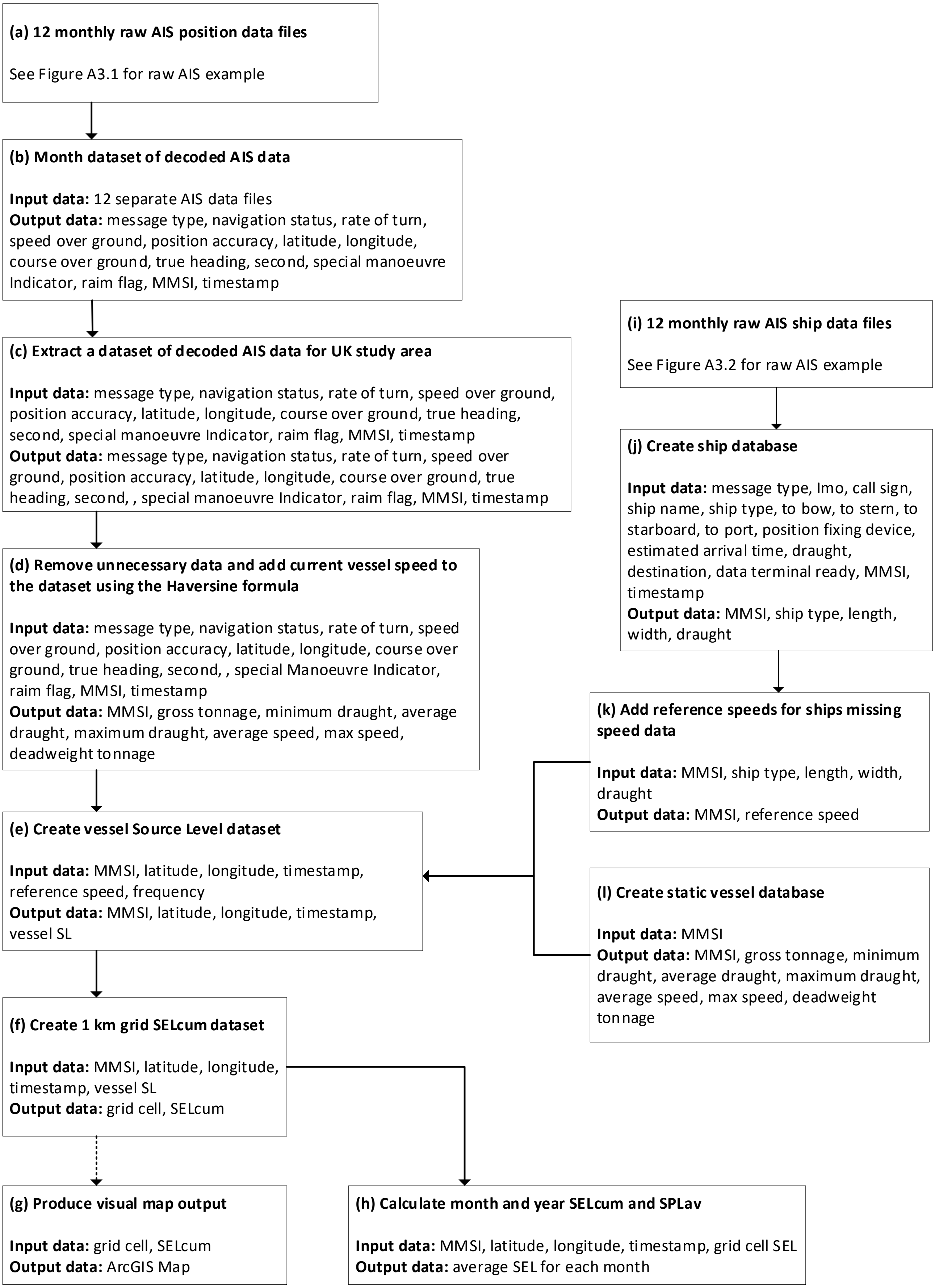

(a) 12 monthly raw AIS position data files
See Figure A3.1 for raw AIS example
(b) Month dataset of decoded AIS data
Input data: 12 separate AIS data files
Output data: message type, navigation status, rate of turn, speed over ground, position accuracy, latitude, longitude, course over ground, true heading, second, special manoeuvre Indicator, raim flag, MMSI, timestamp
(c) Extract a dataset of decoded AIS data for UK study area
Input data: message type, navigation status, rate of turn, speed over ground, position accuracy, latitude, longitude, course over ground, true heading, second, special manoeuvre Indicator, raim flag, MMSI, timestamp
Output data: message type, navigation status, rate of turn, speed over ground, position accuracy, latitude, longitude, course over ground, true heading, second, , special manoeuvre Indicator, raim flag, MMSI, timestamp
(d) Remove unnecessary data and add current vessel speed to the dataset using the Haversine formula
Input data: message type, navigation status, rate of turn, speed over ground, position accuracy, latitude, longitude, course over ground, true heading, second, , special Manoeuvre Indicator, raim flag, MMSI, timestamp
Output data: MMSI, gross tonnage, minimum draught, average draught, maximum draught, average speed, max speed, deadweight tonnage
(e) Create vessel Source Level dataset
Input data: MMSI, latitude, longitude, timestamp, reference speed, frequency
Output data: MMSI, latitude, longitude, timestamp, vessel SL
(f) Create 1 km grid SELcum dataset
Input data: MMSI, latitude, longitude, timestamp, vessel SL
Output data: grid cell, SELcum
(g) Produce visual map output
Input data: grid cell, SELcum
Output data: ArcGIS Map
(h) Calculate month and year SELcum and SPLav
Input data: MMSI, latitude, longitude, timestamp, grid cell SEL
Output data: average SEL for each month
(i) 12 monthly raw AIS ship data files
See Figure A3.2 for raw AIS example
(j) Create ship database
Input data: message type, Imo, call sign, ship name, ship type, to bow, to stern, to starboard, to port, position fixing device, estimated arrival time, draught, destination, data terminal ready, MMSI, timestamp
Output data: MMSI, ship type, length, width, draught
(k) Add reference speeds for ships missing speed data
Input data: MMSI, ship type, length, width, draught
Output data: MMSI, reference speed
(l) Create static vessel database
Input data: MMSI
Output data: MMSI, gross tonnage, minimum draught, average draught, maximum draught, average speed, max speed, deadweight tonnage

Figure 3.2. Steps of the Automatic Identification Systems (AIS) model to create vessel noise exposure maps. The Marine Mobile Service Identity (MMSI) was used as a unique identifier for each ship throughout the process. Solid arrows indicate Java processes, whilst the dashed arrow indicates an ArcGIS process. The first stage was to decode the raw AIS data into a human-readable format. This was done in two steps: **(a)** cutting the unneeded characters from the AIS transmission and then **(b)** running the file through an AIS decoding programme written in Java. To make the data more manageable, it was stored in monthly files. **(c)** The UK study site was extracted from the full dataset. **(d)** Vessel speed was calculated directly from the AIS data using the Haversine formula. **(l)** To get the vessel reference speed needed to create the Source Level (SL) dataset reference speeds were collated from an online vessel database. Static vessel data was collected from https://www.fleetmon.com by writing a Java programme that used Selenium webdriver to search the website for the desired data. The programme took the MMSI numbers from the Month dataset of decoded AIS data for UK study area and searched the FleetMon website for details on the vessels, saving the details into a local static vessel database file. **(k)** For any ships not found on the website, a Java programme was used to calculate the most likely reference speed by matching the vessel to another vessel of the same type with the same dimensions. **(i)** A Java programme was used to create a database of ship data from the raw AIS file so that the dimensions of all ships were available. **(j)** The Ship Database totalled 352,337 vessels. More specific details of the vessels in the study area can be seen in Section 3.7.1. The Ship Database was created to store vessel details needed to calculate vessel reference speeds for vessels that were missing speed data in the online database (https://www.fleetmon.com). **(e)** For creating the Source Level dataset a frequency needed to be added into the calculations. This was done manually in this step. Information on the calculations used is in Section 3.6.3. **(f)** The cumulative Sound Exposure Level (SELcum) for each month was calculated. **(g)** To provide a user-friendly output to the model, exposure maps were produced in ArcMap 10.4.1. **(h)** Monthly average Sound Pressure Levels (SPL) and cumulative SEL were calculated using a Java programme to allow comparisons between months.

3.6.2 Data information

The AIS data, provided by Orbcomm (Fort Lee, New Jersey, United States), contained both satellite and terrestrial coverage. The data incorporated all vessel types in waters surrounding the UK (within latitudes 40 ºN & 65 ºN and longitudes 20 ºW & 12 ºE) between 1st January 2013 and 31st December 2013. Over 453,000,000 rows of AIS data were used, comprised of 352,337 vessels identified by their unique Marine Mobile Service Identity (MMSI) numbers. The types of vessels included in the data are shown in Appendix 3. Raw AIS data was decoded to provide position messages at all transmission points during a voyage for each ship (Figure 3.2). AIS transmissions occurred on average every 12 minutes. Sail boats without engines were excluded from the model.

```
PositionReportClassAScheduled, UnderwayUsingEngine, 0,12.4, false, -5.98192, 110.37428, 96.3, 96, 4, NotAvailable, true, 525019095, 2013-01-01 00:00:02.0,
PositionReportClassAScheduled, UnderwayUsingEngine, -127,0.0, true, 51.469105, 5.662205, 0.0, 511, 59, NotAvailable, true, 44710000, 2013-01-01 00:00:03.0,
PositionReportClassAScheduled, UnderwayUsingEngine, 0,0.0, true, 31.331467, 121.66671, 300.5, 511, 60, NotAvailable, false, 413810919, 2013-01-01 00:00:03.0,
PositionReportClassAScheduled, UnderwayUsingEngine, 0,0.1, true, 51.879814, 4.4528832, 16.1, 285, 48, NoSpecialManeuver, false, 244700126, 2013-01-01 00:00:03.0,
PositionReportClassAScheduled, UnderwayUsingEngine, 0,5.6, true, 36.921078, 122.48088, 60.8, 511, 60, NotAvailable, false, 412200023, 2013-01-01 00:00:03.0,
PositionReportClassAScheduled, UnderwayUsingEngine, 0,0.1, false, 29.764914, 121.99763, 233.5, 252, 59, NotAvailable, false, 412428370, 2013-01-01 00:00:03.0,
PositionReportClassAScheduled, Moored, 0,0.0, true, 40.29598, 122.08721, 78.3, 188, 60, NotAvailable, false, 413443730, 2013-01-01 00:00:03.0,
PositionReportClassAScheduled, AtAnchor, 0,0.4, true, 22.27794, 114.07496, 335.3, 124, 60, NotAvailable, false, 477464300, 2013-01-01 00:00:03.0,
PositionReportClassAScheduled, EngagedInFising, 0,0.0, false, 56.69919, 8.218614, 335.9, 346, 47, NotAvailable, true, 219853000, 2013-01-01 00:00:08.0,
```


Figure 3.3. Examples of AIS messages. Information given includes: message type, navigation status, rate of turn, speed over ground, position accuracy, latitude, longitude, course over ground, true heading, second, special manoeuvre indicator, raim flag, MMSI number, and the timestamp. More details on the information transmitted as AIS data is in Table A3.1.

3.6.3 Source level calculation

The Ship Source Level Model (SSLM), as described in the SONIC report (Colin *et al.*, 2015), was used to calculate the noise emissions for each vessel. The base spectrum used was the Source Spectrum Model (SS_{H+W}) by Wales and Heitmeyer (2002) (Eq. 1), and speed scaling was added to form the SSLM (Eq. 2). Brooker *et al.* (2015) concluded there was minimal benefit to using the SSLM method, but SS_{H+W} results in the same noise emission for all ships, whereas the SSLM method allows vessel specific noise emission through the addition of the speed scaling and thus greater accuracy. The constant added was 8 dB as suggested in the Sonic Report after preliminary validation of their model.

$$SS_{H+W}(f) = 230.0 - 10\,log(f^{\gamma 1}) + 10\,log\left(\left(1 + \left(\frac{f}{\eta}\right)^2\right)^{\gamma 2}\right) \qquad \text{(Eq. 1)}$$

$$SSLM(f, v, l, b, q) = SS_{H+W}(f) + 60log_{10}\left({}^{v}/_{v_{ref}}\right) + c \qquad \text{(Eq. 2)}$$

Where f is the frequency, v is the current vessel speed, l is the ship length, b is the ship breadth, q is the ship type, γ is the power law parameters, η is the breakpoint frequency, v_{ref} is the vessel reference speed, and c is a constant.

Current recommendations for ambient noise (which is inclusive of vessel noise emissions) are to monitor two 1/3-octave frequency bands (63 and 125 Hz) to target areas of intense vessel activity (HM Government, 2012). The SSLM works with a single frequency, so the model was run twice, once at 63 Hz and again at 125 Hz. These two frequencies are both produced by vessels, and are also frequencies that 72 species of the 77 examined in published audiograms (found during the literature review) can hear. For 29 of the species with published audiograms, 125 Hz falls into their optimum hearing sensitivity range, with only 3 of the recorded species optimum hearing sensitivity encompassing the 63 Hz frequency.

Data on vessel speeds (both current and reference), which were necessary for calculating the emissions, were often undefined or recorded as 0 knots within the AIS data. Consequently the current vessel speed (in knots) was calculated at each AIS transmission point using the Haversine formula (Eq. 3); the Haversine formula can be used to calculate the great-circle distances or the shortest distance between the two points on a sphere from their longitudes and latitudes (Sinnott, 1984; Ratsameethammawong and Kasemsan, 2010).

By calculating the Haversine distance (hav) and delta time (the time between two transmission locations) between the current and previous GPS point the likely speed of the vessel could be determined – under the assumption the vessel travelled in a straight line – (Eq. 4). The maximum vessel speed recorded via AIS transmission to date, collected from an online ship database using the MMSI numbers (www.fleetmon.com), was used as a reasonable proxy for a vessel's reference speed (v_{ref}). However, as not all vessels have their maximum speed recorded in online databases some vessel reference speeds were calculated using an alternative method. For each vessel, ship type and dimensions (length & breadth) were calculated and recorded from the AIS data. All ships with no speed data online were matched to another ship of the same type and dimensions (length and breadth) that did have data available allowing an accurate estimate of the maximum recorded speed to be allocated. When no matching vessel was found the ship's dimensions were given leeway of 2 m to get the closest matching vessel. All 458,306 vessels found a match within these parameters.

$$hav\left(\frac{d}{r}\right) = hav(\varphi_2 - \varphi_1) + \cos(\varphi_1)\cos(\varphi_2) hav(\lambda_2 - \lambda_1) \qquad \text{(Eq. 3)}$$

Where d is the distance between two points, r is the radius of the sphere, φ is the latitude, and λ is the longitude.

$$Vessel\ Speed = \frac{hav}{time\ taken} \qquad \text{(Eq. 4)}$$

The SSLM noise calculations were run at each transmission point along a vessel's track, providing a source emission estimate for each individual point based on the specific ship attributes (including length, breadth, ship type and speed). The resulting map was divided into 1 km grid cells and the received levels were calculated for each of the noise sources. The average (SPLav (Eq. 5)) and cumulative (SELcum (Eq. 6)) received levels were then calculated on a cell by cell basis over a one month period providing one noise intensity value per grid cell.

$$SPL_{av} = 10\log_{10}\left(\frac{10^{\frac{SPL_1}{10}}+10^{\frac{SPL_2}{10}} \ldots + 10^{\frac{SPL_x}{10}}}{N}\right) \qquad \text{(Eq. 5)}$$

Where SL_x is the source level for a ship calculated from SSLM and N is the number of SPL.

$$SEL_{cum} = SPL_{av} + 10\log_{10}(N\ x\ T) \qquad \text{(Eq. 6)}$$

Where T is the Time between transmissions in seconds.

To calculate cumulative received levels, the equation (Eq. 6) needs the time period between AIS transmissions (T). The T value was calculated by determining the individual ship transit lines; if a ship's current and next transmission are both in the same grid cell then the full 720 seconds between the transmissions was recorded as the T value. When a ship left the cell between transmissions then a variation of the Liang-Barsky line clipping algorithm (Skytopia, 2006) was used to calculate how long the ship was likely to be in the cell based on its speed and distance from the edge of the cell.

3.6.4 Propagation model integration

Propagation loss (PL) was added into the model, taking into account sediment type and bathymetry. The equation used (Eq. 7) was taken from work by Dekeling *et al.* (2014). The sediment type (EMODnet, 2014) and bathymetry data (BODC, 2016) were collected from online databases and imported into the model using Java programming. Each 1 km grid cell was allocated an average depth, using the bathymetry dataset, and a reflection loss gradient from the sediment type dataset. The allocated reflection loss gradient depended on the sediment type present, for example, sand was allocated a value of 0.25 (Ainslie, 2010). The PL model was run for each adjacent square moving outwards radially from the starting point. To calculate the received level the propagation loss was subtracted from the estimated source level (Eq. 8).

$$PL = 15\log_{10}(R) + 5\log_{10}\left(\frac{\eta H}{\pi Rref}\right) \quad \text{(Eq. 7)}$$

Where R is the range from the centre of the starting square to centre of next square, η is the reflection loss gradient, H is the water depth, and $Rref$ is equal to 1 m.

$$RL = SPL_{av} - PL \quad \text{(Eq. 8)}$$

The received levels occurring in each grid cell, due to propagation from nearby vessels, were added to the original noise level of the cell to produce a final noise output for each 1 km cell. Both the cumulative and average received levels per month were calculated using equations (Eq. 5) and (Eq. 6).

3.6.5 Map generation

The data output by the model was imported into ArcMap 10.3 producing a map of the 1km grid cells with data points in the centre of each cell which stated the received noise value for that cell. The Mercator Web coordinate system was added to the map allowing the points to fit onto the background layer showing the land masses and oceans. The map was coloured using a heat-map technique by applying the Hot Spot Analysis (Getis-Ord Gi*) and the Inverse Distance Weighted (IDW) tools in ArcMap 10.3, showing loud areas in red and quiet areas in blue. The Hot Spot Analysis tool identified statistically significant hot (noisy) spots and cold (quiet) spots. The IDW tool ran an interpolation that estimated cell values by averaging the values of sample data points in the neighbouring cells turning the Hot Spot Analysis data into a smooth map output.

3.6.6 Model validation

This method built upon and combined previous models for Source Level, Sound Pressure Level and Sound Exposure Level calculations, all of which had been validated by the original authors. However, to prove the usefulness of the new ocean vessel noise exposure model validation was performed at various stages.

Model validation consisted of comparing the cumulative Sound Exposure Levels output from the model to actual recordings of shipping noise from earlier publications, and previous modelling publications. Pisani *et al.* (*n.d.*) modelled noise levels measured in the Shetland Islands during 2013. The data in this study covered the area of the previous study during the same months and so the results were comparable. The model was run on the Shetland Islands (latitude from 59.39028 to 61.08184 and longitude from -2.08092 to -0.41488) to compare model outputs. Erbe *et al.* (2012) used modelling to estimate the cumulative sound exposure levels off the coast of Seattle. Although not it the study area, resulting noise exposure levels within the same range help to show validity of the results in this chapter.

3.7 Results

3.7.1 Data analysis

The amount of time ships spent at sea was greater in the summer than winter months, with more AIS transmissions being recorded in July than any other month (Table 3.4 and Figure 3.4). September reported the lowest number of AIS transmissions, suggesting it is the month with the least vessel activity.

Table 3.3. Number of AIS transmissions recorded in each month in 2013. Total number of transmissions used in this chapter was 443,871,900, covering longitude from 20 ºW to 12 ºE and latitude from 44 ºN to 65 ºN. A transmission was recorded every 12 minutes throughout a vessel's journey. Transmissions from vessels whose position was stationary i.e. moored or aground were removed as they would not be contributing to anthropogenic noise.

Month	Data transmissions
January	35,027,525
February	34,756,737
March	38,293,510
April	39,195,759
May	39,538,938
June	40,435,627
July	41,642,674
August	38,349,059
September	32,956,511
October	34,979,901
November	33,685,331
December	35,010,328
Total	***443,871,900***

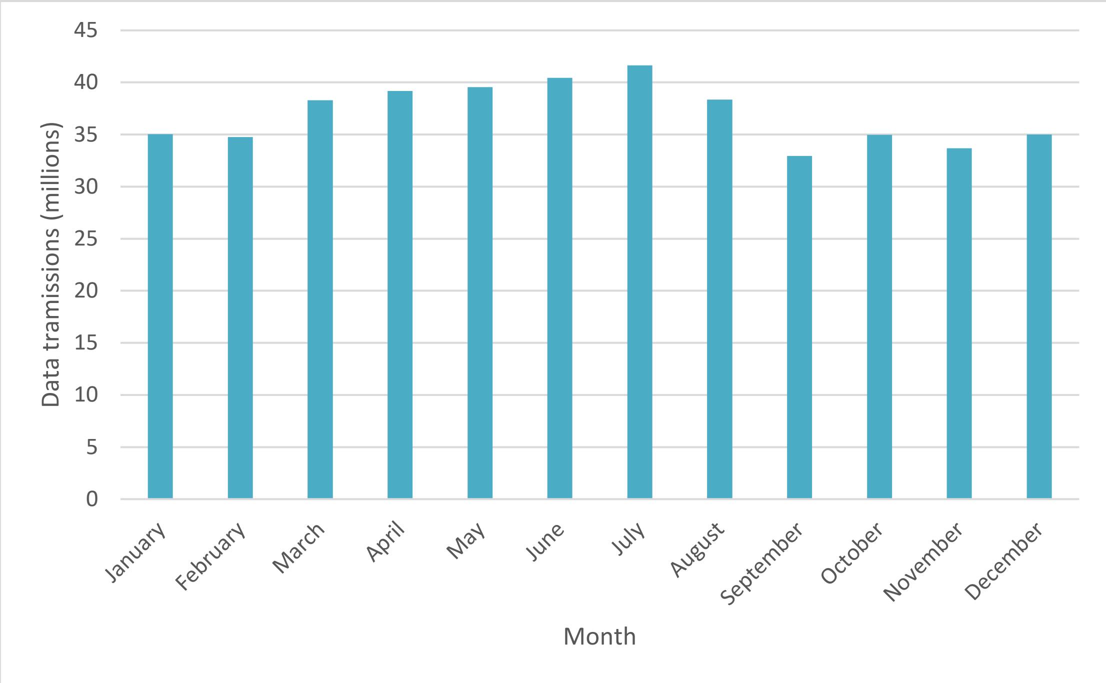


Figure 3.4. Total number of data transmissions (in millions) received from within the study area (longitude from 20 °W to 12 °E and latitude from 44 °N to 65 °N) throughout 2013.

Transmissions from a total of 453,306 individual vessels were used to create the vessel noise exposure map in this chapter. Analyses were run on the transmission data to assess temporal differences, the different ship types, and how many of each type were included. Cargo vessels reported the most transmissions in 2013, with less transmissions recorded during the Autumn and Winter months (Figure 3.5). Activity from vessels transmitting the 'Not Available' code peaked in the summer months and showed the largest temporal fluctuations; July recorded 3,967,845 from transmissions than January (Figure 3.5). Activity of other vessel types showed very little temporal fluctuations. Cargo ships were the largest contributors, providing 30 % of the data, closely followed by the 'Not available' category at 29 % (Table 3.4 and Figure 3.6). 'Not available' is a default value produced when no data is provided by the vessel transmitter. Fishing vessels constituted 17 % of the vessels in the dataset.

The number of vessels flying a UK flag were also counted as Haren (2007) suggests that ships not under the jurisdiction of the UK may not have to follow rules and regulations set for UK waters, which could impact on mitigation measures (such as those suggested in Sections 4.2.3 and 4.2.5). Only 10,073 vessels of the 458,306 in the dataset were registered as UK vessels and as such must follow rules and regulations set by the UK.

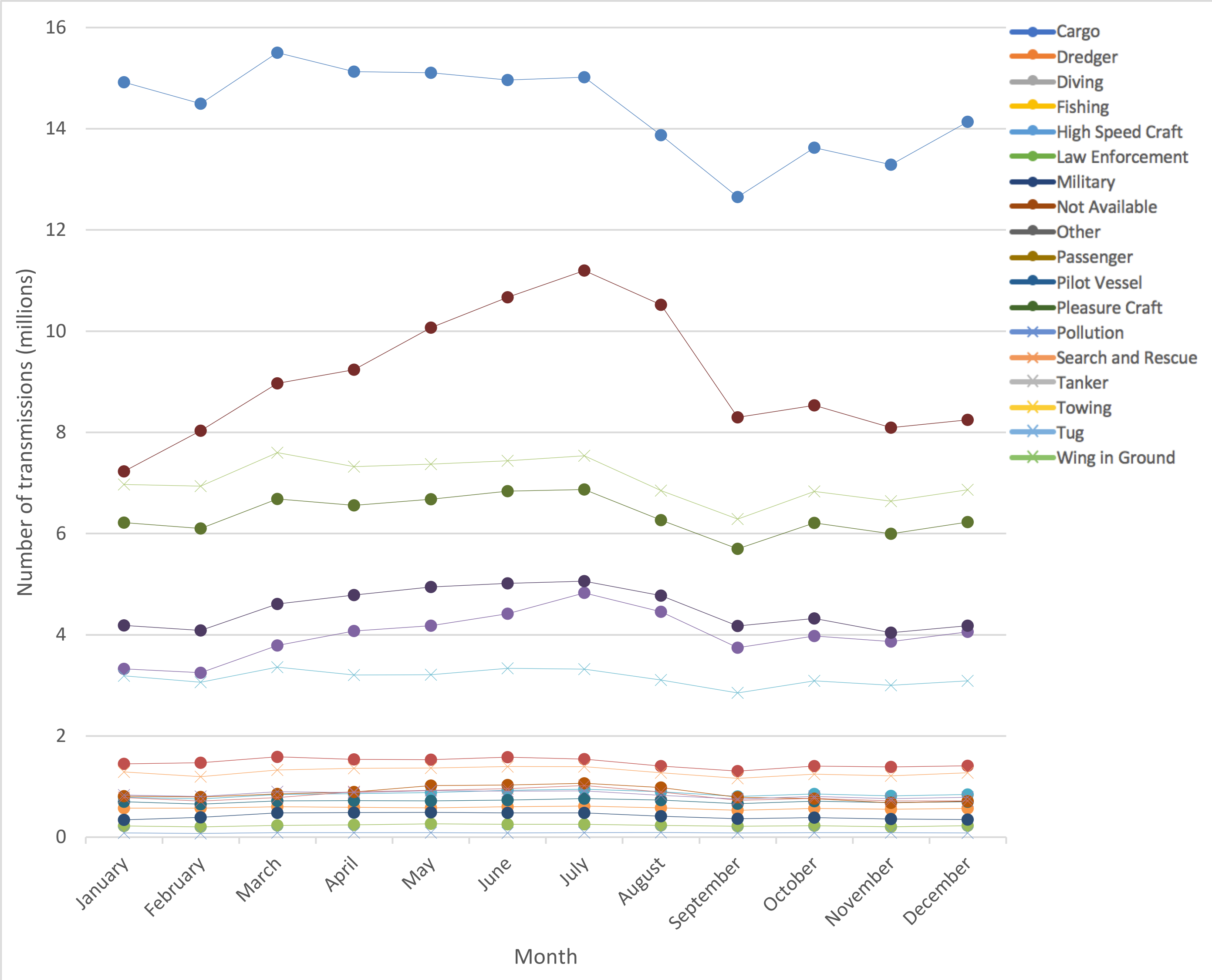


Figure 3.5. The number of AIS transmissions from each vessel type during each month of 2013 covering longitude from 20 °W to 12 °E and latitude from 44 °N to 65 °N. The number of transmissions is indicative of the relative activity, and therefore density, of each vessel type active throughout 2013. A transmission was recorded every 12 minutes throughout a vessel's journey. Transmissions from vessels whose position was either moored or aground were removed.

Table 3.4. The number of ships of each ship type transmitting AIS data during 2013 which constitute the dataset used in this chapter. The Maritime Mobile Service Identity (MMSI) number was used to identify the nationality of the ship to count the vessels with a UK registered port of origin. The first three digits of the MMSI number represent the Maritime Identification Digits (MID), which are three digit identifiers denoting the country responsible for the ship station (Milltech Marine, 2017). MID range from 201 to 775, with numbers 232 to 235 representing the UK.

Ship type	Number of ships	Percentage of total ships (%)	Number of ships flying UK flag	Percentage of ships flying UK flag
Cargo	135,650	30	895	1
Diving Ops	591	0	34	6
Dredger	3,225	1	87	3
Fishing	79,000	17	498	1
High Speed Craft	2,169	0	198	9
Law Enforcement	3,019	1	21	1
Military Ops	1,760	0	85	5
Not Available	133,867	29	5,037	4
Other	16,398	4	417	3
Passenger	11,732	3	302	3
Pilot Vessel	2,478	1	85	3
Pleasure Craft	7,391	2	569	8
Anti-pollution	883	0	12	1
Search and Rescue	2,287	0	164	7
Sailing	4,869	1	814	17
Tanker	29,347	6	419	1
Towing	6,362	1	56	1
Tug	13,153	3	352	3
Wing in Ground	4,125	1	28	1
Total	***458,306***		***10,073***	

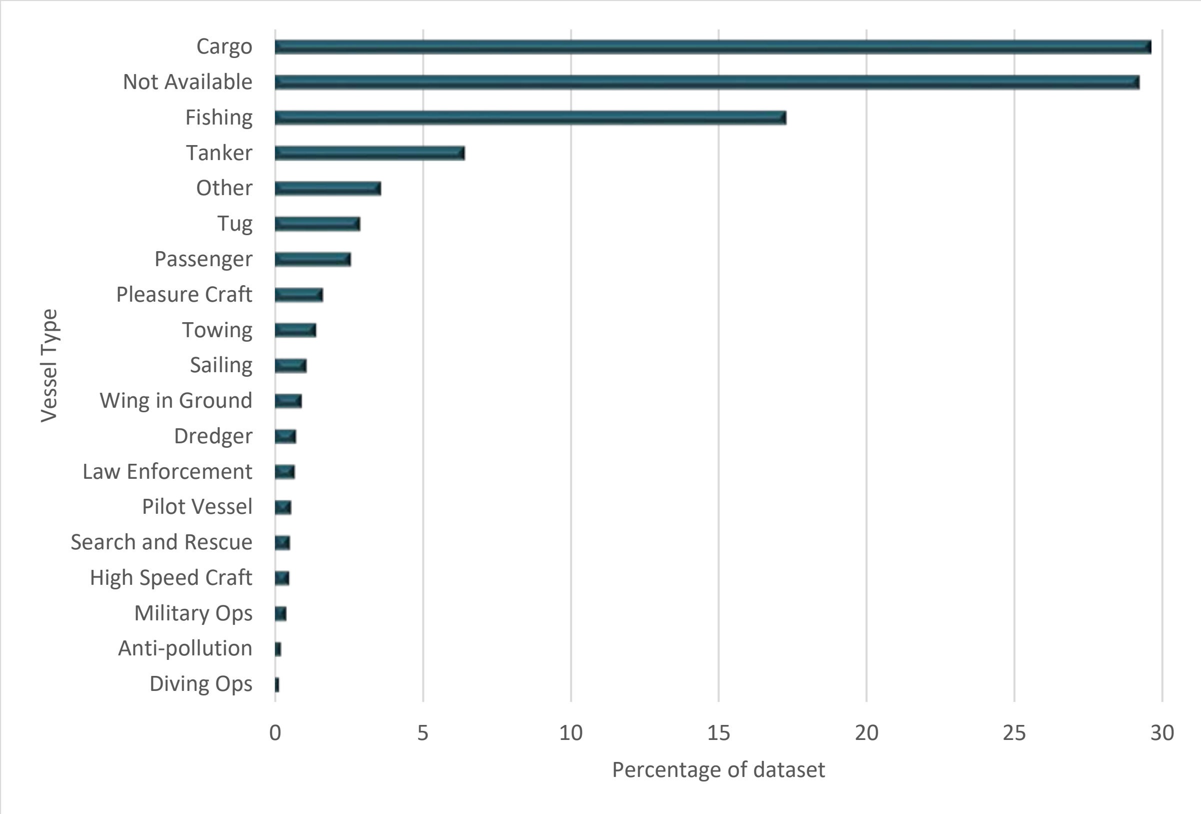


Figure 3.6. Percentage of the dataset made up of each vessel type. Dataset was from longitude 20 °W to 12 °E and latitude 44 °N to 65 °N for 2013.

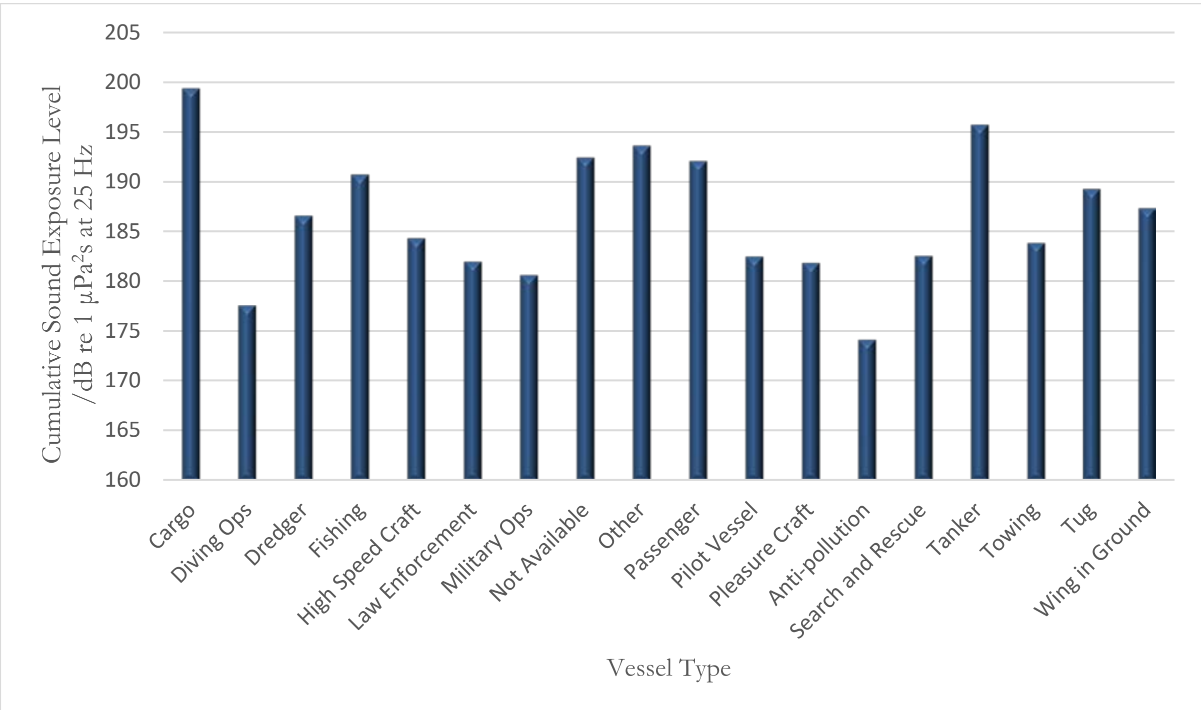


Figure 3.7. Yearly cumulative Sound Exposure Levels (SELcum) at 125 Hz for each vessel type during 2013, covering longitude from 20 °W to 12 °E and latitude from 44 °N to 65 °N. SELcum was calculated from AIS data recorded every 12 minutes throughout a vessels journey.

Cumulative sound Exposure Levels varied between vessel types, with cargo vessels producing the loudest SELcum at 199 dB re 1 µPa^{2}s and anti-pollution vessels emitting the lowest SELcum at 174 dB re 1 µPa^{2}s. The second loudest SEL was produced by tankers, which is interesting as, relatively speaking, the number of tanker transmitting AIS data was much lower than the number of cargo vessels, and yet the SELcum were only 3 dB different. The differences between vessel SELcum were not significant ($H(17)= 17$, $p = 0.4544$).

3.7.2 63 Hz

The model produces heat-map outputs showing the received levels over monthly periods for a specified area (Figure 3.8 and Figure 3.9). Temporal and geographical differences were observed throughout the year (Figure 3.8 and Figure 3.9). The North Sea has less vessel activity and lower noise exposure levels during January to March compared to the rest of the year. The Baltic Sea is noisy all year round, with March showing slightly lower cumulative noise exposure levels. During 2013, the peak cumulative received noise level recorded was in July where one grid cell reached 222 dB re 1 µPa^{2}s and the loudest overall month (July) had a noise exposure level (recorded during July) of 224 dB re 1 µPa^{2}s.

The noisiest area throughout the whole year was shown to be the west coast of the Netherlands and South-East coast of England. The English Channel is shown to be an area of constantly high shipping noise emissions, with no break from noise during any month of the year. The model shows that although there are some quieter spots close to the shores of the channel, the noise emissions away from the coastline never drop below an average received noise level of 159 dB re 1 µPa^{2}s. In September, the shipping noise levels start to decrease hitting their quietest point in December before increasing again in January. The cumulative noise exposure levels in this channel range from 140 dB re 1 µPa^{2}s, in some small areas, to 208 dB re 1 µPa^{2}s throughout the year. The channel between the North Sea and Baltic Sea is another area of loud ship noise emissions, but unlike the English Channel, the noise levels do vary throughout the year.

The Irish Margin in the North Atlantic Ocean is shown to be a much quieter area of the ocean than other parts of the North Atlantic, North Sea and Baltic Sea, especially in the spring months with monthly exposure levels between 100-150 dB re 1 µPa^{2}s (indicated by the green areas in Figure 3.8 and Figure 3.9), and some area as low as 101 dB dB re 1 µPa^{2}s (Figure 3.8 b and Figure 3.8 c).

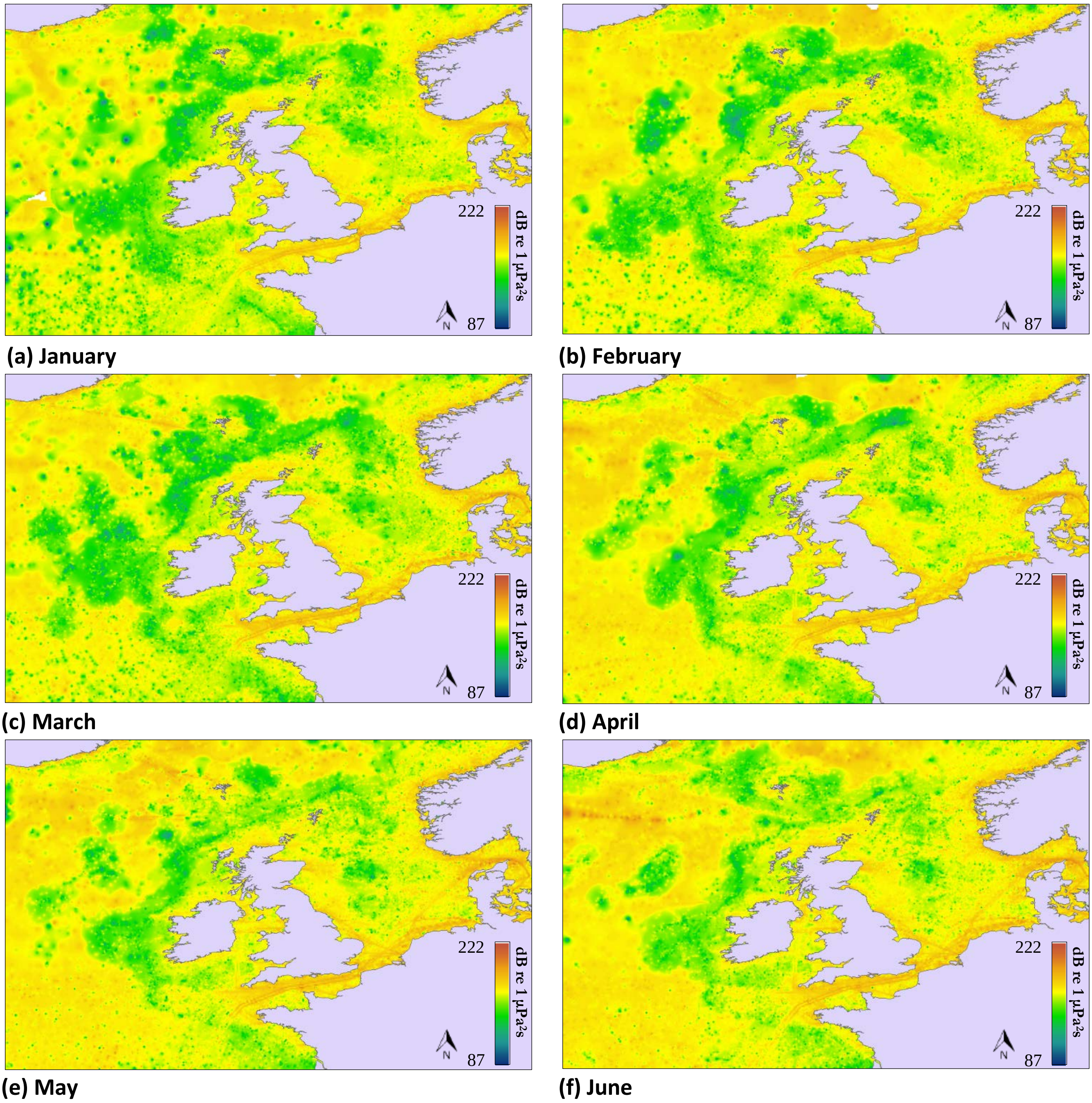


Figure 3.8. Monthly maps (January to June) produced by the model showing the cumulative Sound Exposure Levels (dB re 1 μPa^{2}s) of shipping noise emissions at 63 Hz during 2013. Noise levels range from 87 dB (blue) to 222 dB (red). The noise levels are the cumulative level over 1 km grid squares, with inverse distance weighting to create a smooth map output. Each frame stretches in longitude from 20 °W to 12 °E and in latitude from 44 °N to 65 °N.

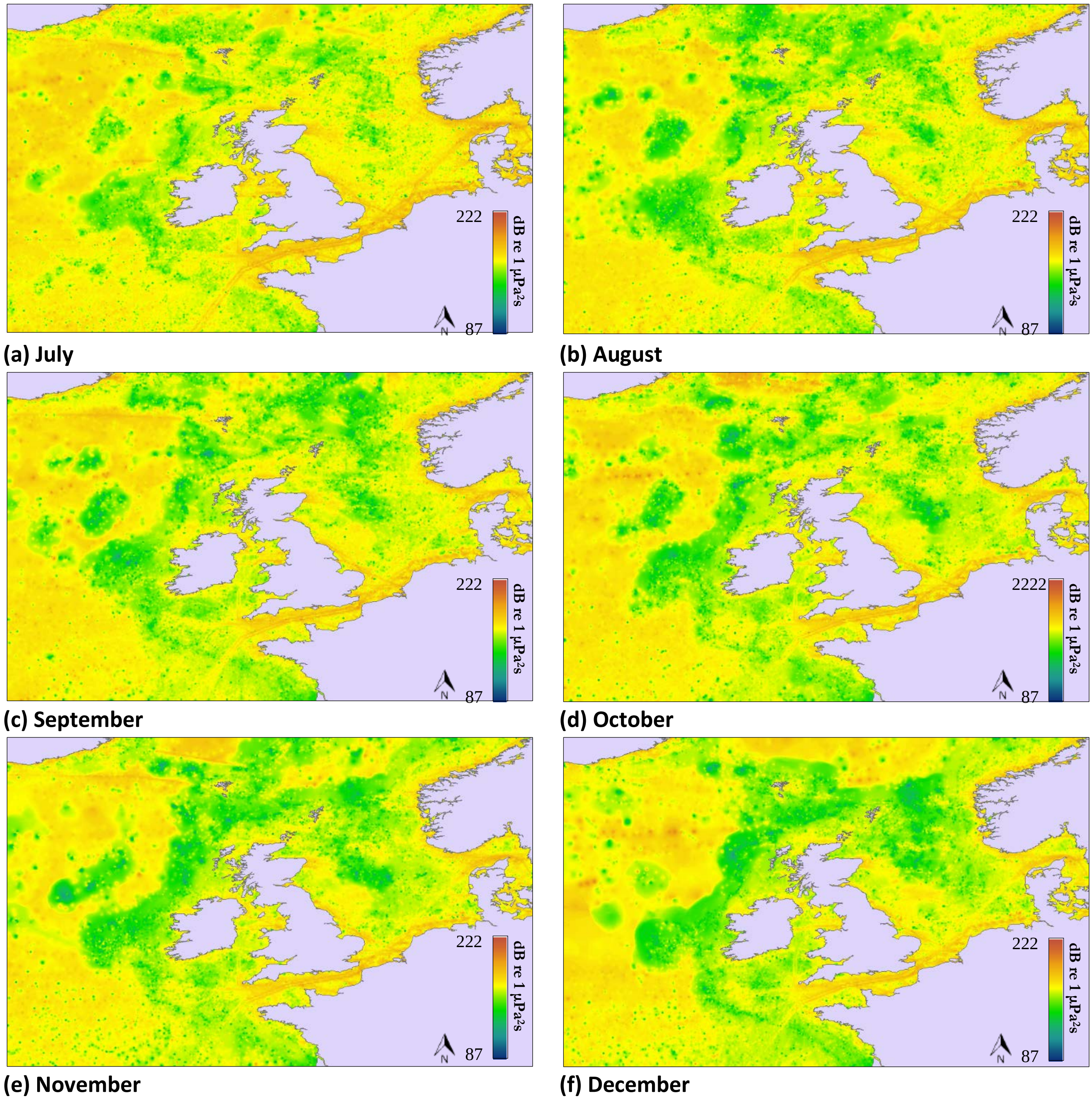


Figure 3.9. Monthly maps (July to December) produced by the model showing the cumulative Sound Exposure Level (dB re 1 $\mu Pa^2 s$) of shipping noise emissions at 63 Hz during 2013. Noise levels range from 87 dB (blue) to 222 dB (red). The noise levels are the cumulative level over 1 km grid squares, with inverse distance weighting to create a smooth map output. Each frame stretches in longitude from 20 °W to 12 °E and in latitude from 44 °N to 65 °N.

3.7.3 125 Hz

The peak cumulative received noise level recorded was in October where one grid cell reached 188 dB re 1 μPa^2s, however, this was not far above all other months as they exhibited peak levels of 187 dB re 1 μPa^2s. The loudest recorded SELcum for any one month was 192 dB re 1 μPa^2s which occurred in July. As with the cumulative Sound Exposure Levels at 63 Hz, the Baltic Sea was loud all year round (Figure 3.10 and Figure 3.11). The lowest SELcum levels for the Baltic were recorded in March, however, the exposure levels still reached as high as 166 dB re 1 μPa^2s.

The Irish Margin was the quietest region, reaching as low as 84 dB re 1 μPa^2s in small areas in January 2013 (Figure 3.10 a). However, in January, a large percentage of the ocean is under 140 dB re 1 μPa^2s (indicated by the green areas in Figure 3.10 and Figure 3.11), but this changes throughout the year and by April noise exposure starts to rise towards 160 dB re 1 μPa^2s.

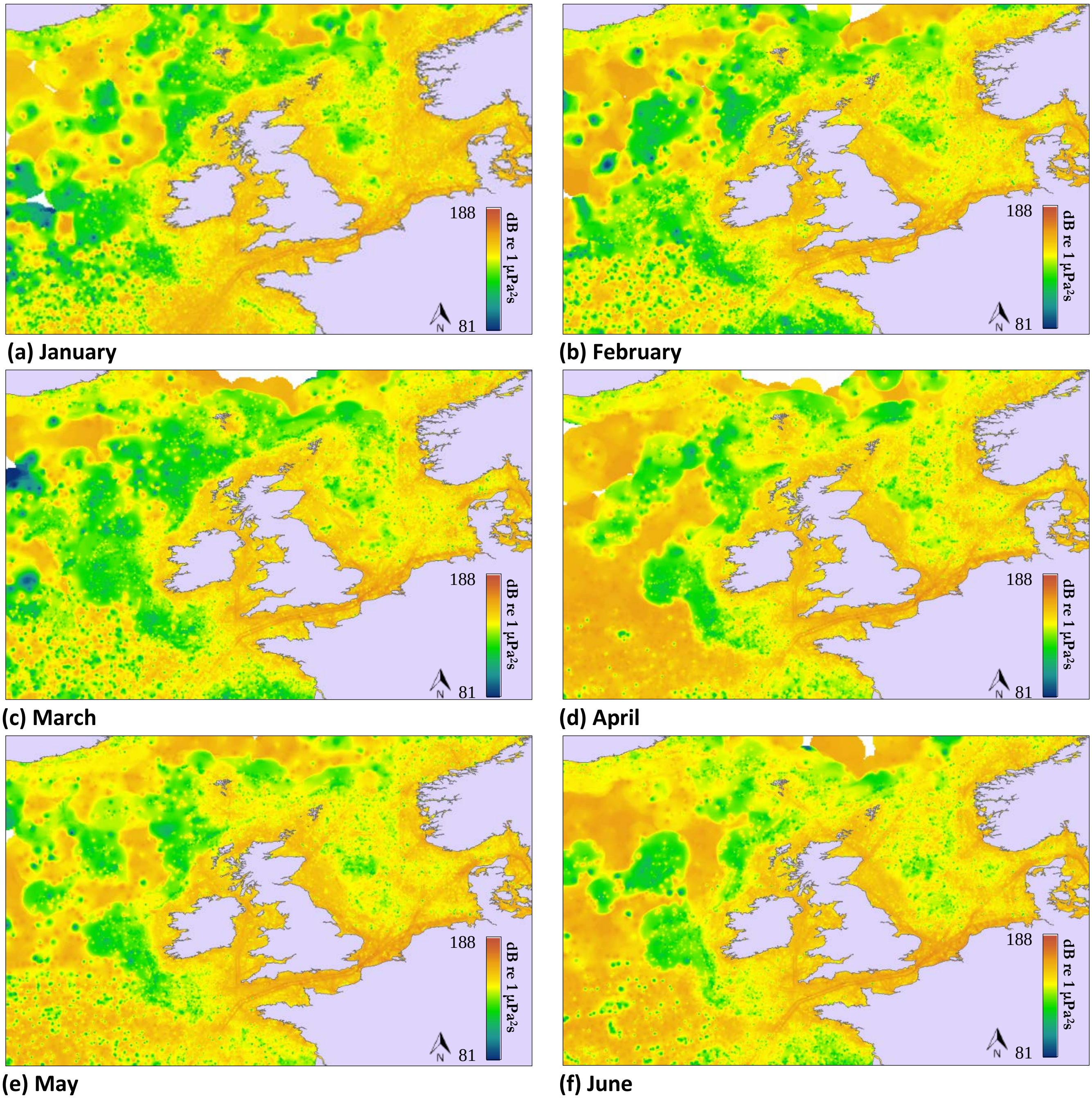


Figure 3.10. Monthly maps (January to June) produced by the model showing the cumulative Sound Exposure Level (dB re 1 $\mu Pa^2 s$) of shipping noise emissions at 125 Hz during 2013. Noise levels range from 81 dB (blue) to 188 dB (red). The noise levels are the cumulative level over 1 km grid squares, with inverse distance weighting to create a smooth map output. Each frame stretches in longitude from 20 °W to 12 °E and in latitude from 44 °N to 65 °N.

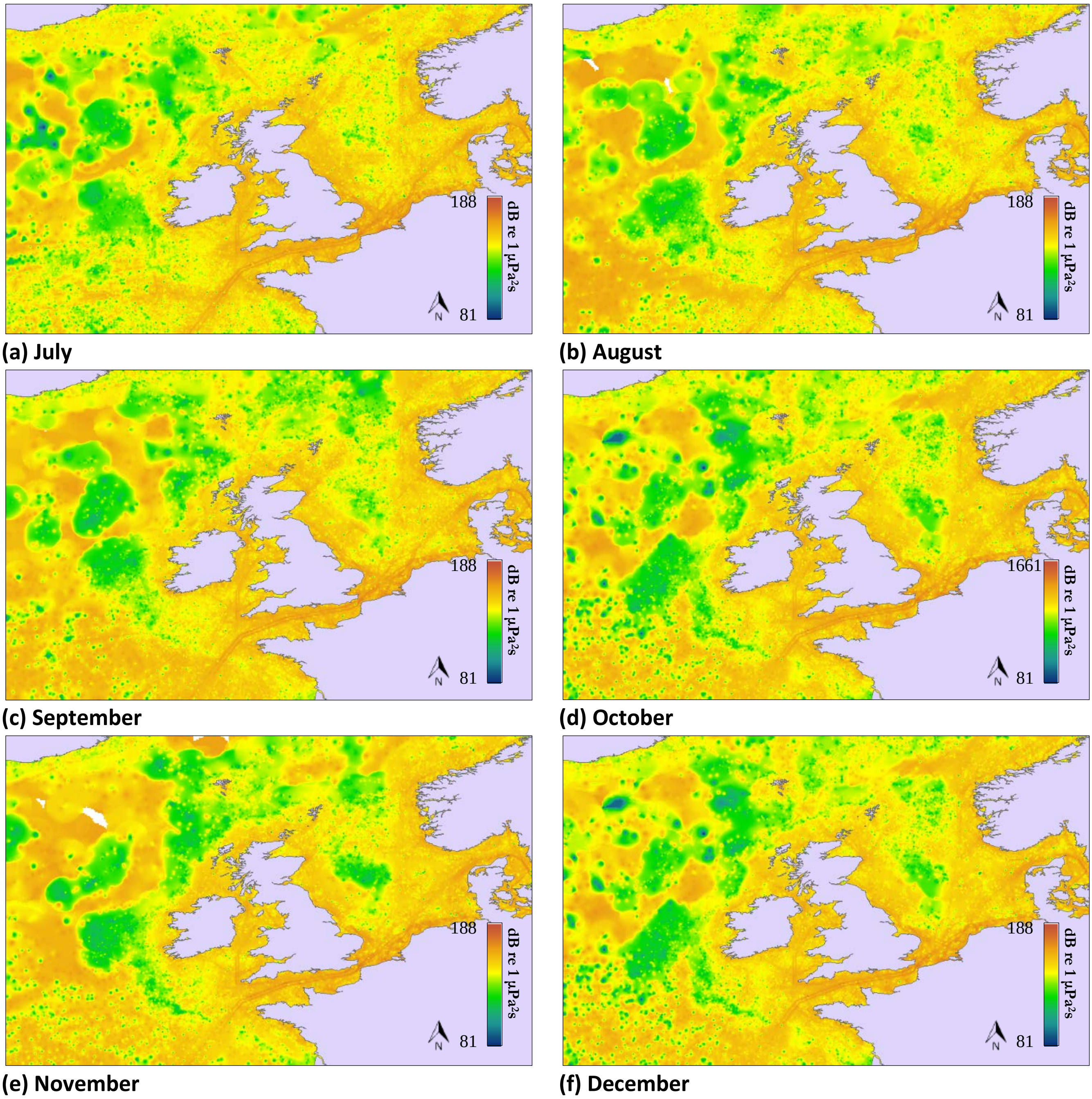


Figure 3.11. Monthly maps (July to December) produced by the model showing the cumulative Sound Exposure Level (dB re 1 μPa^{2}s) of shipping noise emissions at 125 Hz during 2013. Noise levels range from 81 dB (blue) to 188 dB (red). The noise levels are the cumulative level over 1 km grid squares, with inverse distance weighting to create a smooth map output. Each frame stretches in longitude from 20 °W to 12 °E and in latitude from 44 °N to 65 °N.

3.7.4 63 and 125 Hz compared

The SELcum at 63 Hz is much louder (more than 26 dB louder) than the SELcum at 125 Hz throughout the year (Figure 3.12). Consequently, fish species who are more sensitive to lower frequency noise may be at higher risk of impact than species who have sensitivity to higher frequencies. European eel, for example, have an optimum hearing sensitivity of 80 Hz and so may be impacted greater than mullet whose optimum hearing sensitivity is around 400 Hz (*see* Figure A1.1 and Figure A1.2 for published hearing sensitivities of various species). July is the noisiest month for both frequencies. For both frequencies, the patterns of noise across the study area is similar, with the same areas showing higher levels of noise exposure (such as the English Channel) (Figure 3.13 and Figure 3.14).

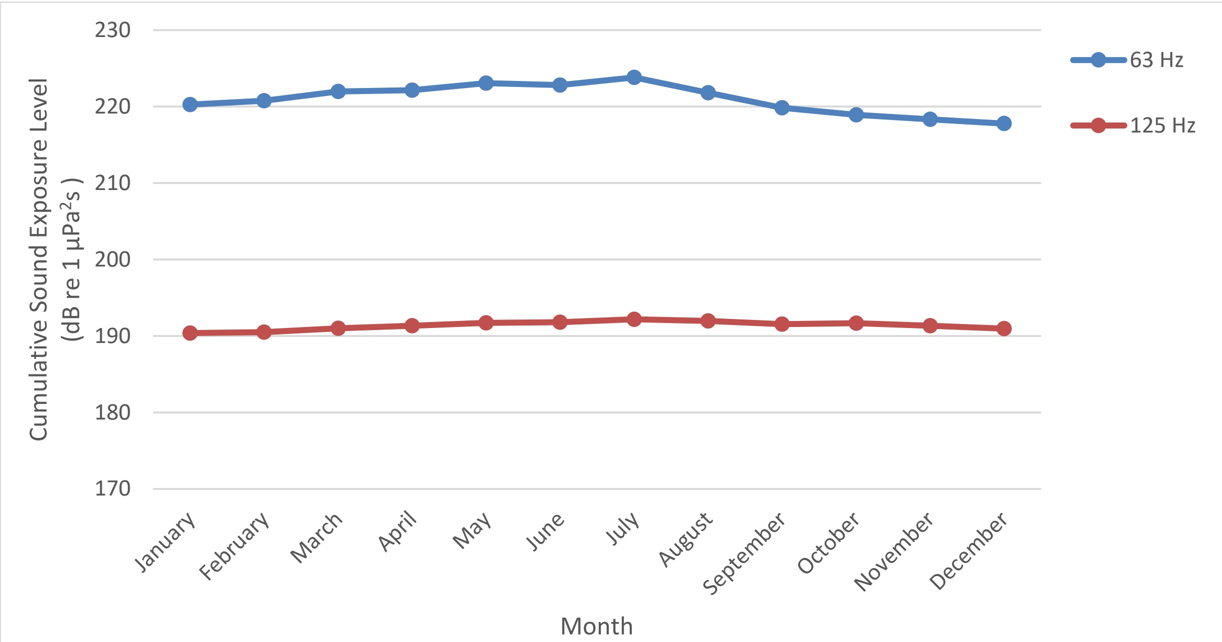


Figure 3.12. Cumulative Sound Exposure Levels for longitude 20 °W to 12 °E and latitude 44 °N to 65 °N during each month of 2013 at 63 Hz and 125 Hz. All vessels transmitting AIS data were included.

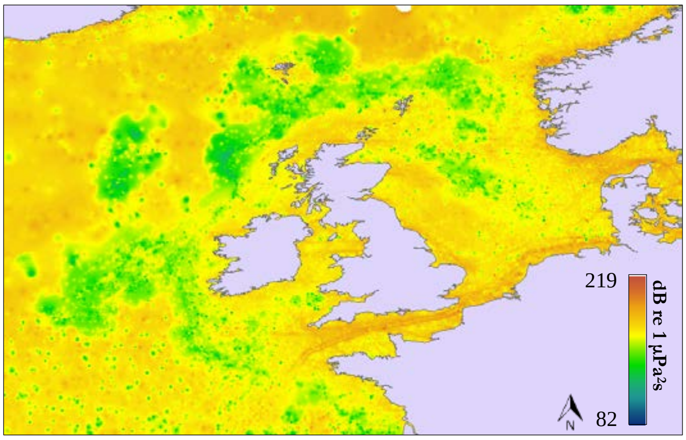


Figure 3.13. Cumulative Sound Exposure Levels (dB re 1 µPa^2s) of vessel noise emissions at 63 Hz during February 2013. Noise levels range from 82 dB re 1 µPa^2s (blue) to 219 dB re 1 µPa^2s (red). The noise levels are the cumulative SEL over 1 km grid squares, with inverse distance weighting to create a smooth map output. Each frame stretches in longitude from 20 °W to 12 °E and in latitude from 44 °N to 65 °N.

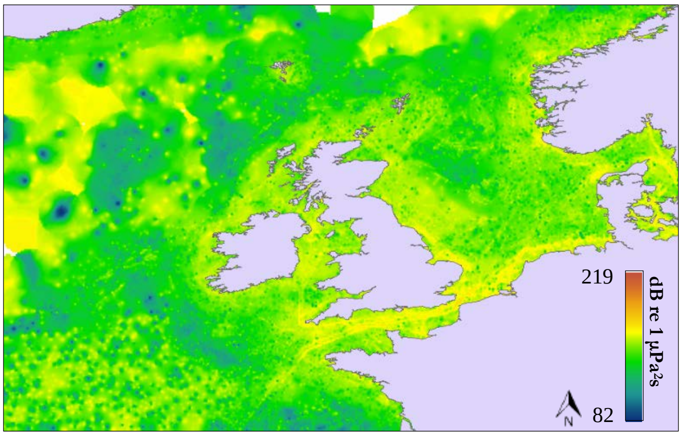


Figure 3.134. Cumulative Sound Exposure Levels (dB re 1 µPa^2s) of vessel noise emissions at 125 Hz during February 2013. Noise levels range from 82 dB re 1 µPa^2s (blue) to 219 dB re 1 µPa^2s (red). The noise levels are the cumulative SEL over 1 km grid squares, with inverse distance weighting to create a smooth map output. Each frame stretches in longitude from 20 °W to 12 °E and in latitude from 44 °N to 65 °N.

3.7.5 Model validation

The validity of the model was checked by comparing the model results to two previous studies on cumulative noise exposure levels. The model in this chapter produced monthly SELcum values ranging from 218 to 224 dB re 1 µPa^2s for 63 Hz, and 190 to 192 dB re 1 µPa^2s for 125 Hz. The year SELcum values were 220 dB re 1 µPa^2s for 63 Hz and 192 dB re 1 µPa^2s for 125 Hz.

Erbe *et al.* (2012) modelled a maximum SELcum of 215 dB re 1 µPa^2s which is near to the range found in this study of 218 – 224 dB re 1 µPa^2s. The cumulative Sound Exposure Levels from the model in this study are slightly higher than those reported by Erbe *et al.* (2012), but the larger geographic area incorporating major shipping routes such as the English Channel and Baltic Sea likely account for the discrepancy.

Work by Pisani *et al.* (*n.d.*) calculated the SELcum around the Shetland Islands at 63 Hz and 125 Hz. The highest registered SELcum values were 191 dB re 1 µPa^2s at 63 Hz and 175 dB re 1 µPa^2s at 125 Hz.

The model built in this chapter covered the Shetland Islands in the same year as Pisani *et al.*'s (*n.d.*) work and so could be directly compared (Figure 3.15, Figure 3.16 and Figure 3.17). The SELcum for the year at 63 Hz was 196 dB re 1 µPa^{2}s with a peak level of 213 dB re 1 µPa^{2}s. For 125 Hz the SELcum for the year was 166 dB re 1 µPa^{2}s with a peak level of 181 dB re 1 µPa^{2}s. For Quarter 1 of the 2013 (January, February and March) the SELcum at 63 Hz was 199 dB re 1 µPa^{2}s with a peak level of 214 dB re 1 µPa^{2}s. These levels are, again, slightly higher than the work used to validate the study, but are likely due to noise propagating in from vessel sources outside Pisani *et al's* study area owing to the larger geographic area incorporated in the current model.

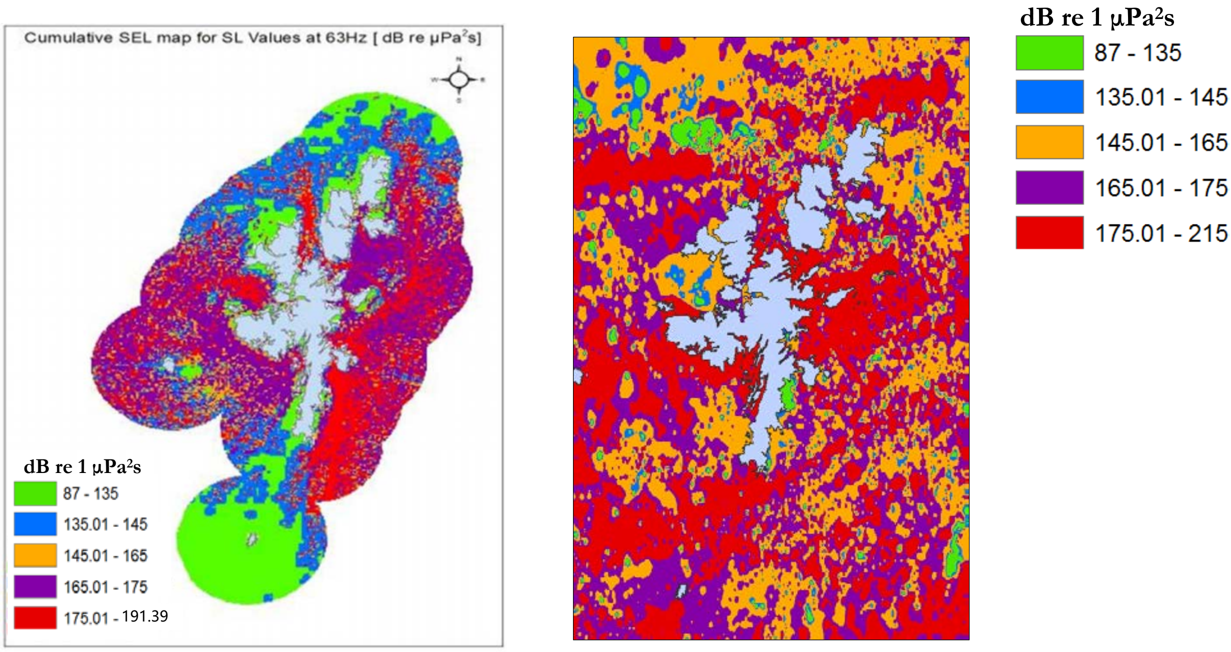


Figure 3.14. Comparison of model output for annual cumulative Sound Exposure Levels for 2013 at 63 Hz with work by Pisani *et al.* (*n.d.*). A similar scale and colour scheme was used to allow easier comparison, however, due to greater geoprahical area and outside vessel noise propagating in to the frame, SELcum levels were slightly higher which is reflected in the legend.

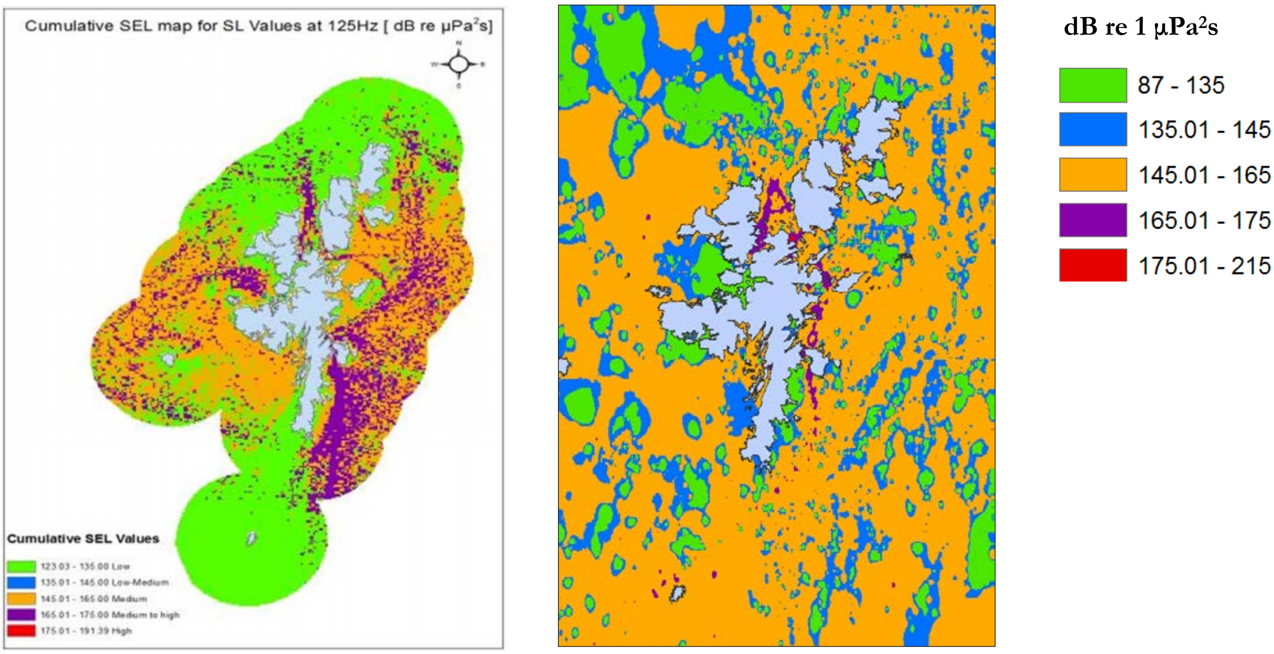


Figure 3.15. Comparison of model output for annual cumulative Sound Exposure Levels for 2013 at 125 Hz with work by Pisani *et al.* (*n.d.*). A similar scale and colour scheme was used to allow easier comparison, however, due to greater geoprahical area and outside vessel noise propagating in to the frame, SELcum levels were slightly higher which is reflected in the legend.

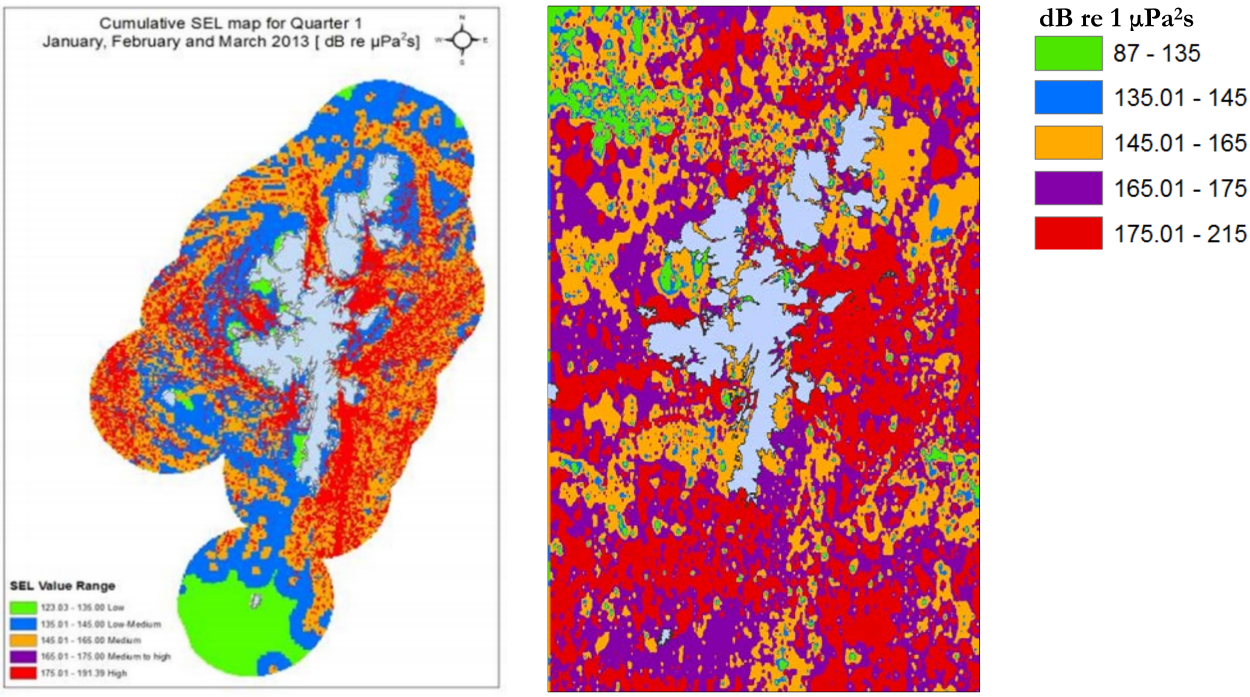


Figure 3.16. Comparison of model output cumulative Sound Exposure Levels, at 63 Hz for January, February and March combined, with work by Pisani *et al.* (*n.d.*). A similar scale and colour scheme was used to allow easier comparison, however, due to greater geoprahical area and outside vessel noise propagating in to the frame, SELcum levels were slightly higher which is reflected in the legend.

3.8 Discussion

The model presents map outputs of average cumulative received noise levels over monthly periods throughout 2013. The loudest recorded months occurred over the summer period with July showing the loudest monthly cumulative Sound Exposure Level (Figure 3.9 a, Figure 3.11 a, and Figure 3.12). The winter months produced quieter cumulative noise levels, particularly December and January (Figure 3.9 f, Figure 3.10 a, and Figure 3.12). The map shows no obvious area of high intensity. The peak noise levels reported from both the 63 Hz (222 re 1 μPa^2s in July) and 125 Hz (188 re 1 μPa^2s in September) results were not visible on the map outputs and so must have been small areas, or even just one grid cell. There were no large hotspots at the peak levels.

The English Channel and the channel between the North Sea and Baltic Sea are the noisiest areas in terms of vessel noise exposure. The map outputs clearly show that there is high noise intensity all year round, with no months showing lower noise emissions in these areas. Consequently, these areas may be of particular concern when considering the effects of noise on populations and indicate where mitigation efforts should be focused. Both these channels hold entrances to major river systems used by many different fish species for spawning. For example, the endangered European eel (*Anguilla anguilla*) migrates from the Baltic Sea into and through the North Sea as a silver eel. During this migration, the eel is in a more fragile state as it undergoes an irreversible physiological transformation in which the eyes and pectoral fins are enlarged, the skin colour changes, and the digestive organs are regressed (Andersson, 2011). Impacts from noise at this time could be particularly damaging. The eels, as juveniles, also have to migrate in the opposite direction so could be prone once again to noise impacts.

Interestingly, between the west coast of England and east coast of Ireland there is only one obvious point of high intensity noise which is that between Liverpool and Dublin. Whilst other ports exist along the two coasts they mainly deal with ferry traffic, indicating that passenger ferries are not responsible for as much of the noise pollution as the larger cargo ships entering and leaving the major ports of Liverpool and Dublin. There are lower numbers of passenger ships than other vessels (passenger vessels made up just 3 % of the total vessels recorded), and they produce lower noise emissions than cargo vessels (yearly SELcum 7 dB re 1 μPa^2s less)

To validate the accuracy of the AIS data used, various analyses were conducted on the data to check that patterns in the data made sense and agreed with previous research. The number of transmissions (and therefore activity) from the vessels was greatest in June and July which coincides with summer tourism. Increases in the number of vessels during the tourist season compared to non-tourist season have been observed (Rako *et al.*, 2013). Numbers of vessels within each vessel classification generally stayed consistent throughout the year, with slight fluctuations in cargo vessels in September (Figure 3.5). The peak in transmissions of the 'Not Available' classification was in the tourist season, and so could be a result of personal craft or vessels not required to carry AIS transmitters (i.e. vessels under 300 gross tonnes) voluntarily transmitting data that is not provided in a complete state. Incomplete data results in a default value of 'Not Available' being transmitted.

Codarin and Picciulin (2015) stated that the absence of seasonal variation in the local noise levels they reported could be explained by the fact that cargo vessel traffic has been found to be homogeneous all year round. Slight temporal variations were found in this study, and those variations do follow the pattern of vessel activity throughout the year. Cargo activity only varied by a few percent across the months of 2013 (Figure 3.18). The slight variation found here could be because the study area and dataset was much larger than that used by Codarin and Picculin (2015), and covered a different geographic area encompassing major shipping routes.

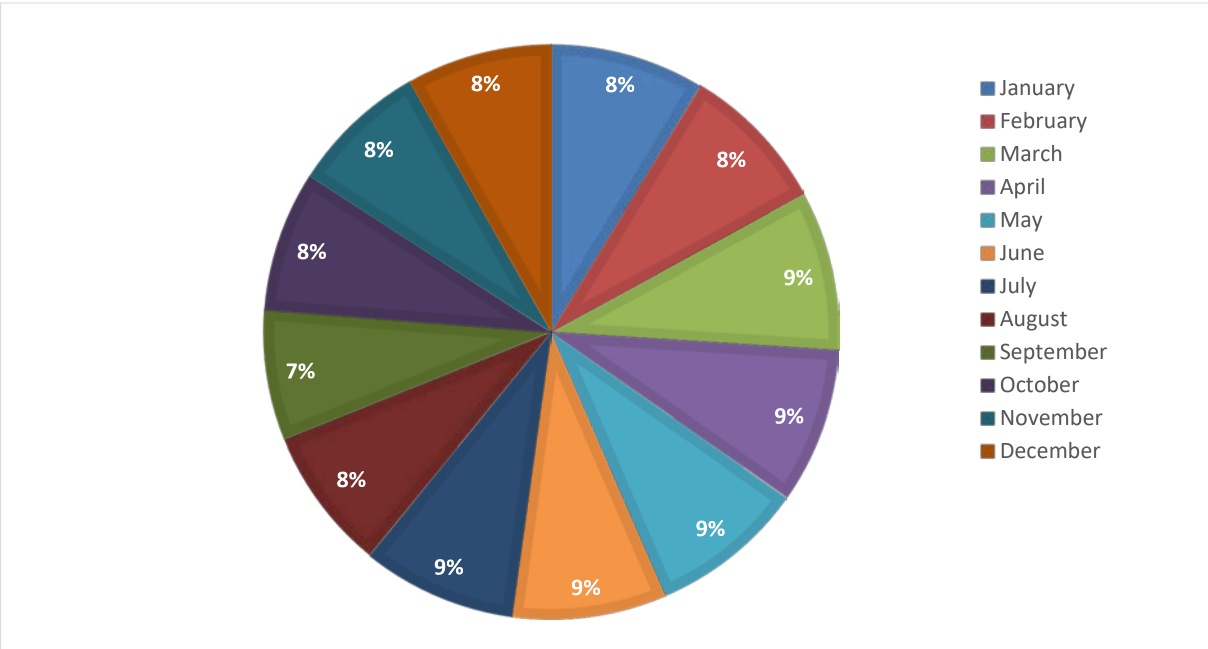


Figure 3.17. Percentage of total cargo ship transmissions occurring each month of 2013.

This chapter shows the ability of ocean noise models to analyse vast data sets to provide information useful to mitigation decision-makers. The work in this chapter focused on the UK, which consisted of up to 41,642,674 AIS data transmissions from vessels in one month. However, the model has the potential to produce data for larger study areas, or even globally, as tests were run on areas consisting of 183,411,585 transmissions. Additional computing capacity may be needed for more complex propagation models over large geographic areas.

Using the model to analyse noise exposure levels at different frequencies allows comparisons to be made, and impacts on species to be predicted. Fish hearing capabilities differ so certain frequencies could impact some species more than others. Vessel emissions at lower frequencies produce noise at higher intensities (Figure 3.12), and fish species exhibiting higher sensitivity to low frequency noise may be more at risk of injury from this type of noise exposure. Producing user-friendly maps allows easy identification of potential hotspots for any frequency specified as an input parameter to the model. Therefore, if interested in European eel, for example, the model can be set to 80 Hz (their optimum hearing sensitivity) and the output will indicate likely problem areas for the species.

The propagation model currently used is a practical spreading model that incorporates both sediment type and bathymetry data and is computationally efficient (Dekeling *et al.*, 2014); the propagation model can be run for the UK map (Figure 3.3) in a matter of minutes. The propagation equation used in the current model is more appropriate for shallow water than deeper, so more complex propagation models should be considered for future improvements.

Modelling acoustic propagation conditions is an important issue in underwater acoustics, and as a result there are several mathematical/numerical models based on different approaches that have been developed during past research (Hovem, 2013). Many of these approaches were considered during the development of this model to find the optimum method, or optimal combination of methods, so as to incorporate temperature and the sound speed profile (Table 3.5). Sound absorption is important for long range propagation and is known to increase with frequency and be dependent on temperature, salinity, depth and the pH value of the water (Hovem, 2013). The choice of a range dependent model is vital when running large scale propagation calculations as it allows the environmental input parameters, such as bathymetry and temperature, to vary with distance from the source (Wang *et al.*, 2014). There can be large differences in computational speed for different models, and often a decision between higher fidelity/accuracy and the computational time is required (Wang *et al.*, 2014). In addition, for given propagation conditions there will be a number of modelling solutions which could provide the appropriate accuracy, and so computational power may become the distinguishing factor. Based on the findings in Table 3.5, it is recommended that a Parabolic Equation or Wave

Number Integration method seems most appropriate for vessel noise mapping and should be integrated into the model for future use.

Table 3.5. Alternative models for propagation for recommendation in future vessel noise mapping research. Sources: Porter and Bucker, 1987; Zeiger *et al.*, 2012; Hovem, 2013; Wang *et al.*, 2014; Spiga, 2015. From this table, the recommendation is to use Parabolic Equation or Wave Number Integration as these methods focus on low frequency noise and work well with varying depths.

Name	Description	Works for:	Notes
Ray Tracing	Uses sound propagation conditions when the sound originated from a point source changes little over distances	– Low frequency – High frequency – Deep water – Shallow water – Range dependent	– More valid for high frequency than low frequency (especially limited below 200 Hz) – Sufficiently accurate for applications involving echo sounders, sonar, and communications systems for short and medium short distances – Ray theory has limitations and may not be valid for precise predictions of sound levels
Beam Tracing	Approximates a given source by a fan of beams and tracing the beams propagation through the medium and summing the contributions of each of the individual beams	– High frequency – Deep water	– Computationally very fast – Incorporates directivity pattern of certain frequencies – Uses sound speed profile and water-air/sediment interfaces – Created as an improvement on ray tracing models
Normal Mode	Uses separation of variable to solve the local vertical part of the wave equation, and then applies various solutions to solve the horizontal component	– Low frequency – Deep water – Shallow water – Range dependent – Range independent	– Works best when horizontal sound speed is constant but vertical sound speed changes – Incorporates sound speed profile and seabed properties – Best suited to low frequencies and mildly range dependent environments
Parabolic Equation	Uses the Helmholtz equation with an approximation that only the out-going wave is considered	– Low frequency – Deep water – Shallow water – Range dependent	– Good for irregular sound speed profiles – Commonly used as considered better than other methods – Incorporates sediment type and seawater absorption – Generally used for frequencies under 1 kHz due to computational requirements
Wave Number Integrations	Solves the wave equation at close range using a numerical approach	– Low frequency – High frequency – Deep water – Shallow water – Range independent	– Is an exact solution – Can be used for range dependent models but the model is not publically available

RAM	Range dependent Acoustic Modelling based on the Parabolic Equation model	– Low frequency – Range dependent	– Can be computationally difficult depending on scenario – Freely available prewritten code
Energy Flux	A hybrid method between rays and modes	– Low frequency – High frequency – Shallow water – Range dependent	– Incorporates bathymetry and sediment type – Assumes a homogenous sound speed profile (only true in coastal waters) – Computationally fast

This work acknowledged the opinions of previous noise model authors' and has broadened the study by utilizing an entire year of AIS data to map vessel noise emissions temporally and look at monthly comparisons and trends in noise emissions. This means that no predictions or generalisations had to be made, and seasonal variations in both vessel movements and marine ecosystems were accounted for in the data. The model created here can be further developed to create more accurate tools for noise assessment, prediction and mitigation.

There is doubt about the efficacy of AIS-based approaches to noise modelling owing to only certain vessels carrying operational AIS transmitters (Merchant *et al.*, 2016). Yet, it has been demonstrated (in the Sutors, Moray Firth, Scotland) that noise emissions generated by AIS-carrying vessels are generally greater than those produced by non-AIS vessels for frequency bands 100–10,000 Hz, and most noise emissions were attributable to the vessels operating with AIS transmitters (Merchant *et al.*, 2016). Noise models using AIS data should account for most of the vessel noise emissions present, assuming that the source levels and propagation models used are sufficiently accurate. Such models can be applied to predict shipping noise levels under various scenarios and suggest areas in need of mitigation.

There are still many vessels without AIS transmitters, such as recreational craft and small fishing vessels, and adding data from these sources would benefit future work as it will add extra dimensions to the research. However, even without these extra sources, this study builds a robust methodology for use both now and in the future as AIS/EMA/VMS data become more freely available.

The essential information taken from the AIS data during the running of this model was the MMSI number and GPS data. All other information needed, such as the vessels' reference speed and ship attributes could be acquired from online databases. This means that if small craft without AIS transmitters were able to provide GPS locations via satellite navigation systems or mobile phones they could still be included in the model. Although small craft GPS is not as easily available as AIS data, with adequate planning it could still be collected and included in the noise map outputs if necessary. As well as AIS data, Vessel Monitoring Systems data used by larger fishing vessels and Electronic Monitoring System Aggregate data used by dredging vessels could be input into the model to add more depth to the resulting vessel noise maps.

A limitation of previous mapping attempts was the accurate assignment of a ship's reference velocity – its speed when operating under normal service power and loading, in average weather conditions (Eyres and Bruce, 2012) – for use in the noise calculation. Reference speeds vary considerably between different ships and until now these reference speeds were assigned per ship type, with all tankers being allocated the same velocity, all cargo ships being allocated the same velocity, and so on. In the model designed here, the service speed is specific to the individual ship depending on its type, length and breadth, and previous recorded speeds from historical AIS data. This method allows for more accurate estimation of noise emissions as speed has been shown to be an influencing factor.

Whilst ocean temperature and the sound speed profile are not built into this model, it is interesting that there is still a visible difference in ship noise intensity between the summer and winter months. The high intensity in the summer is likely driven by a larger number of ships travelling through the area; there were over 10,000 more vessels transmitting AIS in the study area during July compared to January. There are better propagation conditions during winter months as the sound speed is typically the same throughout the water column due to mixing, whereas in summer, there may be a warm surface duct leading to increased downward refraction of sound waves and, therefore, increased bottom loss (Jensen *et al.*, 2011). During winter months, noise may propagate further and so building environmental factors such as temperature, and the sound speed profile, into AIS models will give an even more accurate picture of anthropogenic noise pollution in marine soundscapes.

There is concern that vessels flying foreign flags will not follow the rules and regulations of the state they are travelling through (Haren, 2007). As only 10,073 of the 458,306 vessels in the dataset covering UK waters fly the UK flag, any mitigation measures put in place through regulations may not have much impact on the overall noise exposure levels. This emphasises the need for international cooperation to combat marine noise pollution.

Although this study used historical AIS data, the model itself has the potential to be used as a predictive tool by creating different scenarios of shipping (for example, reducing the number of ships in an area in a certain month) and identifying the impacts resulting from those scenarios. This could be done small scale within a channel, or large scale throughout oceans to predict how ship noise pollution could be altered through changes to shipping routes, vessel speed restrictions, or other legislation.

3.9 Conclusion

Under the Marine Strategy Framework Directive (MSFD) Member States are required to develop strategies to achieve good environmental status of marine waters (Graaf *et al.*, 2012). This model can assist in developing conservation and mitigation strategies by highlighting locations with inhabiting species that may be subjected to the effects of shipping noise. It could also support Marine Planning in oceans (through better informed licensing applications), and aid decision-makers in determining new marine protected areas and other management strategies.

Noise hotspots and areas in need of mitigation can be easily identified and the model can be adapted to look at various future scenarios such as increased cargo shipping on the Motorways of the Sea. The model created in this study allows a year of AIS data to be turned from raw transmission messages to a usable map output in a matter of hours, and so there is potential for rapid analysis of ocean shipping noise using both historical and real-time AIS data for accurate and up-to-date noise modelling of the oceans.

There is general consensus that reducing noise exposure levels is likely to be the most effective available means of reducing impacts on marine mammals (Simmonds *et al.*, 2014). If this were also true for fish, there are several ways this could be achieved, such as avoiding noisy activities at certain times and in places where sensitive fish species are present, or by reducing the sound source itself. The AIS based model created in this Chapter has the potential to investigate different mitigation procedures to identify the most beneficial option to protect fish species from vessel noise pollution.

4. Chapter Four: Using AIS data modelling of noise emissions as a predictive tool

Chapter 1 discussed two types of mitigation to minimise the effects of noise pollution impacts; either through acquiring biological information, or by changes to the noise source. Chapter 2 focused on a method for acquiring the necessary biological information using a prioritisation framework. This chapter focuses on an application of the model built in Chapter 3 to investigate a predictive method for minimising impacts through changes to the noise source. Mapping historical trends in ocean noise is helpful, and knowing what happened in the past identifies problems, but it does not address the underlying issues or provide a solution. However, knowledge of past events does allow us to predict what may happen in the future. By utilising the model created in Chapter 3, this chapter explores the possibility of using historical AIS data to build models that can predict shipping noise emissions. Investigating different scenarios and using models to predict the outcomes can provide a method of testing different solutions to better understand whether they would work effectively if actually applied to the real world. Predictive modelling, therefore, allows researchers to test theories without affecting day-to-day activities of those who would be impacted by the suggested changes. Five possible scenarios that could arise in the future are considered, and models built to predict the likely noise levels which would prevail under such circumstances. It also demonstrates the validity of using AIS data as a predictive tool, and shows the potential of mitigating vessel noise pollution impacts through changes to the noise source.

4.1 Introduction

Current noise impact mitigation efforts are directed towards reducing the risk of injury from intense acute exposure (e.g. Koschinski and Lüdemann, 2013; Bellmann, 2014; Elmer and Savery, 2014; Whale and Dolphin Conservation, 2015). Longer-term trends and impacts from continuous noise have received substantially less attention in management, and so has become the focus of this chapter (National Research Council, 2003; Kunc *et al.*, 2016; Harris, 2017). Chapter 1 (Section 1.5), showed significantly more short-term studies were found during the literature search than long-term (Figure 1.14). Longer-term monitoring is needed to establish baselines for assessing whether mitigation measures implemented are a success (Hatch *et al.*, 2016).

Although there are two suggested types of mitigation of noise pollution impacts, for marine mammals, decreasing noise generated at the source is generally considered to be the most effective means of reducing impacts from noise (Simmonds *et al.*, 2014). No evidence was found during the literature search to suggest this may also be the case for fish species. Long-term studies on fish are lacking, and there is no certainty as to which mitigation method, using biological information or source level reduction, would be the most effective for reducing the effects of continuous vessel noise exposure. Modelling to compare the two mitigation methods can help determine which may be the more effective for fish. Current methods of reducing exposure levels are: limiting noise-producing activities at times of significant behavioural activities, near vulnerable species, and in sensitive habitats; lowering noise levels at the source; and minimising the propagation of noise emissions through the use of bubble curtains (National Park Service US, 2003; Dolman, 2007; Zielinski *et al.*, 2014; Gedamke *et al*, 2016).

Restricting areas or the amount of traffic allowed in a region is known as "geographic mitigation". Geographic mitigation occurs when vessels have to avoid areas of critical concern for certain marine species, such as those of abundance or diversity, or where topography can negatively influence noise impacts (e.g. bays, channels, or canyons where acoustic noise can become concentrated) (Tyack, 2008), and designated critical habitats (including habitats at risk of noise impacts as identified using frameworks such as the one discussed in Chapter 2). It has been previously suggested that if high noise-risk areas can be identified, mitigation could consist of year-round or seasonal restrictions of vessel traffic within a certain distance of those areas (Jasny *et al.*, 2005; Weilgart, 2007b). A year-round restriction would be difficult to introduce as it could have repercussions on human activities and without the necessary evidence it is unlikely that such policy decisions would be implemented. In such cases modelling is essential to predict the likely outcome. AIS-based models can use real ship traffic data to investigate likely noise emissions, and can be manipulated to predict future emissions. Any number of different scenarios can be addressed by changing the criteria so that the most acceptable beneficial method of mitigation can be considered.

Reducing the noise source level and the propagation ability of the noise emitted is known as source-based mitigation, and consists of reducing the output of noise straight from the source via technological advances in mechanics, and changes in vessel movements and behaviour (Jasny *et al.*, 2005). To limit vessel source level emissions ships could be required to move slower near critical areas. This does, however, correspond to longer travel times, increasing not only the duration over which noise exposure will occur, but also economic concern (Harris, 2017). As it would be difficult to persuade vessels to slow down to allow large scale research to be conducted, modelling of predicted noise emissions would be a more suitable method of research.

It has been suggested that models based on planned increased vessel movements in the Moray Firth may be able to forecast associated increases in noise exposure (Lusseau *et al.*, 2011; New *et al.*, 2013). This provides an indication that AIS-based noise mapping could be successfully applied to target ship noise mitigation efforts in other marine habitats (Merchant *et al.*, 2014b). Mathematically representing a simplified version of an environmental system, allows reasonable alternative scenarios to be predicted, tested, and compared (Yorkor, 2013). This work will further evidence the usefulness of AIS models as a method for mitigation, by using AIS data to predict vessel noise emissions on a large geographic scale.

The purpose of this work is to investigate various mitigation solutions to noise pollution impacts on fish by limiting the likelihood of initial exposure to the noise source. This chapter uses geographic and source-based mitigation to investigate possible future scenarios relating to noise emissions produced by vessels, and uses the AIS-based modelling approach created in Chapter 3 to predict source emissions in each of the following scenarios:

Scenario 1: Increased numbers of cargo vessels

This is the 'do nothing' scenario. The number of vessels on the world's oceans are predicted to increase, and, within the EU, Motorways of the Sea are becoming viable alternatives to congested roads, absorbing a significant part of the increased road freight traffic (Zhang, 2006; Danesi *et al.*, 2008; López-Navarro, 2014). The model is used to predict the likely vessel noise emissions if no mitigation is used in the future and vessel numbers continue to increase.

Scenario 2: Reduction in the number of cargo ships

This scenario investigates whether a reduction in cargo ship numbers would significantly reduce vessel noise emissions. In an ideal world, with noise pollution impact research increasingly being undertaken, policy directives may lessen the amount of road freight traffic transferring to the oceans. Politically and economically this is unlikely to happen. However, if enough evidence is collected to show the harmful effects of vessel noise on fish, it may be possible to mitigate further damage to species by reducing the number of ships in certain geographic areas or at certain times of the year by restricting cargo vessel movements and speeds.

Scenario 3: Spatial restrictions on vessels

This scenario looks at Marine Protected Area (MPA) designation. Currently, MPAs are not required to protect from noise exposure. Even though the IMO clearly acknowledges noise as a potential hazard to the marine environment, there are currently no specific laws to provide protection from anthropogenic noise exposure within MPAs. The literature review conducted for Chapter One found only one study that mentioned an MPA considering protection from noise; the Channel Islands National Marine Sanctuary Advisory Council requested research be done to identify and consider policy options to mitigate noise threats (Haren, 2007). This scenario considers likely vessel noise exposure when spatial restrictions are in place within and around MPAs, to investigate whether noise protected areas are a realistic method of mitigation.

Scenario 4: Reduction in Source Level emissions

This scenario investigates whether the use of noise-reducing technology in vessels could influence vessel noise exposure levels in the ocean. Source level noise-reduction technologies could reduce vessel noise exposure throughout the world's oceans and provide a productive mitigation measure that would be welcomed by the shipping industry (Harris, 2017).

Scenario 5: Reduction in vessel speed

This scenario investigates source-based mitigation by reducing the speed, and thus the noise emissions of a vessel. Studies have shown that a reduction in speed results in a reduction in noise emissions (McKenna et al., 2013), thereby subjecting marine species to less noise exposure. Reductions in speed when passing critical habitats, Marine protected Areas, or places where vulnerable fish species are known to inhabit would help mitigate against noise impacts but again the economic implications of longer journey times may deter such policy decisions.

4.2 Scenarios

4.2.1 Scenario 1: Increased numbers of cargo vessels

4.2.1.1 Introduction

The 2008 Climate Change Act committed the UK government to make a reduction in greenhouse gas emissions by at least 80 % (in comparison to 1990 levels) by 2050 (Department of Energy and Climate Change, 2015). Heavy use of road transportation has resulted in a significant increase in environmental impacts (Medda and Trujillo, 2010). It was reported that road traffic accounted for 68 % of total transport emissions in 2012 (Department for Transport, 2014). The substantial rise in transportation needs over recent decades is in response to changes within the framework of production systems, such as global supply sources, fragmented production chains or just-in-time systems (López-Navarro, 2014). Road networks are overloaded and traffic congestion in road network bottlenecks has the potential to act as a barrier to sustainable socio-economic development (Zhang, 2006). Intermodal freight transport has grown significantly in recent years and is destined to play a significant role in the future (Macharis *et al.*, 2011).

To comply with current clean air policies, an alternative mode of transport has been established. The European Commission introduced the concept of Motorways of the Seas (MoS) in the 2001 Transport Policy White Paper as a policy instrument to rebalance usage of transport modes and focus on intermodal transport development. The UK has more than 40,000 kilometres of coastline but the sea is a largely underused transport resource. Sea transport offers effective routes to bypass natural barriers (e.g. mountains ranges such as the Alps), sometimes providing shorter, or quicker routes, throughout Europe (Zhang, 2006). Through the establishment of frequent and high quality maritime-based logistics services between Member States, Motorways of the Sea are becoming viable alternatives to congested roads, absorbing a significant part of the expected increase in road freight traffic (Zhang, 2006; Danesi *et al.*, 2008; López-Navarro, 2014).

The EU transport policy supports the development of the Motorways of the Sea system as part of 30 priority projects of the Trans European Transport Network (TEN-T) (European Parliament, 2004). The concept aims to introduce new, integrated intermodal maritime-based logistics chains with links that connect a limited number of selected ports located at strategic points on European coastlines (Kapros, 2010). The concept is that these chains will be more sustainable and commercially more efficient than road transport (European Commission, 2007a).

The original plan for Motorways of the Sea (MoS) included the development of four MoS (Figure 4.1): Baltic Sea; western Europe (Atlantic Ocean – North Sea/Irish Sea); south-western Europe (western Mediterranean Sea); and south-eastern Europe (Adriatic, Ionian and eastern Mediterranean Seas). However, many more projects have been added to the programme which is still ongoing.

One of the reasons the European Commission has highlighted the need for a shift from road to sea transport is the issues associated with road transport and CO_2 emissions (López-Navarro, 2014). General perceptions are that sea transport is environmentally better than road transport, but some studies have questioned this assumption (European Environmental Bureau, 2004; European Environment Agency, 2008).

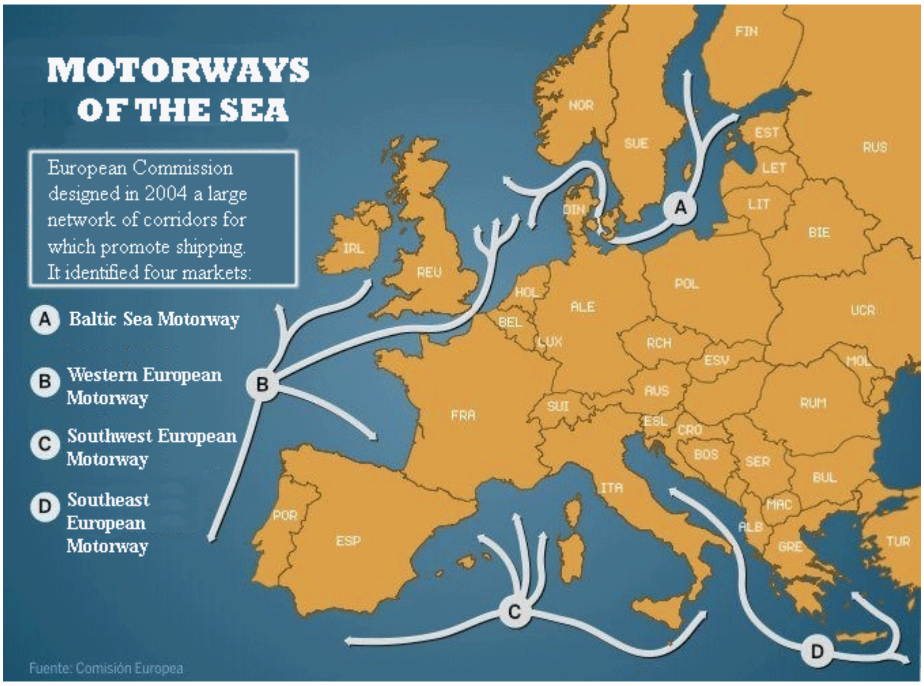


Figure 4.1. Motorways of the Sea. In 2004 the European Commission identified 4 major shipping trade routes and designed a network of vessel corridors to facilitate shipping movements. See European Parliament (2004) report for information on original MoS plans. Image provided by the European Commission Mobility and Transport (*n.d.*).

The oceans are not considered the same as road and rail infrastructure, which is likely due to the mistaken assumption by policymakers that the ocean is a free highway, and, therefore, does not deserve public subsidy in the same way (Baird, 2007). This means that the oceans have not benefitted from monetary subsidies, research, or environmental considerations as road and rail have previously done. Environmental aspects are usually disregarded in studies analysing and comparing road and sea transport on different routes. This is surprising given the fact that environmental arguments are usually presented as decisive in justifying intermodal solutions (López-Navarro, 2014). Research on the advantages and disadvantages of Motorways of the Seas has covered traffic, operator costs and benefits, transit times, reliability and frequency, tolls, congestion mitigation, sensitivity analysis and air pollution, but very little has been done on the impacts of noise pollution (Mange, 2006; Hjelle, 2010; Vanherle and Delhaye, 2010; López-Navarro, 2014). Zhang (2006) stated in his report than the negative impacts of road traffic on the environment (noise pollution and CO_2 emissions) can be reduced if cargo transport moved from terrestrial networks to the oceans. However, he, as well as many other researchers, failed to consider the fact that noise pollution impacts on aquatic life is an important issue in the oceans; many cost-benefit analysis papers do not even mention noise pollution (e.g. Mange, 2006; Zhang, 2006; Hjelle, 2010; (Mange, 2006; Hjelle, 2010; Vanherle and Delhaye, 2010; López-Navarro, 2014).

An increase in vessel traffic will only compound the already existing problems associated with noise pollution in our oceans (Harris, 2017). The noise impacts for fish populations by creating more MoS, and from the increasing traffic on the already existing MoS, is still unknown. The use of predictive modelling using AIS-based models to provide useful and accurate predictions of vessel noise emissions to indicate the likely outcomes of future noise exposure on marine fish species is investigated in this scenario.

This particular scenario aims to examine the impacts of increasing numbers of cargo vessels. A scenario that is very likely to happen in the coming decades. An AIS-based modelling approach is used to simulate the potential changes in noise exposure that might occur if the number of cargo vessels increased. The model uses historical AIS data to predict future noise emissions and highlight the potential impacts of vessel noise exposure on fish.

4.2.1.2 Method

An ocean vessel cumulative noise exposure (SELcum) map was built using the AIS data modelling method created in Chapter 3. The data used was from satellite and terrestrial AIS data (provided by Orbcomm) within latitudes 40 °N & 65 °N and longitudes 20 °W & 12 °E for 2013. Only cargo vessels were used in this scenario; all non-cargo vessels were removed from the dataset. AIS data reports the vessel type as part of the transmission message and so a Java programme was written to collect all cargo MMSI numbers from the vessel database (created as part of Chapter 3), producing a list of only cargo vessels. The full AIS dataset was then mined for transmissions from ships matching the cargo MMSI numbers, using a Bash AWK – a text processing programming language – script, and a new cargo subset of AIS data was created. The MMSI list was then adjusted to increase the vessels in the study area by 10 %, 20 % and 30 %. This was accomplished by running a programme that randomly added a percentage of MMSI numbers to the existing MMSI dataset (equating to 13,994 extra vessels for each 10 %). The extra MMSIs were duplicates of already existing MMSI numbers so that real AIS transmission were available for each of the new vessels resulting in real calculated source levels being produced. This way, no data is fabricated so calculated Source Levels and subsequent Sound Exposure Levels remains realistic. All other data such as speed, vessel size, MMSI and calculated noise emissions were not altered from the actual data. The modelling method reported in

Chapter 3 was then used to produce 1 km grid cell Sound Exposure Levels and a heatmap style visual map output. The model was run four times, once with the actual data collected via AIS receivers, and then again with the simulated data for each percentage increase.

4.2.1.3 Results

Cumulative Sound Exposure Levels increased with increasing Cargo vessel activity. It is clear from the results that cargo vessel emissions are widespread throughout the oceans, but there are common routes travelled by cargo vessels between ports (Figure 4.2). The noise exposure of the study area ranged from 80 to 194 dB re 1 μPa^2s. The green areas on the map output represent noise exposure levels of up to approximately 130 dB re 1 μPa^2s, with yellow areas reaching 150 dB re 1 μPa^2s and orange to red areas representing noise exposure levels of 150 dB re 1 μPa^2s or louder (Figure 4.2 and Figure 4.3). Noise exposure levels rose as the numbers of active cargo vessels increased. This was as hypothesised, however, the rise in noise exposure levels of less than 1 dB for each 10 % increase was lower than expected.

Cargo noise exposure is concentrated on the Motorways of the Sea (MoS), reaching cumulative levels of 184 dB re 1 μPa^2s in the English Channel and Baltic Sea over 2013 (Figure 4.2). After an increase of active cargo vessels of 30 % these same areas reach levels of 186 dB re 1 μPa^2s (Figure 4.3).

Overall year SELcum for the whole study area (combining all grid cells) was 199.36 dB re 1 μPa^2s for the non-manipulated data (0 % increase) and 200.60 dB re 1 μPa^2s for the 30 % increase scenario (Figure 4.4). Peak noise exposure levels changed by just 2 dB re 1 μPa^2s between the 0 % and 30 % increase simulations. The maximum noise exposure level output from the model was 194 dB re 1 μPa^2s at 125 Hz for the 30 % increase simulated scenario, and 192 dB re 1 μPa^2s at 125 Hz for the real 2013 dataset. Neither of the values are visible on the map and so must relate to a very small geographic area. To identify small hotspots a finer cell output scale would be needed during the interpolation, which causes a trade off with computational power and time.

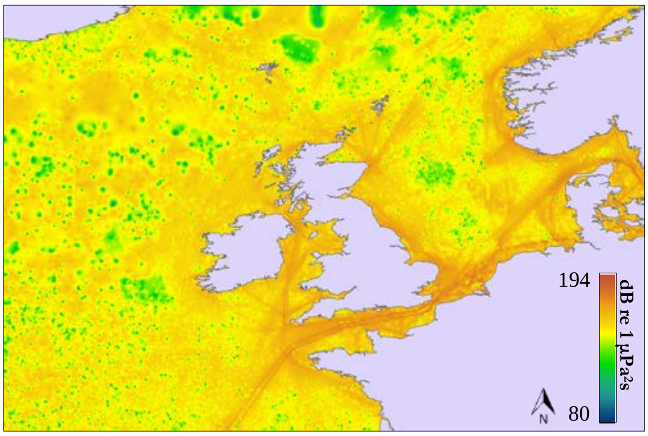


Figure 4.2. Cumulative Sound Exposure Levels (dB re 1 μPa^{2}s) of vessel noise emissions at 125 Hz throughout 2013. Noise levels in the individual grid cells range from 80 dB re 1 μPa^{2}s (blue) to 194 dB re 1 μPa^{2}s (red). The noise levels are the cumulative SEL over 1 km grid squares, with inverse distance weighting to create a smooth map output. The frame stretches in longitude from 20 °W to 12 °E and in latitude from 44 °N to 65 °N.

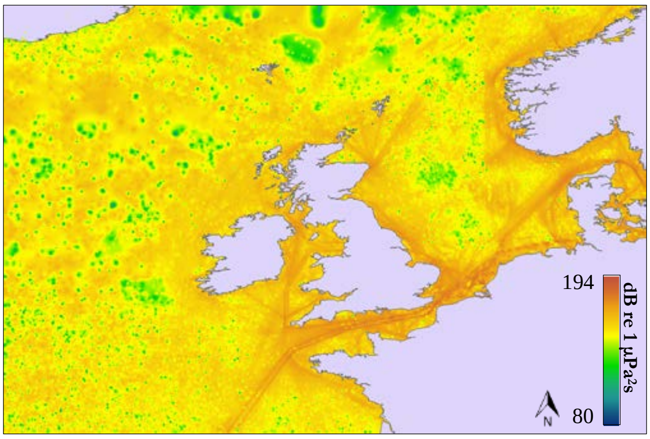


Figure 4.3. Cumulative Sound Exposure Levels (dB re 1 μPa^{2}s) of vessel noise emissions at 125 Hz throughout 2013 for a scenario with 30 % more cargo vessels active on the ocean. Noise levels in the individual grid cells range from 80 dB re 1 μPa^{2}s (blue) to 194 dB re 1 μPa^{2}s (red). The noise levels are the cumulative SEL over 1 km grid squares, with inverse distance weighting to create a smooth map output. The frame stretches in longitude from 20 °W to 12 °E and in latitude from 44 °N to 65 °N.

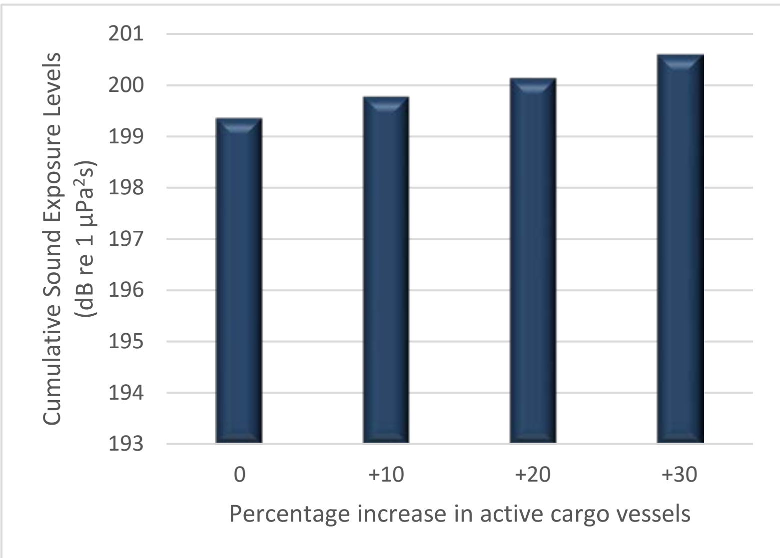


Figure 4.4. Year cumulative Sound Exposure Levels for cargo vessels in 2013 (0), and predicted SELcum values if the number of cargo vessels active on the ocean increases by 10 %, 20 % and 30 %. These values represent the cumulative noise exposure of all grid cells combined (not individual cells as depicted in the map outputs). Data is for a study area stretching from 20 °W to 12 °E and in latitude from 44 °N to 65 °N.

4.2.1.4 Discussion

Using cargo ships is considered an economically sound and efficient means of transporting large quantities of goods over long distances and is being encouraged as it is viewed as the environmentally friendly option over road and rail freight (U.S. Department of Transportation Maritime Administration, *n.d*).

As freight transport on the ocean is becoming the preferred method over road and rail options the numbers of cargo vessels and cargo activity is likely to increase in coming decades. Modelling this scenario allows predictions to be made about future vessel noise exposure levels, the likely increase in intensity level, and the geographic areas that will experience greater effects and potentially be a more harmful environment for marine species. An increase of active cargo vessels of 30 % caused some large areas, such as the English Channel, to reach levels of 186 dB re 1 $\mu Pa^2 s$, which is a 6 dB rise from the original data from 2013 (Figure 4.2 and Figure 4.3).

The overall year cumulative Sound Exposure Level showed a rise of less than 1 dB with a 30 % increase in cargo activity (Figure 4.4). Although some areas saw definite noise level increases, overall, there is very little difference between the simulation results. It can be inferred from the results that as cargo activity increases so will vessel noise exposure levels, though perhaps not to the levels that some people may assume. It is important to remember though that this scenario only considered cargo vessels increasing, but realistically there will be increases in several vessel types each year. Cargo was chosen specifically because it was the loudest contributor to annual cumulative Sound Exposure Levels and made up 30 % of the AIS dataset for 2013. It was hypothesised, therefore, that cargo would have the largest impact on overall noise exposure levels.

AIS data is transmitted from various vessel types (*see* Table 3.4 for the vessel types) and so is not limited to analysing trends in cargo vessel noise. The model used here only needs latitude and longitude data and some information about the vessel characteristics such as reference speed and type. This information can be collected in other ways as well as through AIS data collection, such as Vessel Monitoring Systems on fishing vessels or aggregate data from dredgers, or even through the collection of GPS data from satellite navigation or mobile phones. Data can be easily collected, especially for smaller geographic areas, and so modelling provides a useful tool for anyone wanting to look at trends or make predictions about vessel activity or noise exposure levels.

4.2.2 Scenario 2: Reduction in the number of cargo ships

4.2.2.1 Introduction

With the number of cargo vessels on the increase, and with the plan to move more freight transport from the roads to the sea, the likelihood of reducing the number of cargo vessels active on the ocean is remote. However, if it were possible, it may prove a beneficial mitigation measure to reduce vessel noise exposure as the cargo vessels made up the largest percentage of ocean vessels in the AIS data for 2013 within latitudes 40 ºN & 65 ºN and longitudes 20 ºW & 12 ºE (Table 3.4). Cargo vessels also produced the loudest Sound Exposure Levels during 2013 compared to other vessel classifications, reaching a cumulative level of 199 dB re 1 $\mu Pa^2 s$ (Figure 3.7), and so it was hypothesised that a reduction in cargo vessel activity will have a positive influence on the risk of vessel noise exposure for fish.

This scenario replicates the above scenario but with the object of reducing the number of cargo vessels by 10 %, 20 %, and 30 %. The results are useful in that they will reflect how cargo vessel noise exposure could alter if there were less cargo activity occurring. The ability to predict such trends, even if they are thought unlikely to occur, is useful for decision makers; if modelling of vessel noise exposure shows a reduction in noise levels when vessel activity is reduced it may provide incentive to work towards making that mitigation measure possible, even if only in smaller very vulnerable geographic areas.

4.2.2.2 Method

An ocean noise map was built using the AIS data modelling method created in Chapter 3. The data used was from satellite and terrestrial AIS data (provided by Orbcomm) within latitudes 40 ºN & 65 ºN and longitudes 20 ºW & 12 ºE for 2013. Only cargo vessels were used in this scenario; all non-cargo vessels were removed from the dataset. A list of cargo vessels was collected using the same method as Scenario 1. The MMSI list was adjusted to decrease the number of vessels in the study area by 10 %, 20 % and 30 %. This was done by running a programme that randomly removed 10 % of MMSI numbers from the MMSI dataset (equating to 13,994 less vessels). All other data such as speed, vessel size, MMSI and calculated noise emission were not altered from the actual data. The modelling method reported in Chapter 3 was then used to produce 1 km grid cell Sound Exposure Levels and a heatmap style visual map output, but 13,994 of the vessels would be ignored and left out of the model for each 10 %

decrease. The model was run four times, once with the actual data collected via AIS receivers, and again with the simulated data for each percentage decrease.

4.2.2.3 Results

The noise exposure levels within the study area ranged from 80 to 194 dB re 1 μPa^2s (Figure 4.5 and Figure 4.6). The green areas on the map output represent noise exposure levels of up to approximately 130 dB re 1 μPa^2s, with yellow areas reaching 150 dB re 1 μPa^2s and orange to red areas representing noise exposure levels of 150 dB re 1 μPa^2s or louder. Noise exposure levels decreased as the numbers of active cargo vessels reduced. This was as hypothesised, however, as with the increased cargo scenario, the reduction in noise exposure levels was lower than expected, dropping from 199.36 dB re 1 μPa^2s to just 197.62 dB re 1 μPa^2s for a 30 % decrease in vessel activity (Figure 4.7).

On the Motorway of the Sea (MoS) through the English Channel, cumulative noise exposure levels reached 184 dB re 1 μPa^2s in 2013 (Figure 4.5). A decrease of active cargo vessels of 30 % only reduced these levels to a peak of 182 dB re 1 μPa^2s (Figure 4.6).

Vessel transmissions collected reduced by approximately 1.4 to 2 million for each 10 % decrease in vessel activity (Figure 4.8). Activity for each month was reduced by 10 % and so the activity per month did not change between simulations, so as not to influence temporal patterns.

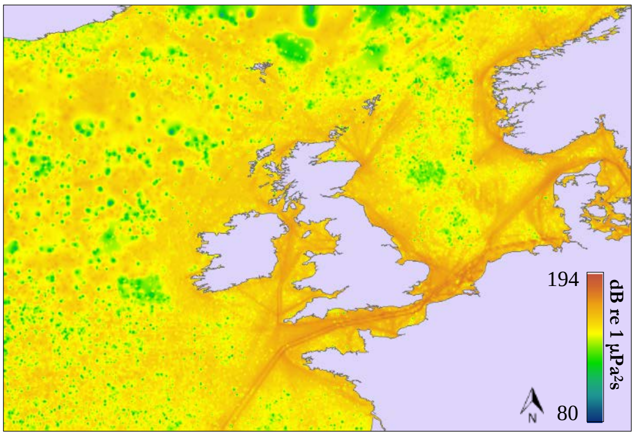


Figure 4.5. Cumulative Sound Exposure Levels (dB re 1 μPa^{2}s) of vessel noise emissions at 125 Hz throughout 2013. Noise levels in the individual grid cells range from 80 dB re 1 μPa^{2}s (blue) to 194 dB re 1 μPa^{2}s (red). The noise levels are the cumulative SEL over 1 km grid squares, with inverse distance weighting to create a smooth map output. The frame stretches in longitude from 20 °W to 12 °E and in latitude from 44 °N to 65 °N.

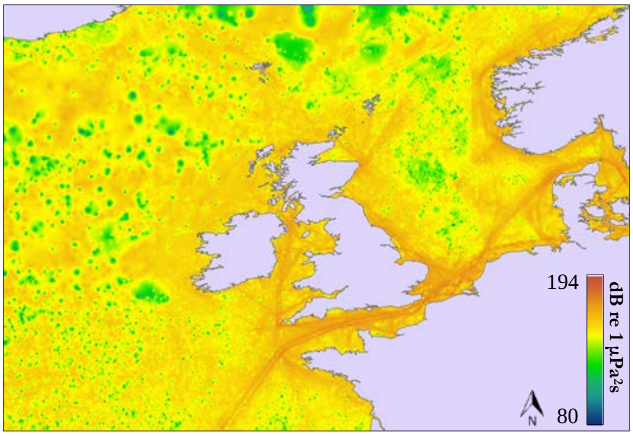


Figure 4.6. Cumulative Sound Exposure Levels (dB re 1 μPa^{2}s) of vessel noise emissions at 125 Hz throughout 2013 for a scenario with 30 % fewer cargo vessels active on the ocean. Noise levels in the individual grid cells range from 80 dB re 1 μPa^{2}s (blue) to 194 dB re 1 μPa^{2}s (red). The noise levels are the cumulative SEL over 1 km grid squares, with inverse distance weighting to create a smooth map output. The frame stretches in longitude from 20 °W to 12 °E and in latitude from 44 °N to 65 °N.

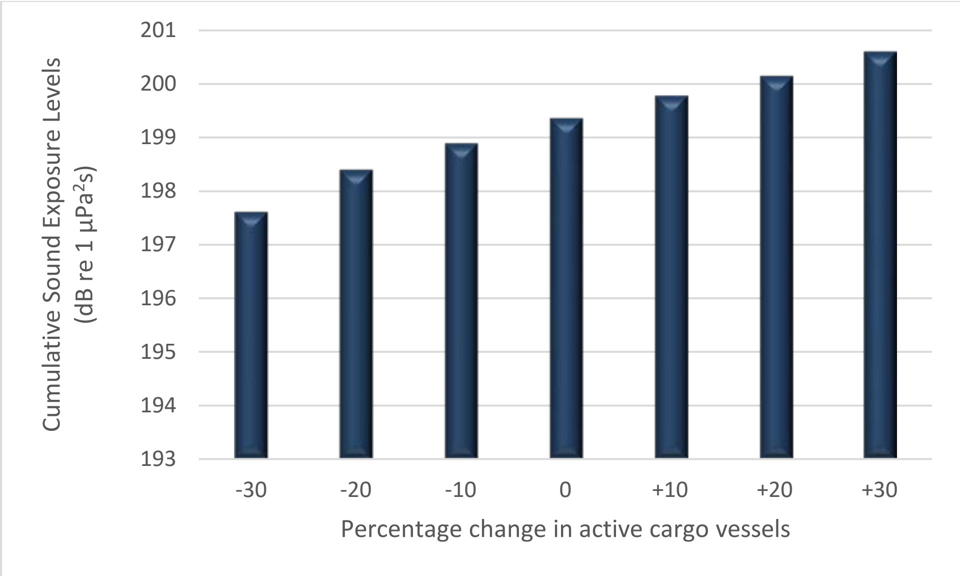


Figure 4.7. Year cumulative Sound Exposure Levels for cargo vessels in 2013 and all potential scenario simulations from a decrease of 30 % vessel activity to an increase of 30 % vessel activity (including those from Section 4.2.1 so all cargo trends could be easily compared). These values represent the cumulative noise exposure of all grid cells combined (not individual cells as depicted in the map outputs). Data is for a study area stretching from 20 °W to 12 °E and in latitude from 44 °N to 65 °N.

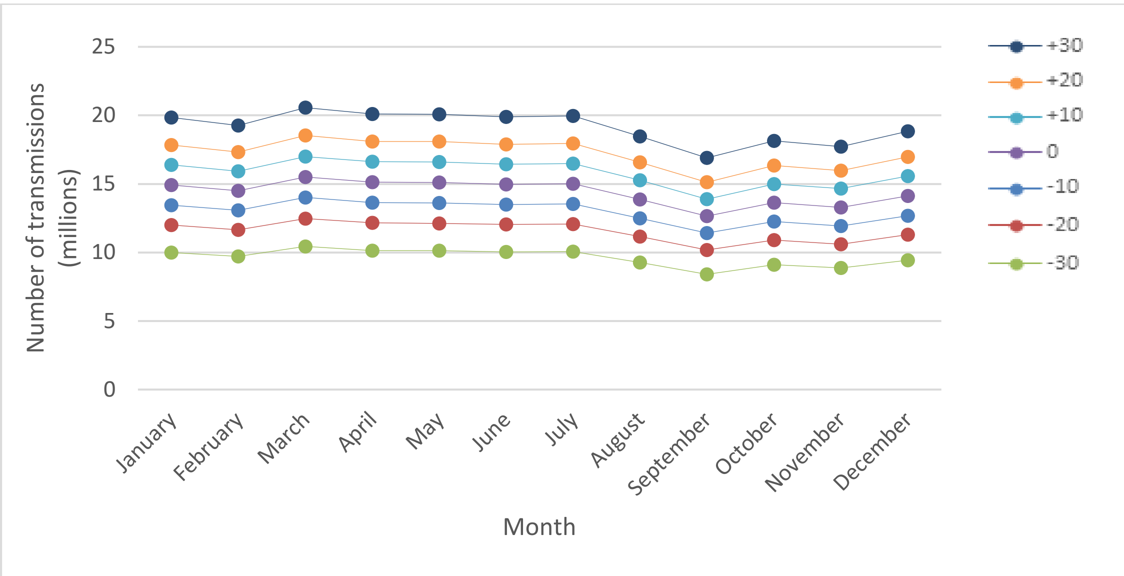


Figure 4.8. Number of transmissions collected for cargo vessels each month of 2013 in the different scenario simulations (including those from Section 4.2.1). Data is for a study area stretching from 20 °W to 12 °E and in latitude from 44 °N to 65 °N. The number of transmissions reduced by approximately 1.4 to 2 million with each 10 % change in vessel activity. Transmissions are collected every 12 minutes of a vessels journey.

4.2.2.4 Discussion

Vessel operation statistics indicate steady growth in vessel traffic over the past few decades, and with the movement of freight transport from the roads to the motorways of the sea this trend is unlikely to reverse (U.S. Department of Transportation Maritime Administration, *n.d.*). Reducing the cargo activity on the oceans would be difficult, especially with the clean air policies pushing for greater use of Motorways of the Sea. However, as cargo vessels contribute greatly to vessel activity on the oceans and are perhaps the noisiest vessel type with Source Levels of up to 190 dB re 1 $\mu Pa^2 \cdot m^2$ reducing the levels could have a large impact on the noise exposure levels faced by marine species. The shipping industry would require strong evidence that a reduction in cargo activity would significantly lower noise levels to below, or closer to, known harmful levels. Without large scale and expensive field studies, the best way to produce such evidence is through the use of models such as this one.

The results of this model, however, confirm that, for the UK, reducing cargo vessel activity would not greatly reduce the noise exposure levels in the marine environment and so would not be worth considering as a future mitigation measure. A 30 % decrease in cargo activity across the study area resulted in a less than 2 dB re 1 $\mu Pa^2 s$ reduction in overall noise exposure. This decrease was much lower than hypothesised and shows that the costs to the shipping industry of reducing freight transport would outweigh the small benefit of reduced noise exposure for marine species.

As vessel removal was randomised, there was no way to know which vessels were removed. More systematic removal of a certain percentage of vessels, such as the largest, or the fastest, would provide additional information to guide mitigation. Future work could involve more varied and complex scenarios such as a decrease in large cargo vessels but with an increase in small cargo vessels to test whether noise exposure could be reduced without the need to reduce overall freight transport.

Alternative mitigation methods such as reducing the Source Level of vessels or rerouting cargo vessels away from fish populations may be better options.

4.2.3 Scenario 3: Spatial restrictions on vessels

4.2.3.1 Introduction

A protected area is defined by the International Union for Conservation (IUCN) as "a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (International Union for Conservation of Nature, 2008). A Marine Protected Area (MPA) is defined by the IUCN as 'Any area of the intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment' (Lockwood *et al.*, 2006).

Marine Protected Areas (MPAs) in the UK consist of:

- Special Areas of Conservation (SAC)
- Special Protected Areas (SPA)
- Areas of Special Scientific Interest (ASSI)
- Sites of Special Scientific Interest (SSSI)
- Nature Conservation Areas
- Ramsar sites
- Marine Nature Reserves
- Marine Conservation Zones (MCZ)
- OSPAR MPAs

MPAs provide protection for not only the species inhabiting the area but the habitat and environment itself. They are designed to conserve both marine biodiversity (species and habitats) and geodiversity (the variety of landforms and natural processes that underpin the marine landscapes), offering long-term support for the services our seas provide to society (The Scottish Government, 2015). Establishment of new MPAs is ongoing in order to reduce the overexploitation of marine resources, improve fisheries management and preserve biodiversity (Boersma and Parrish, 1999; Agardy, 2000).

The Agreement on the Conservation of Cetaceans of the Black Sea, Mediterranean Sea and Contiguous Atlantic (ACCOBAMS) (2009) report and Hoyt (2005) both claim that the designation of SACs and MPAs can be used to protect marine mammals and their habitats from environmental stressors including the cumulative and synergistic effects of noise, and if well designed and managed, MPAs could play a key role in the conservation of cetaceans and marine ecosystems. This should also hold true for fish species.

Research and monitoring are essential tools in MPA management (Cicin-Sain and Belfiore, 2005). Modelling tools such as map-based indicators can be invaluable when making decisions about MPA designation or restoration (Bryant *et al.*, 1998). Such tools use biological response metrics of exposure to detect and quantify the impacts of a stressor allowing for accurate prediction and consequent decision-making (Jameson *et al.*, 2002). Within the designated areas noise levels must not rise above ambient levels considered tolerable for the species present. This ambient level would include cumulative impacts propagating in from outside the designated areas (Agreement on the Conservation of Cetaceans of the Black Sea Mediterranean Sea and Contiguous Atlantic, 2009). To do this the ACCOBAMS (2009) report stated the requirement of additional research to establish baseline noise data and evaluate thresholds for noise levels that can be considered acceptable or tolerable without harm. Noise maps, such as the ones created in Chapter 3, have a definite role to play here.

Species listed in Annex II of the European Union Habitat Directive (such as Atlantic salmon) require Special Areas of Conservation (a form of Marine Protected Areas) to be set up and designated as a key tool for species conservation and protection (Joint Nature Conservation Committee, 2015). In the late 1990s evidence emerged to show that Marine Protected Areas improve fish yields, and at the same time conserve biological diversity (Jennings and Kaiser, 1998). However, since those studies were published vessel density has increased dramatically so the question must be asked as to whether MPAs are still as efficient at protecting the marine environment.

Marine Protected Areas (MPAs) are arguably one of the most powerful tools available to combat ever-increasing overexploitation and loss of marine resources (Agardy *et al.*, 2011). However, people have a blind faith in the ability of MPAs to succeed, but MPAs are not always properly planned or thought-out (Agardy *et al.*, 2011). The great majority of marine protected areas (MPAs) fail to meet their management objectives (Jameson *et al.*, 2002). When subjected to numerous uncontrollable external stressors the environment will become degraded and the MPA will not be effective. There are 5 shortcomings of designated Marine Protected Areas (Table 4.1), which can lead to failure of MPAs.

Table 4.1. List of shortcomings of Marine Protected Areas as stated by Agardy *et al.* (2011)

MPA shortcomings
• MPAs that by their small size or poor design are ecologically insufficient
• MPAs may be inappropriately planned or managed
• MPAs fail due to the degradation of the unprotected surrounding ecosystems
• MPAs can do more harm than good due to displacement and unintended consequences of management
• MPAs can create a dangerous illusion of protection when in fact no protection is occurring

As many MPAs are situated along coastlines, within shipping lanes, and near anthropogenic activities the occurrence of chemical and biological pollution is high (Boersma and Parrish, 1999), resulting in many MPAs being affected by human activities that lie outside their boundaries, ranging from marine transportation and fishing to land-based sources of marine pollution, e.g., agriculture, urban runoff, and industry (Cicin-Sain and Belfiore, 2005). Jameson *et al.* (2002) expressed the view that if MPAs are not really protected from uncontrollable sources of pollutants then they should not be designated as protected areas. Noise pollution is now considered a serious threat to marine animals as its effects are less perceptible than other pollutants such as oil spills, and it cannot be easily stopped or confined to the outside of sensitive or protected areas (Codarin *et al.*, 2009). Underwater noise is not restricted by national boundaries and although the Water Framework Directive and Marine Strategy Framework Directive are being developed there is currently no specific policy to control its impact.

On a local scale, designation of an MPA can improve point sources of pollution by regulating discharge from coastal and vessel sources (Allison *et al.*, 1998). However, because of the trans-boundary nature of noise, local prevention is less effective than large-scale regulation. To combat such widespread pollutants it has been proposed that reserve size should be 50-90 % of the habitat to be protected (Boersma and Parrish, 1999), a rather unrealistic scenario due to the costs and trade-offs involved. The designated MPAs around the UK are all relatively small with the furthest point from a boundary being less than 200 km (Figure 4.9). Noise has been known to propagate thousands of kilometres so it is probable that there is no quiet area within any of the MPAs in the UK.

Predictive modelling can help identify whether the hypothesis that MPAs protect vulnerable fish species is true, and if not, how much larger an MPA would need to be to mitigate the effects of noise pollution. A field study to identify ideal reserve size is highly improbable as controlling the boundaries of an MPA would be difficult to enforce and would be met with criticism from the transport and fishing industries.

An advantage of modelling is that many different scenarios can be run in a relatively short period of time, and without impacting everyday vessel movements. MPA boundaries can be (theoretically) moved without the need for changing legislation or incurring costs, and to extremes that may be unrealistic to adopt.

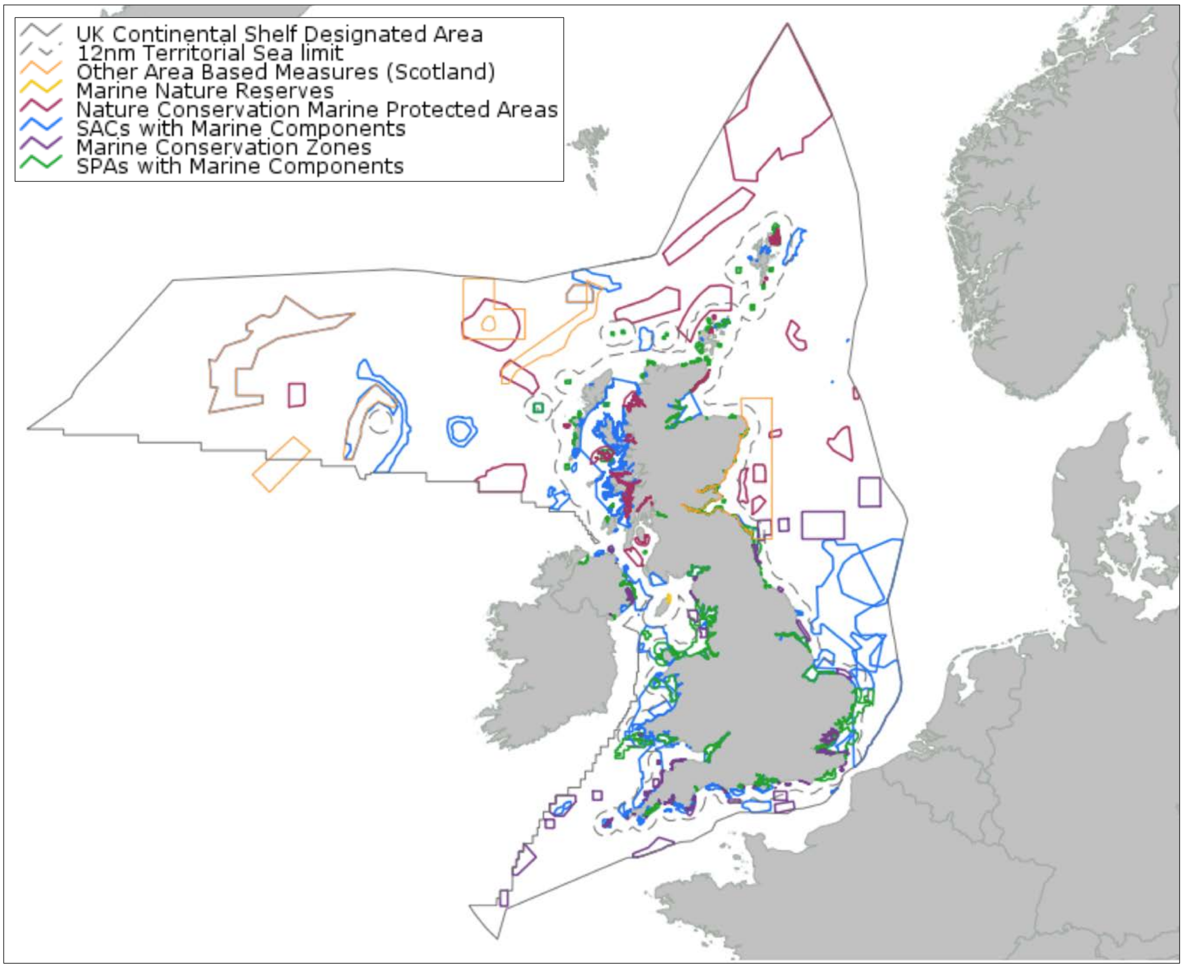


Figure 4.9. Map of Marine Protected Areas around the UK. Picture sourced from the Joint Nature Conservation Committee (Joint Nature Conservation Committee, 2002).

As mentioned throughout Chapter 1, noise can have both physiological (*see* Section 1.4.1) and behaviour (*see* Section 1.4.2) impacts on many fish species (see Table 1.4 for summary or Table A1.1 to full details). Ocean noise can affect animal behaviour and disrupt trophic linkages (Williams *et al.*, 2015c). Masking of vocalisations and acoustic cues due to shipping noise has been observed to occur in some species (Table 1.3), and this can disrupt their ability to navigate, find food, avoid predators, and choose mates successfully (Table 1,2).

Marine Protected Areas are a useful conservation tool but their efficiency at protecting from noise pollution is questionable. Noise can propagate hundreds or thousands of kilometres in the marine environment and so the likelihood of the small-sized Marine Protected Areas in the UK protecting its inhabitants from noise pollution is improbable. This scenario investigates the levels of noise pollution likely to be present in UK Marine Protected Areas and looks at what happens to the overall noise intensity if ships are removed from the area/surrounding area. This scenario could help in the delineation of noise protected areas in the future, and provide evidence on whether noise protection is feasible within limited MPA boundaries.

Currently, there are few legislative tools that can be used to conserve marine biodiversity throughout all of the oceans' biogeographic zones, particularly the high seas (Boersma and Parrish, 1999). MPAs are an established policy tool and are important because within their boundaries are ecologically rich, and often critical habitat, areas for marine mammals and fish (Haren, 2007). They should, therefore, be safeguarded from anthropogenic noise. The aim of this scenario was to use AIS modelling to determine whether vessel noise emissions within MPAs and the surrounding non-controlled areas is propagating at harmful levels into areas designated as protected, and how these noise levels can be reduced by altering shipping movements within the MPA and surrounding areas.

4.2.3.2 Method

AIS transmissions from satellite and terrestrial AIS data (provided by Orbcomm) covering all vessel types within latitudes 40 ºN & 65 ºN and longitudes 20 ºW & 12 ºE during 2013 was used to produce a map of UK waters. A map of UK Marine Protected Areas (Joint Nature Conservation Committee, 2002) was used to identify MPAs that could be easily input into the model to predict the influence vessel traffic within MPA boundaries has on the cumulative noise exposure. Two MPAs were identified, Swallow Sand Marine Conservation Zone (MCZ) (latitudes 55.99232 ºN & 55.49139 ºN, and longitudes 0.00998 ºW & 1.34134 ºE), and Fulmar Marine Conservation Zone (latitudes 56.07825 ºN & 56.60192 ºN, and longitudes 1.83573 ºE & 2.52787 ºE) (Figure 4.10). Swallow Sand is situated approximately 100km offshore from the Northumberland coast. It is a relatively large site in terms of Marine Conservation Zone designations, with an area of 4,746 km^2 (Joint Nature Conservation Committee, 2016). Fulmar Marine Conservation Zone is approximately 224 km from the Northumberland coast and is designated for its important spawning and feeding grounds. Both these MPAs support high numbers of commercial species such as mackerel and sprat (The Wildlife Trusts, *n.d.*). These MPAs were chosen because their square boundary makes it easy to remove AIS transmissions from within the area. Additionally, because of their close proximity they can be combined to make a larger MPA to indicate whether size is important when considering new MPA designation in terms of reducing noise pollution impacts. To join the two MPAs into one combined MPA latitudes 55.49139 °N & 56.60192 °N and longitudes 0.00998 °W & 2.52787 °E were used. The AIS data was manipulated to remove all vessel transmissions from within each of the MPA boundaries using a Java programme that extracts all the AIS data outside of the boundary (Figure 4.11), and the model was run to produce noise exposure maps for Swallow Sand, Fulmar and the combined MPA.

Figure 4.10. Location of the MPAs used in this study. Swallow Sand is highlighted with a white border, Fulmar with a yellow border and the combined MPA with a red border.

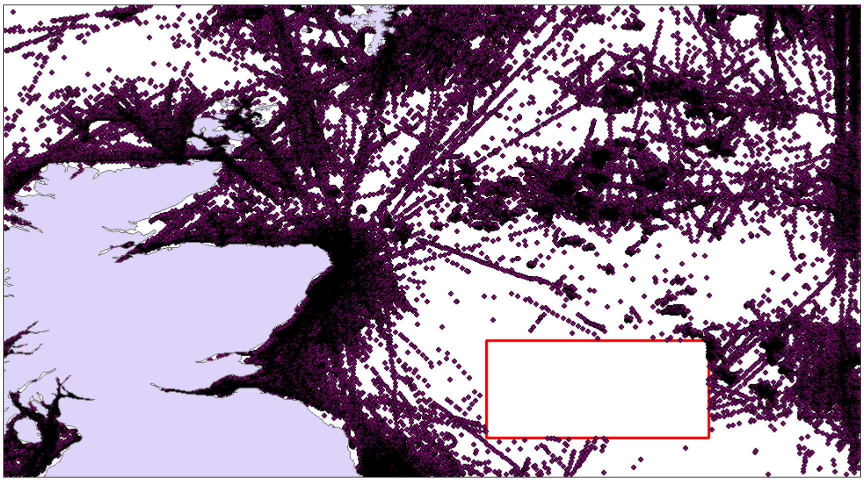
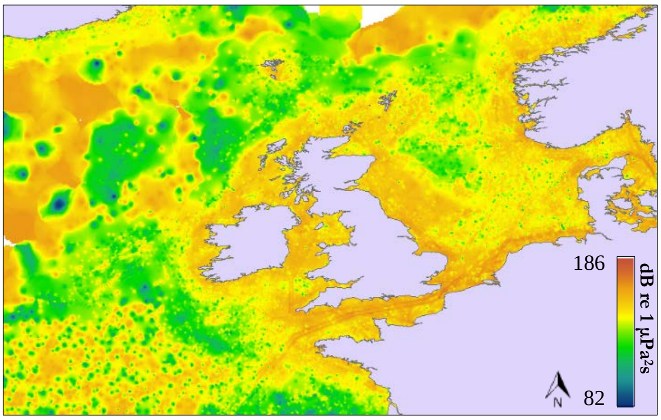


Figure 4.11. Example of dataset after vessels within the MPA boundary were removed. The red outline shows the boundary of the combined MPAs (Swallow Sand and Fulmar). All transmissions from within the boundary were removed using a Java programme.

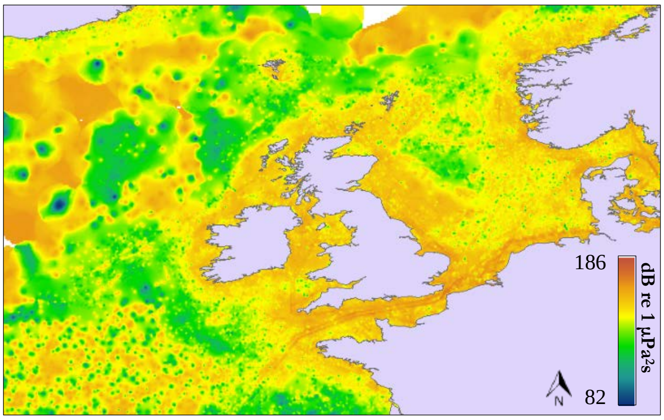
4.2.3.3 Results

Noise exposure levels in the study area ranged from 82 to 186 dB re 1 µPa^{2}s, with the green areas on the map output representing noise exposure levels of up to approximately 130 dB re 1 µPa^{2}s, with yellow areas ranging from 130 dB to 145 dB re 1 µPa^{2}s and orange to red areas represent noise exposure levels of 145 dB re 1 µPa^{2}s or louder (Figure 4.12). Geographic mitigation did influence the vessel noise exposure levels by a reduction of up to 26 dB re 1 µPa^{2}s, however, owing to the small MPA area it is not easily visible on the map output. For this reason, the interpolation was rerun at a finer cell output scale (Figure 4.13). To help identify the location of the MPA an outline was added to the map. For Fulmar MCZ cumulative noise exposure levels inside the boundary ranged from 110 to 162 dB re 1 µPa^{2}s. After vessels were removed from the area, these levels dropped to a range of 101 to 155 dB re 1 µPa^{2}s, with all SELcum levels of 140 dB re 1 µPa^{2}s or more occurring at the boundary (Figure 4.13). Within Swallow Sand MCZ cumulative noise exposure levels ranged from 104 to 165 dB re 1 µPa^{2}s, with the majority of the area at 155 dB re 1 µPa^{2}s. When no vessels were allowed inside the boundary noise exposure ranged from 106 to 160 dB re 1 µPa^{2}s with the majority of the area at 150 dB re 1 µPa^{2}s (Figure 4.14). The combined MPA cumulative noise exposure levels ranged from 100 to 165 dB re 1 µPa^{2}s. With vessels removed, the combined MPA exposure levels ranged from 106 to 160 dB re 1 µPa^{2}s, but nearly half of the reserve was at levels below 140 dB re 1 µPa^{2}s (Figure 4.15).

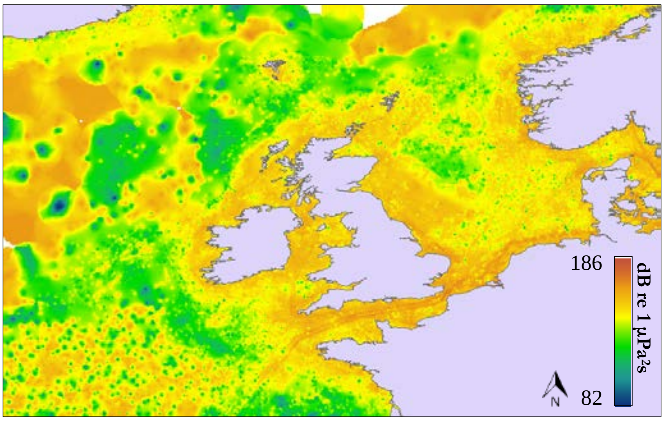
Removing vessels from within an MPA can influence the noise exposure inside the boundaries and so geographic mitigation could help reduce vessel noise impacts on fish. The larger the reserve the greater the area of lower intensity noise. The amount of noise reduction also seems to depend on the location of the MPA; Swallow Sand is situated in a high vessel traffic area and saw less noise reduction than Fulmar or the combined MPA.



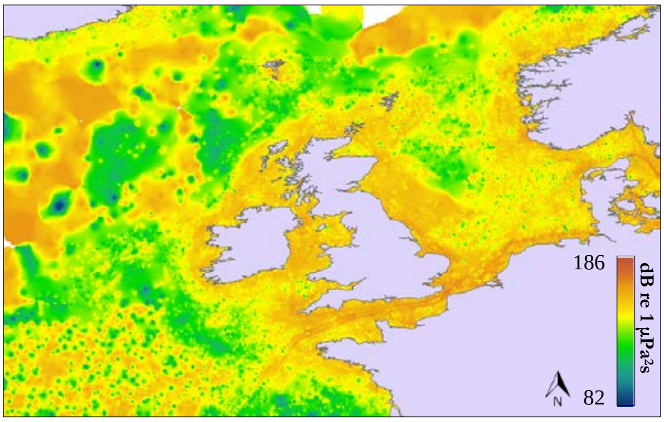
(a) No vessel restricted MPAs



(b) Fulmar MCZ with vessel restrictions

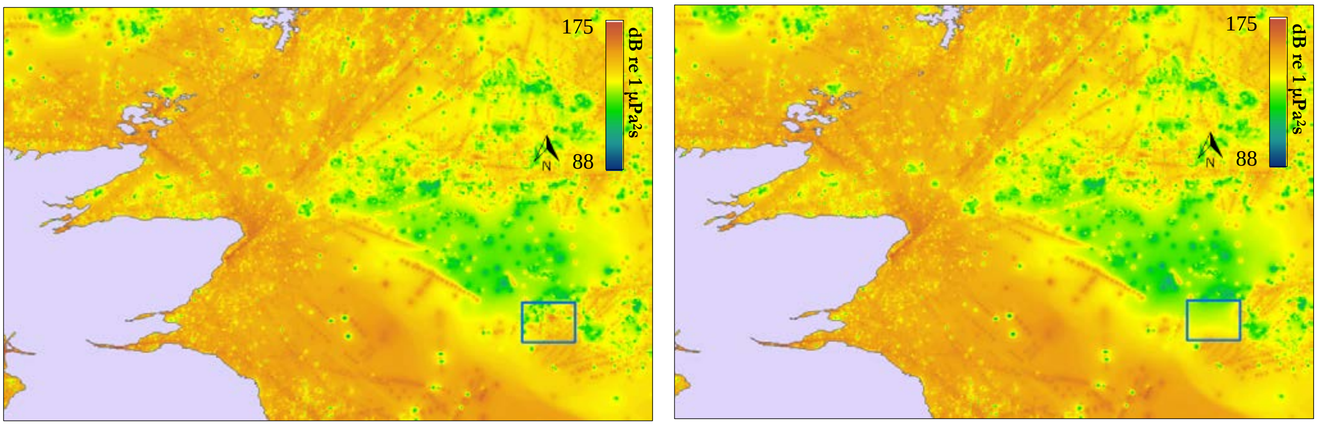


(c) Swallow Sand MCZ with vessel restrictions



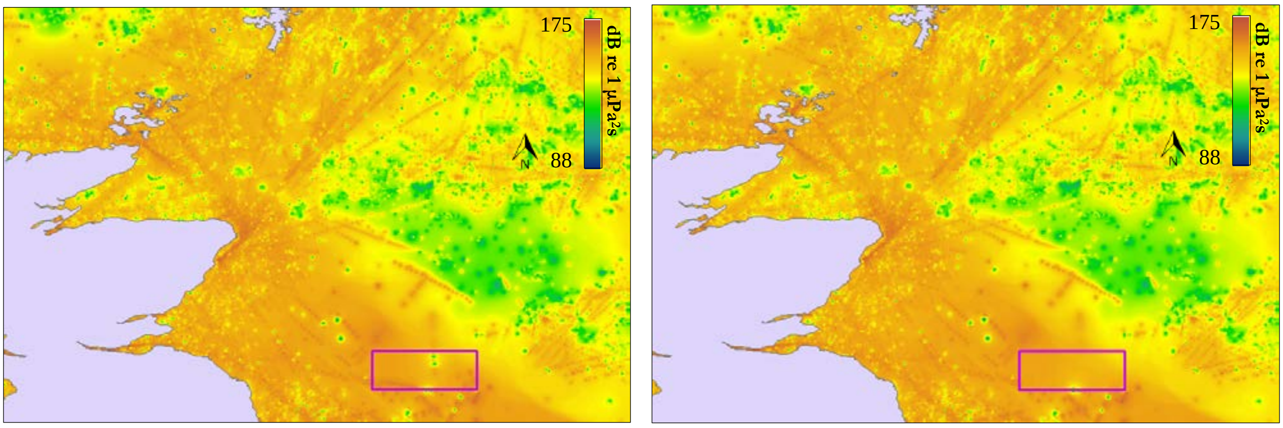
(d) Combined MPA with vessel restrictions

Figure 4.12. Cumulative Sound Exposure Levels (dB re 1 $\mu Pa^2 s$ at 125 Hz) for 2013. **(a)** Model output for 2013 without any geographic restrictions on vessels movements. **(b)** Vessel restrictions apply in Fulmar MCZ. All AIS transmissions were removed from the dataset within the MPA boundary. **(c)** Vessel restrictions apply within the Swallow Sand MCZ. **(d)** Vessel restrictions apply in the combined MPA (encompassing of Fulmar and Swallow Sand). Each frame stretches in latitude from 44 °N to 65 °N and in longitude from 20 °W to 12 °E. The model was rerun on a smaller area as MPA size was too small to observe a difference at the current scale. For higher resolution images see Figure 4.13, Figure 4.14 and Figure 4.15.



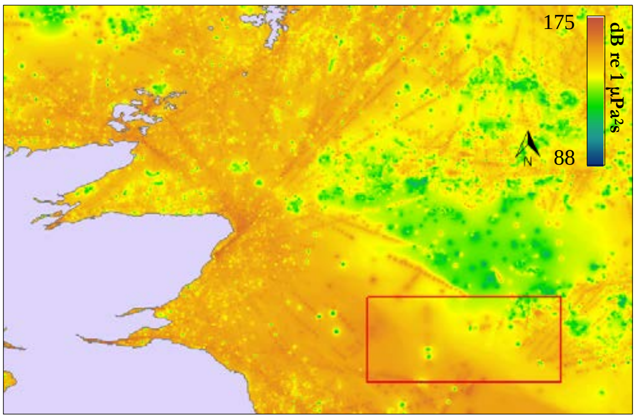
(a) Fulmar MZC without vessel restrictions **(b)** Fulmar MCZ with vessel restrictions

Figure 4.13. Model output of the Fulmar MCZ vessel restriction scenario. **(a)** The SELcum (dB re 1 μPa^{2}s at 125 Hz) from 2013 calculated using real AIS data. **(b)** The predicted SELcum (dB re 1 μPa^{2}s at 125 Hz) if vessels were not allowed inside the MPA boundary by manipulating the 2013 AIS data. Each frame stretches in latitude from 55.05 °N to 60.43 °N and in longitude from 5.50 °W to 4.26 °E.

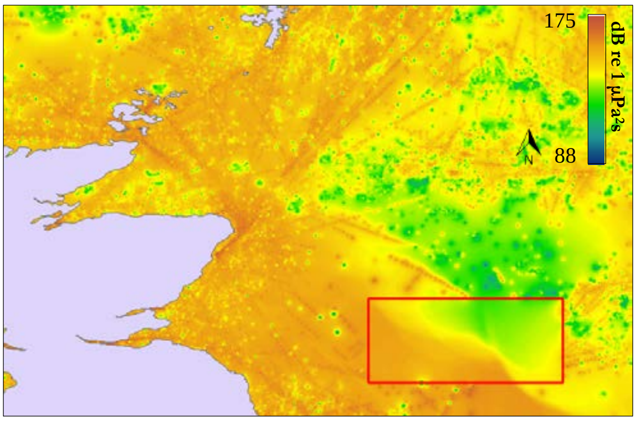


(a) Swallow Sand MZC without vessel restrictions **(b)** Swallow Sand MCZ with vessel restrictions

Figure 4.14. Model output of the Swallow Sand MCZ vessel restriction scenario. **(a)** The SELcum (dB re 1 μPa^{2}s at 125 Hz) from 2013 calculated using real AIS data. **(b)** The predicted SELcum (dB re 1 μPa^{2}s at 125 Hz) if vessels were not allowed inside the MPA boundary by manipulating the 2013 AIS data. Each frame stretches in latitude from 55.05 °N to 60.43 °N and in longitude from 5.50 °W to 4.26 °E.



(a) Combined MPA without vessel restrictions



(b) Combined MPA with vessel restrictions

Figure 4.15. Model output of the Combined MPA vessel restriction scenario. **(a)** The SELcum (dB re 1 μPa^{2}s at 125 Hz) from 2013 calculated using real AIS data. **(b)** The predicted SELcum (dB re 1 μPa^{2}s at 125 Hz) if vessels were not allowed inside the MPA boundary by manipulating the 2013 AIS data. Each frame stretches in latitude from 55.05 °N to 60.43 °N and in longitude from 5.50 °W to 4.26 °E.

4.2.3.4 Discussion

An ecologically coherent network of MPAs will assist in the protection of our marine species and habitats (Department of the Environment Northern Ireland, 2015), and a collection of areas working synergistically together will provide more benefits than an individual area could on its own (Joint Nature Conservation Committee, 2002). Removing vessel activity from Swallow Sand had little impact on the vessel noise exposure levels within the MPA. Noise exposure levels did reduce in the Swallow Sand MCZ (reduction of 7 dB re 1 μPa^{2}s in the loudest grid cell) but not to the same extent as the noise exposure levels in Fulmar (reduction of 26 dB re 1 μPa^{2}s in the loudest grid cell) when vessel activity was removed (Figure 4.12 and Figure 4.13). This difference could have been due to Swallow Sand experiencing heavier surrounding vessel traffic and louder SELcum to begin with compared to Fulmar. MPAs in areas with heavy surrounding traffic may not benefit as much from restricting vessel traffic inside the boundary as those in lower surrounding traffic areas, owing to the propagation of noise from vessel sources outside the area. This hypothesis was tested by combining the two MPAs into one large MPA, the result of which lowered exposure levels in the majority of the area (Figure 4.15 b). Swallow Sand still showed noise levels above 150 dB re 1 μPa^{2}s towards the end of the MPA but noise exposure towards the centre of the combined MPA did reduce to below 140 dB re 1 μPa^{2}s suggesting outside propagation does influence vessel noise exposure in small reserves near shipping lanes.

Vessel traffic, especially cargo vessel activity (Figure 4.2), follows the coastline around the UK, and intersects with many of the designated MPAs. Establishing MPAs away from shipping lanes will allow smaller reserves to have a greater level of protection from noise, than those situated in close proximity to shipping lanes, as they will receive less propagation from nearby vessel traffic. A benefit of modelling using AIS predictive capabilities is that various different scenarios can be examined and the boundaries of MPAs can be altered and manipulated in various directions to find the best placed boundary to protect the required area from noise pollution.

As the boundary of an MPA is small (in comparison to the whole ocean) the regulations set within the MPA will not influence the wider issue of ocean noise pollution. It is likely that noise impacts occurring within an MPA are due mainly to noise from vessels outside the area propagating in. Whether there is benefit to stretching MPA boundaries all the way to the coastline was discussed in relation to the Channel Islands National Marine Sanctuary but no action was taken at the time (Haren, 2007). Predictive modelling using AIS data from within and around the Sanctuary would have provided more detailed information that may have helped in the decision to take action or not. Without testing multiple models, discussions on potential mitigation measures are speculative and may never receive action due to high levels of doubt about the wisdom of incurring the costs of implementing the measure without evidence of a successful outcome. Running multiple scenarios through modelling and addressing and answering questions initially and prior to introducing any mitigation measures provides a sound basis for decision making, and every likelihood of a beneficial result.

Geographic mitigation can reduce noise exposure levels for vulnerable species within the boundary, but such mitigation measures rely heavily on detailed spatial knowledge of species distribution patterns combined with the ability to avoid generating noise in the area (Simmonds *et al.*, 2014). As well as information on vessel source level emissions, and the subsequent noise exposure levels, biological information, such as species distributions, is needed to ensure restrictions are situated in the optimal location.

For species facing severe threats from shipping noise, geographic restrictions should be implemented on vessel activity to the maximum extent possible (Harris, 2017). Geographic mitigation is effective if planned well, but without other mitigation measures outside of the area, the net output of noise from vessels remains largely the same as it was before the restrictions were implemented. However, spatial and temporal restrictions can only protect species with consistent or predictable distribution patterns. MPAs can protect known important habitats from noise, if well designed and sufficiently large, but for wide ranging or unpredictable species it may not be the best form of mitigation.

AIS modelling is able to demonstrate whether vessel noise emissions within MPAs and the surrounding non-controlled areas is propagating at harmful levels into areas designated as protected, and how these noise levels can be reduced by altering shipping within the MPA and surrounding areas. In the case of Fulmar MCZ, reducing shipping noise within the boundary did not significantly decrease vessel noise exposure (owing to small geographic area and propagation from vessels outside the boundary) and, therefore, does not protect species within the MPA from the impacts of vessel noise pollution. This model could be used to adjust the boundaries of the MPA until the species within do have some refuge from vessel noise exposure. However, designating larger protected areas could cause conflict with the shipping industry and may not be viable but at least the potential harm to the environment and marine fish can be highlighted in future discussions.

Time closures are not always possible to implement. Ships travelling to the Baltic Sea have only one entrance through a channel, which if closed at times of migration or spawning, would heavily impact the vessel traffic to the ports in the Baltic. In such cases, reducing the source level of vessels could be a beneficial alternative as vessel noise exposure levels would be reduced without impacting on port business.

Marine protected areas in all their myriad forms are a useful conservation tool, but planners should be aware that failures of MPA planning and management results in wasted resources, scepticism about MPAs, and lost opportunities (Agardy *et al.*, 2011). Marine protected areas (MPAs) are just one of a number of spatial tools that can be used to protect species from chronic ocean noise. Other tools include ship speed restrictions, critical habitat designations for endangered species, or changing of shipping lanes (Zacharias and Gregr, 2005; Dolman, 2007; Hatch and Fristrup, 2009).

4.2.4 Scenario 4: Source Level reduction

4.2.4.1 Introduction

It is recommended that geographic mitigation and the technologies to reduce source emissions are a priority to reduce noise exposure risk to marine life, and, as with other forms of pollution, reduction at the source is the most effective approach to reducing overall impacts (Simmonds *et* *al.*, 2014). Geographic mitigation is effective, but limited spatially and so will not affect the overall net vessel noise exposure levels in the ocean.

Vessels produce noise at low frequencies (100 – 1,000 Hz) and, for the largest commercial vessels, at intensities from 150 dB to over 190 dB (re 1 µPa at 1 m) (Arveson and Vendittis, 2000; Hildebrand, 2009; McKenna *et al.*, 2012). The noise emissions are produced through propulsion, propeller singing and cavitation, hydraulic flow over the hull, turbulence around various external ship elements, and other machinery such as engines, generators, fans, and on-board navigational sonar (Richardson *et al.*, 1995; Erbe, 2002; Southall, 2005; Hildebrand, 2009; National Marine Fisheries Service, 2016). Source-based mitigation could be a very effective form of mitigation for vessel noise impacts on marine species. Reducing the source level would lessen the severity of damage to exposed fish by lowering the intensity of the noise emissions. The likelihood of exposure would also decrease as the noise would not propagate as far from the source. Such mitigation can be achieved through technological advances in propulsion mechanics, fitting non-cavitating propellers, or larger propellers that reduce tip speed and consequently reduce cavitation (Weilgart, 2007b; Southall and Scholik-Schlomer, 2008; McKenna *et al.*, 2017). The IMO (2013) reported concerns about the impacts of vessel noise on marine species and provided information on ways to reduce vessel emissions, with emphasis on the need to reduce propeller cavitation. Eliminating or significantly reducing the amount of cavitation from vessel engines by fitting non-cavitating propellers that just pierce the surface of the water, will correspond to a significant reduction in noise production (Weilgart, 2007b). Such technology is already in use on some vessels. Techniques to minimise engine noise, including electric-powered generators, have already been incorporated in some passenger vessel designs (Jasny *et al.*, 2005). Implementing the noise-reduction technologies will require cooperation from shipping companies, ship builders, and designers, as well as support from port authorities, ship classification, or green certification societies (Simmonds *et al.*, 2014).

This scenario is potentially the most appealing to the shipping industry as it will not increase their journey times or fuel consumption through vessels being rerouted around protected areas. Cavitation and other forms of noise production from normal vessel operations indicates "inefficiencies in engineering" which the shipping industry is keen to improve on (Jasny *et al.*, 2005). There is a high initial cost in implementing noise-reducing technologies, especially if having to remodel older vessels, but the improvements have cost efficiencies long-term, such as reduced need for engine maintenance and increased fuel efficiency (Haren, 2007; Weilgart, 2007b). Many noise-reducing implementations are cost-affordable, and provide additional savings or operational benefits through the improved efficiencies (Southall and Scholik-Schlomer, 2008).

The aim of this scenario is to demonstrate how AIS modelling can be used to investigate noise-reducing technologies, and whether such technologies are worth the investment in terms of vessel noise impact mitigation for marine species.

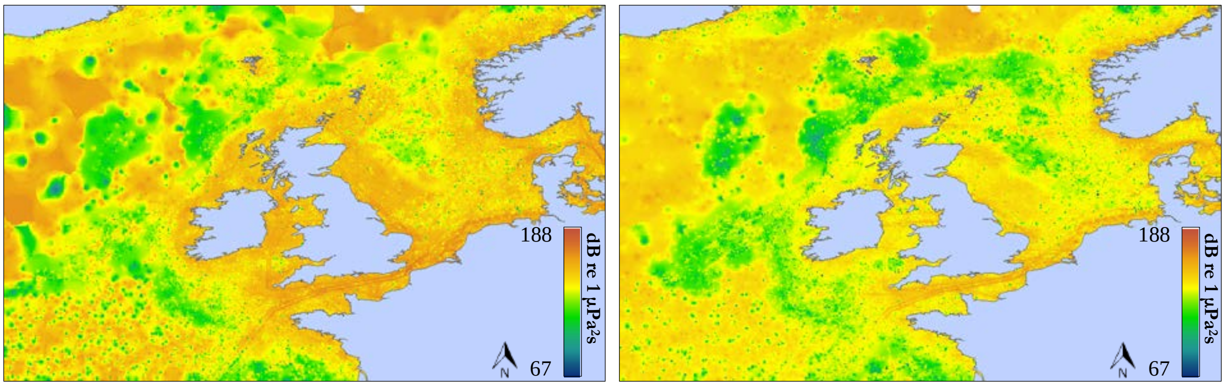
4.2.4.2 Method

AIS transmissions from satellite and terrestrial AIS data (provided by Orbcomm) covering all vessel types within latitudes 40 ºN & 65 ºN and longitudes 20 ºW & 12 ºE during 2013 were used to produce a map of UK waters following the method in Chapter 3. To investigate the potential of source-based mitigation, the SSLM equation in the model was manipulated to reduce all vessel Source Levels by 20 dB re 1 $\mu Pa^2 \cdot m^2$. The model was then run with these reduced Source Level values to produce cumulative Sound Exposure Levels for the study area to investigate how noise exposure would differ if all vessels were built with noise-reducing technologies.

4.2.4.3 Results

Vessel noise exposure levels at 125 Hz during February 2013 reached peak levels of 186 dB re 1 μPa^2s (Figure 4.16). The simulated noise-reduced vessel, however, only reached peak levels of 173 dB re 1 μPa^2s in only a few individual grid cells. No hotspots (consisting of 2 of more adjacent 1 km grid cells) of noise greater than 160 dB re 1 μPa^2s can be seen on the model output (Figure 4.16 b). The green areas on the map output represent noise exposure levels of up to approximately 130 dB re 1 μPa^2s, with yellow areas ranging from 130 dB to 145 dB re 1 μPa^2s and orange to red areas represent noise exposure levels of 145 dB re 1 μPa^2s or louder.

Coastal areas and shipping lanes seem much quieter with noise-reduced vessels in use, particularly around the UK coastlines. The use of noise-reducing technologies to reduce Source Level emissions of vessels does influence overall cumulative noise exposure. This shows a promising mitigation measure that could be used to help combat the harmful impacts noise exposure has on fish species.



(a) February SELcum from standard vessels

(b) February SELcum from noise-reduced vessels

Figure 4.16. Model output of the Source Level reduction scenario for February, with a 20 dB re 1 $\mu Pa^2 \cdot m^2$ reduction. **(a)** The SELcum (dB re 1 $\mu Pa^2 s$ at 125 Hz) from 2013 calculated using real AIS data without manipulation of the vessels' Source Level. **(b)** The predicted SELcum (dB re 1 $\mu Pa^2 s$ at 125 Hz) if vessels were built with noise-reducing technology by manipulating SSLM equation in the model. Each frame stretches in longitude from 20 ºW to 12 ºE and in latitude from 44 ºN to 65 ºN. February was simulated as endangered species such as the European eel migrate during this month and likely face noise barriers in the English Channel and Baltic Sea.

4.2.4.4 Discussion

Jasny *et al.* (2005) stated that source-based mitigation must be a vital component of any long-term policy to reduce shipping noise. Systematic improvements of source level noise-reduction technologies could have widespread positive influences on vessel noise exposure throughout the world's oceans and would provide the most productive mitigation measure (Harris, 2017). These are strong views stated by researchers in review articles, with no data to evidence the statements. This AIS model has shown that predicted noise exposure would drastically reduce if source-based mitigation was used, providing much larger reductions in noise exposure levels than the geographic mitigation investigated in the previous scenarios. This suggests that decision-makers should strongly consider incorporating noise-reducing technologies in future vessel builds.

This scenario provides evidence that AIS modelling can be used to investigate the influence noise-reducing technologies can have on vessel noise exposure in the marine environment. For this scenario, a noise reduction of 20 dB re 1 $\mu Pa^2 \cdot m^2$ was simulated for every vessel. It is likely that noise-reducing technology will be focused on freight vessels such as cargo vessels and tankers, and will not be fitted to every type of vessel. The model could be run for just the freight vessels, and this again demonstrates the flexibility of AIS modelling. A huge variety of options can be simulated by manipulating real historical AIS data to predict future trends.

The results of this AIS modelled scenario, indicate that if noise-reducing technology were implemented in vessels on an international scale it would cause a significant reduction in vessel noise exposure levels, and therefore, ambient noise levels throughout the world's oceans.

4.2.5 Scenario 5: Reduction in vessel speed

4.2.5.1 Introduction

Noise-reducing technological advances will be instrumental in ensuring a quieter ocean for marine species, making source-based mitigation a vital component of any long-term policy to reduce shipping noise (Jasny *et al.*, 2005). Simmonds *et al.* (2014) recommended that the continuing development and use of noise-reducing technologies should be prioritised and set in policy. However, even though the shipping industry is largely agreeable with noise-reducing technological advancements (due to them fixing inefficiencies and decreasing long term costs e.g. more fuel efficiency (Haren, 2007)), or less requirements for repeated engine maintenance the initial installation of such devices will be costly (Weilgart, 2007b; McKenna *et al.*, 2017). Additionally, it may be even less economically viable to alter existing ships, meaning that noise-reducing technologies could only be installed on new vessels (Jasny *et al.*, 2005; McKenna *et al.*, 2017).

A more immediate method of reducing vessel source level emissions is to reduce the speed of vessels. An increase in average received noise levels has been observed when large ships abided by a 20 knot speed regulation compared to a 10 knot speed regulation (McKenna *et al.*, 2017); average received SPLs were 10-15 dB higher during the 20 knot speed regulation period, compared to the 10 knot speed regulation. Other studies have also shown evidence for lower recorded noise levels when travelling at reduced speeds at low frequencies (Kipple and Gabriele, 2004; McKenna *et al.*, 2013).

Setting speed limit regulations could be a useful mitigation method for reducing noise impacts on fish. However, as with other large-scale mitigation measures involving changes in policy, there is an arduous process to complete with multiple stakeholders. In order to set up the Channel Islands National Marine Sanctuary, mentioned in Scenario 3, the authors of the study had to consult with agencies such as the Department of Defence and the Coast Guard as they were liable to have concerns over regulating international vessel traffic (Haren, 2007). To avoid legal processes that can limit or slow down research, modelling can effectively investigate whether speed regulations would indeed work. Predicting outcomes of speed reductions using real AIS data provides evidence that can be used to argue the case for speed regulations to be put in place in certain areas.

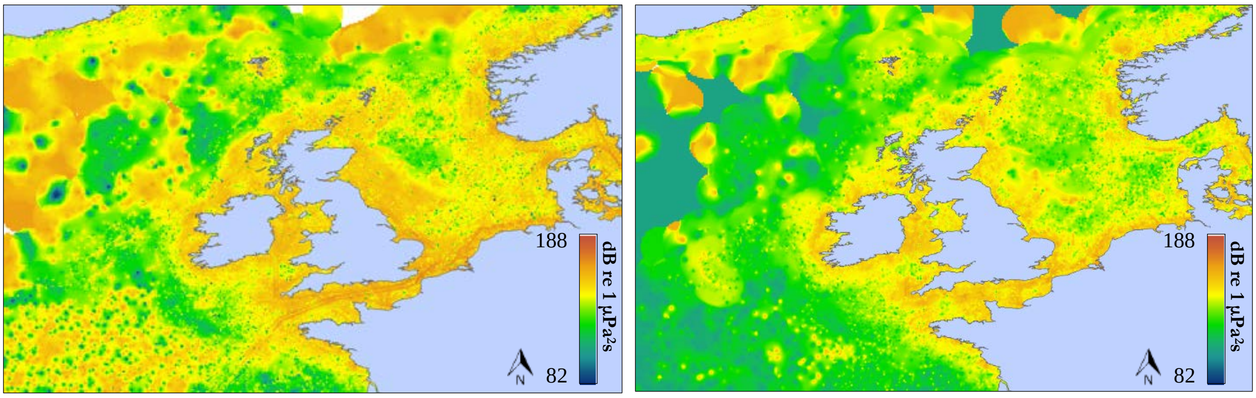
This scenario aims to show the benefit of using AIS modelling as a method to investigate whether speed regulations would make an effective mitigation method, and provide predictions of noise emissions and levels of exposure for different speed limits.

4.2.5.2 Method

An ocean noise map was built using the AIS data modelling method created in Chapter 3. The data used was from satellite and terrestrial AIS data (provided by Orbcomm) within latitudes 40 °N & 65 °N and longitudes 20 °W & 12 °E for 2013. All vessel types were included in this scenario. The vessel database used by the AIS model to calculate the current speed of the vessel at the time of transmissions (*see* Figure 3.3) was manipulated so that all vessel's current speed was reduced to 10 knots. Ten knots was used as the speed restriction in this scenario as at speeds greater than 10 knots propagation distance can increase by a factor of 10 (Kipple, 2002). Additionally, 10 knots has been used as a speed limit in national park regulations (e.g. National Park Service, 2003). The AIS model was used following the method in Chapter 3 to produce 1 km grid cell cumulative Sound Exposure Levels with a visual map output created in ArcMap 10.4.1. The model was run four times for this scenario, twice with the original 2013 AIS data (for February and for the whole year), and again with the simulated speed data (for February and for the whole year).

4.2.5.3 Results

There is a dramatic difference in cumulative Sound Exposure Levels at 125 Hz between vessels operating at normal speeds and those same vessels limited to 10 knots (Figure 4.15). Cumulative Sound Exposure Levels reached as high as 188 dB re 1 μPa^2s when vessels were travelling at normal speed (Figure 4.17 a). Reducing the speed of all ships in the study area to 10 knots had the effect of keeping noise exposure levels for February below 168 dB re 1 μPa^2s (Figure 4.17 b), and for the whole of 2013 below 176 dB re 1 μPa^2s (Figure 4.18). A peak level of 188 dB re 1 μPa^2s was reported from the model, but over too small an area to be visible on the map output (Figure 4.16 a). The noisiest areas reported by the model, the English Channel and the Baltic Sea, saw large reductions in noise levels. Noise exposure levels for the majority of the area reduced from greater than 170 dB re 1 μPa^2s to no more than 140 dB re 1 μPa^2s. The green areas on the map output represent noise exposure levels of up to approximately 140 dB re 1 μPa^{2}s, with yellow areas ranging from 140 dB to 150 dB re 1 μPa^{2}s and orange to red areas represent noise exposure levels of 160 dB re 1 μPa^{2}s or louder.



(a) February without vessel speed restrictions **(b)** February with vessel speed restrictions

Figure 4.17. Model output of the vessel speed restriction scenario for February, with a speed limit of 10 knots. **(a)** The SELcum (dB re 1 μPa^{2}s at 125 Hz) from 2013 calculated using real AIS data without manipulation of the vessels' current speed. **(b)** The predicted SELcum (dB re 1 μPa^{2}s at 125 Hz) if vessels were restricted to moving at a maximum of 10 knots by manipulating vessels' current speed parameter in the 2013 AIS data. Each frame stretches in longitude from 20 ºW to 12 ºE and in latitude from 44 ºN to 65 ºN.

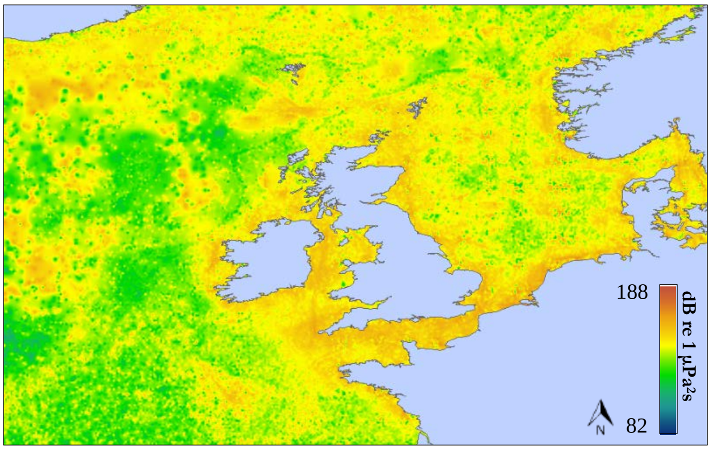


Figure 4.18. Model output of the vessel speed restriction scenario for all months for 2013 combined, with a speed limit of 10 knots. Shows predicted SELcum (dB re 1 μPa^{2}s at 125 Hz) if vessels were restricted to moving at a maximum of 10 knots by manipulating vessels' current speed parameter in the 2013 AIS data. Each frame stretches in longitude from 20 ºW to 12 ºE and in latitude from 44 ºN to 65 ºN.

4.2.5.4 Discussion

Although it is unrealistic for all ships to travel at such low speeds throughout their entire journey, this scenario demonstrates the influence speed restrictions can have on vessel noise exposure. Cumulative Sound Exposure Levels decreased by up to 40 dB in some areas, resulting in levels as low as 106 dB re 1 μPa^2s.

This scenario suggests that, in terms of noise pollution, vessel speed restrictions would be a beneficial mitigation method. Speed restrictions could be regulated in important locations such as spawning grounds, feeding grounds and Marine Protected Areas. This would prevent known impacts of vessels such as avoidance or diminished anti-predator responses (Sarà *et al.*, 2007; Simpson *et al.*, 2015).

Speed restrictions would have instantaneous benefits, unlike other mitigation measures that may take time to implement (for example, MPA designation or building vessels with noise reducing technologies). However, a concern with implementing speed restrictions to areas of importance around the UK is that the majority of ships passing through the areas are foreign-flagged, meaning they may not be subject to UK rules and regulations owing to the vessels being outside of the UK jurisdiction (Haren, 2007). Of the 458,306 vessels transmitting data used in this scenario, only 10,073 vessels are registered with a UK flag (Table 3.4). If only 2 % of ships restrict their speed then there may be no overall influence on vessel noise exposure levels. Unless states have the ability to apply regulations to foreign ships, international cooperation is needed. Implementing speed restrictions would involve new policy being created which would take both time and financial resources, especially if international agreements need to be made. Predictive AIS modelling has provided new information that can be used as evidence to support the case for implementing speed restricting mitigation measures, without using extensive resources.

This scenario has shown that it is possible to use historical AIS data to predict future trends and results of mitigation measures such as speed restrictions. Although only one speed restriction model was run here as a comparison to no restrictions, it is possible to run the model on any, or several different, speed limits. Running a number of models investigating different speed limits would allow an optimum speed restriction to be found; a speed limit that reduces noise to below harmful levels, without being unnecessarily damaging to journey times of vessels.

4.3 Discussion

Owing to overloaded road networks and clean air acts dictating carbon emission targets for road transport intermodal freight transport has grown significantly and ocean vessels are destined to play a significant role in the future (Zhang, 2006; Macharis *et al.*, 2011). Vessel operation statistics show steady growth of ocean vessel traffic over the past few decades (U.S. Department of Transportation Maritime Administration, *n.d.*), and with this vessel noise exposure in the marine environment is likely to increase (Figure 4.5). Predicting the impacts of such trends on marine fish species is needed in order to plan efficient mitigation strategies. Model based approaches represent a powerful way of evaluating ocean noise levels at a lower cost than large scale field research, and make population, community and ecosystem level research more feasible (Dekeling *et al.*, 2016; Rossi *et al.*, 2016). Modelling of different future scenarios allows better understanding of noise patterns and trends and the implications changes in noise levels can have (Hatch *et al.*, 2016; Haver *et al.*, 2017). The various scenarios investigated in this chapter were designed to clearly demonstrate the potential of AIS modelling in this regard.

Cargo vessels are widespread contributors to ocean noise levels, with frequencies ranging from 100 to 1,000 Hz and source levels rising over 190 dB re 1 $\mu Pa^2 \cdot m^2$ (Arveson and Vendittis, 2000; Hildebrand, 2009; McKenna *et al.*, 2012; McDonald *et al.*, 2014). This means that they produce noise levels that cause adverse reactions in many fish species (Sarà *et al.*, 2007; Popper and Hastings, 2009b; Simpson *et al.*, 2015; Kastelein *et al.*, 2017).

It was thought that reducing the levels of cargo vessel activity in the oceans would greatly reduce the noise exposure levels in the marine environment and so would be worth considering as a future mitigation measure. However, a 30 % decrease in cargo activity across the study area resulted in a less than 2 dB re 1 μPa^2s reduction in overall noise exposure, which suggests the costs of implementing such an effort intensive mitigation measure would outweigh the benefits it would provide for fish species. Instead of a decrease in worldwide cargo vessels, which has little impact on noise exposure risks faced by fish species, geographic restrictions could eliminate all vessel activity completely from an area of importance to species, such as spawning grounds. Removing all vessel activity from an area can reduce noise levels within the boundary by up to 20 dB re 1 μPa^2s. Localised geographic mitigation measures are, therefore, more beneficial than reducing just cargo vessels over a greater area. Unfortunately, without other mitigation measures outside of the area, (or alternatively creating much larger geographic areas), the net output of noise from vessels remains largely the same as it was before the geographic mitigation measure was implemented. To create widespread reduction in vessel noise exposure, all vessels must somehow be influenced by the mitigation measure. Source-based mitigation may have potential to reduce net ocean noise exposure for widely distributed populations by reducing the source level of individual vessels. Fitting vessels with noise-reducing technologies that reduce Source Level emissions by 20 dB re 1 $\mu Pa^2 \cdot m^2$ can reduce monthly cumulative noise exposure levels by over 10 dB re 1 $\mu Pa^2 s$. Owing to improved efficiencies in maintenance and fuel consumption from noise-reducing technologies, more and more vessels may be built in ways that reduce their Source Level emissions. This implies that noise exposure levels from vessels in the long-term may reduce, or at the very least remain static as vessel activity increases and more vessels take to the water. Fitting new technologies into older vessels would be an expensive endeavour and would not be widely accepted by the shipping industry so ways to reduce Source Level emissions without remodelling older vessels was, therefore, important to investigate. The Source Level of a vessel can be influenced by the speed the vessel is travelling (the speed of the propeller and resulting cavitation, noise generated from the engine and the friction on the hull). The Source Level equation used to calculate likely vessel noise emissions incorporate speed scaling to account for the speed of the vessel. As a result, reducing the speed of vessels can readily be modelled to identify trends and to see the likely effect speed reduction will have on noise exposure levels. Implementing speed restrictions of 10 knots can reduce cumulative Sound Exposure Levels by up to 40 dB re 1 $\mu Pa^2 s$ in some areas. This means that vessels travelling through known fish habitats or areas of importance could reduce their speed, and therefore their Source Level noise emissions, and reduce the risk of harm to the species inhabiting the area. Speed restrictions could be quick to implement if vessels voluntarily adhered to the limits, which has been done in noise reduction trials in the Port of Vancouver's ECHO programme (Port of Vancouver, 2017).

Simmonds *et al.* (2014) recommended the implementation of noise-reducing technologies and the spatial and temporal exclusion of noise as a priority to minimise risk of exposure for marine species. Comparing models of multiple mitigation strategies can justify which of the available alternative measures should be viewed as a priority, which will in the long term provide most benefit for least cost, and identifies those whose implementation would have little significant effect. From the scenarios run in this chapter, although both noise-reducing technologies and geographic mitigation displayed a positive influence on cumulative vessel noise exposure levels, the highest priority should be to reduce vessel speed in biologically important areas. This action would prove the most beneficial as it would be relatively quick and easy to implement in comparison to other measures, although it would still require agreement from all interested parties.

AIS modelling is a powerful tool for noise pollution mitigation management. It can not only look at historical trends in ocean vessel noise, but can predict future trends and outcomes of mitigation actions. This chapter suggests that the most beneficial mitigation action, out of those considered, is to set vessel speed restrictions in areas of importance to fish species. This measure could reduce ocean noise exposure from vessels by up to 40 dB re 1 μPa^2s in some areas. Long-term, noise reducing technologies would help combat issues of noise pollution impacts on fish, but the cost of altering vessels already in service is high, and so would not be viable unless government incentives were made available. Noise-reduced technologies can help to improve fuel efficiency and reduce maintenance costs, so as new ships are commissioned and built the shipping industry will possibly incorporate such measures voluntarily.

4.4 Conclusion

The scenarios identified here are all potential mitigation measures that could be used to reduce the impacts of vessel noise pollution on fish. Knowing which method to use, when, and where, is difficult, especially when the needs of the many different marine species and the economy have to be taken into consideration. Reducing the source level emissions of vessels could influence all vulnerable species positively, whereas geographic or temporal mitigation would only aid species with predictable distributions in the locale of the mitigation measure. This work uses AIS modelling to predict potential influences each mitigation measure scenario could have on vessel noise exposure levels. The results provide useful information that can be used by decision makers to help identify and implement the most appropriate mitigation action for the challenge under consideration. When multiple mitigation measures are deemed necessary modelling can suggest the order of implementation for maximum cumulative effect.

This chapter has shown that it is possible to use historical AIS data to predict future trends and likely influences of various mitigation measures. The outputs of AIS-based models could contribute to a greater understanding of the likely impacts of noise in the future by predicting possible effects of future changes, and allow predictions about the efficacy of alternative mitigation actions (Dekeling *et al.*, 2016). The models indicate that AIS-based noise mapping could be successfully applied to target vessel noise mitigation efforts across wide ranging marine habitats. The scenarios exemplified in this work gives greater credence to the usefulness of AIS models as a method for mitigation, by using AIS data to predict vessel noise emissions on a large geographic scale.

5. Chapter Five: Combining prioritisation, historical and predictive modelling as a method for mitigation

Behavioural changes in response to noise pollution have the potential to influence species migrations. Avoidance of noisy areas, disorientation, and reduced predator awareness have all been observed as a response to vessel noise (Ona and Godø, 1990; Amoser *et al.*, 2004; Sarà *et al.*, 2007; Simpson *et al.*, 2015; Kastelein *et al.*, 2017). Alterations in behaviour patterns could lower the number of individuals reaching spawning areas, reducing progeny in the short term, and influencing the abundance of future generations. The effects of noise on migration is of particular interest for long distance migrants, such as salmon and eel. Combinations of modelling and field research were used to map the distribution patterns of these two noise-sensitive species to identify any potentially harmful noise hotspots. Such information can be helpful for mitigation decision-makers, when deciding where, for instance, noise emission restrictions are needed, and best placed for maximum effect. The method presented combines the two types of mitigation for noise pollution impacts by overlaying models of estimated source levels with biological information collected from the two migrating species, allowing identification and prioritisation of mitigation measures to aid species on long distance migrations. European eel and Atlantic salmon were chosen as representative species in case studies to investigate whether they are exposed to vessel noise during migration, identify any noise hotspots that may have potential to harm them, and pinpoint areas that may act as barriers to migration. AIS and species migration data are used to produce exposure risk maps to highlight areas of species co-occurrence with vessel traffic, and to assess acoustic exposure along the migration route and surrounding ocean, during particular periods. This is important as the migrating species may be subject to a greater degree of harm if exposed to higher noise levels at certain times or places on their route. Exposure risk maps allow more informed recommendations to be made, such as vessel speed restrictions for certain months of the year in specific areas.

5.1 Introduction

The omnipresence of anthropogenic noise pollution in marine ecosystems is an issue for many fish species (Andrew *et al.*, 2002; World Health Organization, 2011). Evidence of negative noise impacts on fish is increasing, with changes in foraging, communication, antipredator behaviour, nesting, territoriality, and physiology already recorded (e.g. Codarin *et al.*, 2009; Bruintjes and Radford, 2013; Voellmy *et al.*, 2014; Luczkovich *et al.*, 2016; Nedelec *et al.*, 2017). Although vessel noise is not as intense as other anthropogenic noise sources it is constant over time, can be considered both continuous (as background noise) and intermittent (when passing in close proximity), and is widespread throughout the world's oceans (Celi *et al.*, 2016; Radford *et al.,* 2016b). Vessel noise could pose a serious hazard not only to individual animals, but also to entire populations (Weilgart, 2007b; Clark *et al.*, 2009; Slabbekoorn *et al.*, 2010). Although noise impact research and knowledge has increased greatly over recent decades, the overall picture remains incomplete, especially in relation to population level consequences of noise exposure (McGregor *et al.*, 2013). There is concern that the persistence of species, or populations, could be negatively affected. Behavioural impacts of noise exposure may be the most detrimental for populations owing to the spatial scale over which effects can occur (Normandeau Associates Inc., 2012; Simpson *et al.*, 2015). Individual behaviours that are required for migration have been seen to alter in noisy conditions, suggesting migration may be negatively impacted by anthropogenic noise. Previously mentioned studies (e.g. Sarà *et al.*, 2007; Simpson *et al.*, 2015; Luczkovich *et al.*, 2016; Herbert-read *et al.*, 2017) have considered how anthropogenic noise impacts behaviours used during migration, such as communication, movement patterns and foraging.

During migration, fish use their lateral line as their main sensory ability for rheotaxis, and to perceive neighbour movement and form a school. Noise can impact the ability of individuals to coordinate their movements by masking information normally detected through the lateral line about neighbours' positions, or it can impair information processing though stress or distraction (Montgomery *et al.*, 1997; Voigt *et al.*, 2000; Herbert-read *et al.*, 2017). Noise has been seen to impact schooling in fish by altering social interactions, which can cause dispersion of a school resulting in increased predation risk (Hamilton, 1971; Herbert-read *et al.*, 2017; Ioannou, 2017).

Diminished antipredator behaviour success has been observed in at least 5 studies in recent years (e.g. Everley, 2013; Voellmy *et al.*, 2014; Nedelec *et al.*, 2015; Simpson *et al.*, 2015; Bruintjes *et al.*, 2016). This may mean that species migrating through noisy areas may have lower chances of survival owing to higher predation risk. Swimming patterns, such as lateralisation, which are important for successful predator avoidance as well as other cognitive tasks, have also been negatively impacted by noise (e.g. Simpson *et al.*, 2015). Additionally, increased stress and distraction can indirectly affect behaviours important for migration (Chan *et al.*, 2010).

At the time of migration, species are often in a more vulnerable stage of their life cycle (e.g. larvae). Kastelein *et al.* (2017) observed smaller fish reacting to ~ 10 dB quieter noise than larger fish. Sea bass of 68 days-post-hatch (dph) were observed to be more sensitive to noise compared to juveniles of 115 dph providing evidence for smaller fish having higher sensitivity (Debusschere *et al.*, 2014). Studies have shown larvae to be vulnerable to noise through reduced growth, diminished settlement ability and changes in swimming behaviours (e.g. Davidson *et al.*, 2009; Holles *et al.*, 2013; Nedelec *et al.*, 2015; Simpson *et al.*, 2016). Any detrimental effect on survival at an early life stage, even if subtle, may have consequences for population dynamics as high mortality of the early stages can have a large influence on population fluctuations (Gagliano *et al.*, 2007). Migrating species tend to use currents (Figure 5.1) to aid movement over long distances to save energy and optimise growth (Ware, 1975; LaBar *et al.*, 1978; Weihs, 1987 (as cited in Dadswell *et al.*, 2010); Holm *et al.*, 2004; European Commission, 2007). If currents cross shipping lanes with harmful noise exposure levels then the species may suffer injury whilst travelling through the area, or exert extra energy to divert around the noise, or may even choose to avoid the noise barrier and abandon the migratory journey. Larvae, especially, use passive transport mechanisms and are limited by swimming ability (Friedland *et al.*, 1999; Holm, 2000). Whilst using passive transport, which can last a number of months, fish are exposed to the prevailing climatic and environmental conditions (Friedland *et al.*, 1999).

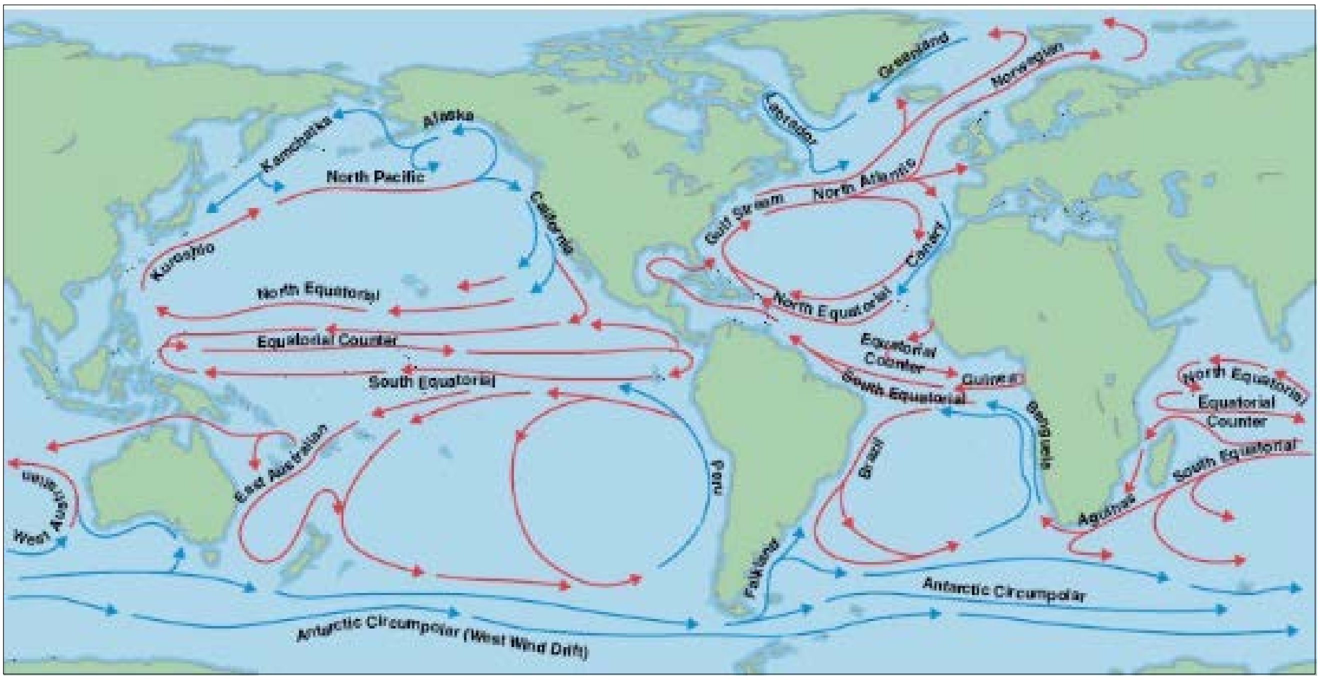


Figure 5.1. Global ocean currents. The arrows indicate direction of currents, with warm-water currents represented by red arrows, and cold-water currents represented by blue. It is suggested that the European eel uses the Gulf Stream and the North Atlantic currents to aid their migration. Image taken from (Cooperative Institute for Meteorological Satellite Studies, *n.d.*).

Manmade structures are known to fragment habitats and prevent access to rivers and spawning grounds (e.g. dams) (Atlantic Salmon and Sea-run Fish Restoration in Maine, 2015). This can result in populations being exposed to higher rates of extinction from demographic, environmental, and genetic stochasticity (Reed, 2004; Chevin and Lande, 2010; Chust *et al.*, 2013). Shipping lanes have the potential to act as barriers, causing fragmentation of the oceans. Fragmentation of marine environments could have a negative impact on species migration, limiting access to feeding or spawning grounds. Noise barriers, coupled with noise impacts on fish could have significant population level consequences. The migration patterns of some species remain largely unknown (European Commission, 2007c). Much more information, analysis, and research are needed to achieve a clearer picture of marine survival of migrating species, and whether mitigation actions put in place have the potential to increase survival (Atlantic Salmon and Sea-run Fish Restoration in Maine, 2015).

Unfortunately, large scale population studies are limited, and it is often difficult to create meaningful predictions about population-level consequences (National Research Council, 2005; Morley *et al.*, 2014), because fish may be able to move away from the source, disturbances may be sporadic and compensation may prevent long-term impacts (National Research Council, 2005; Bejder *et al.*, 2006; Normandeau Associates Inc., 2012). It is also difficult to generalise laboratory studies on noise impacts to large scale migration of wild fish, as captive specimens used in laboratory studies reared in hatchery systems can become habituated to noise (Harding *et al.*, 2016). Owing to the caveats of long term, large scale studies of noise impacts on populations, modelling can provide a beneficial tool to predict potential impacts and indicate where mitigation may be needed.

Geographical and seasonal restrictions on noise emissions is a useful method of noise impact mitigation as it can protect critical life cycle events, such as feeding and spawning, and migration to the habitats where these events take place (McGregor *et al.*, 2013). To predict when and where restrictions are needed, species distribution and likely noise exposure levels need to be known.

Noise trends can be analysed using map-based tools; mapping vessel density, source level concentration and exposure levels over time or geographic location can identify trends. Automatic Identification System (AIS) models, as shown in Chapter 3, can produce large scale maps indicating vessel noise exposure levels and providing details on areas of high noise risk. Species distribution maps can then be used to identify areas or locations of high risk for the individual species. Such methods can be used to study the potential risks associated with vessel noise on migrating species, allowing conclusions to be drawn and predictions to be made that can help guide mitigation measures. This is important for species that have migratory life stages that are likely to encounter noise sources during migrations, as adverse reactions to noise could impact their homing accuracy (Sara *et al.*, 2007).

Under the Marine Strategy Framework Directive (MSFD) Member States are required to develop strategies to achieve good environmental status of marine waters. To develop a UK strategy to ensure that migration routes are not impacted by anthropogenic noise sources and/or that conservation efforts are focused where they are most needed, large-scale mapping and predictive tools can be very useful. Conclusions from such models can identify impacts before they occur, and guide effective mitigation for the most vulnerable populations (Williams *et al.*, 2015b). Removing barriers to improve passage significantly decreases fragmentation by increasing access to essential habitats, which in turn restores ecological complexity, protects against environmental stochasticity and helps maintain genetic diversity (Atlantic Salmon and Sea-run Fish Restoration in Maine, 2015).

The aim of this chapter is to combine source level modelling with biological information of migrating species to create a novel method to aid mitigation decision-making. Two migrating species with differing life histories and opposite migration patterns, European eel and Atlantic salmon, were used as case studies. Distribution maps were analysed against AIS-based vessel noise emission maps to identify exposure risk from vessel traffic, and recommendations made for mitigation focus to help marine-stage survival.

5.2 Case study species

5.2.1 European eel

The European eel stock and glass eel recruitment has declined significantly (by around 95 %) since the 1980s (Moriarty and Dekker, 1997; Kettle *et al.*, 2011), and the International Council for the Exploration of the Sea (ICES) advised in 2006 that the stock was outside safe biological limits and that the current fisheries were no longer sustainable. The European eel is considered critically endangered on the International Union for the Conservation of Nature and Natural Resources (IUCN) Red List of species and has been listed in the Appendix II of the Convention on Trade in Endangered Species of Wild Fauna and Flora (The Convention on International Trade in Endangered species of wild Fauna and Flora, 2007). Even though already listed as critical, eel numbers are said to be much lower than recorded due to an ambiguity in catch reporting; catch data for all life stages of eels declared to the UK Environment Agency are unreliable and underestimated (Knights *et al.*, 2001). HM Customs and Excise net export data suggested that the declared glass eel catch was under-reported by a factor of 3.4 to 15 times, and that the true annual catch of glass eel is in the region of 10 tonnes (ICES 2006). Similarly, declared catches of yellow and silver eels in the Severn in 2005 were 1125 kg and 120 kg respectively, from 121 licensed instruments, but ICES suggested that official catch returns are a factor of 2.4 to 7.2 times higher than those reported (2002– 2004 data; ICES 2006). The recovery time of the European eel is expected to be very slow; around 80 years, even in the case of complete closure of fisheries (Åström and Dekker, 2007).

Many factors are suspected of being involved in the decline of the European eel: overfishing, climate change, limited access to upper reaches of the watershed owing to dams and other obstructions to migration, entrapment of downstream migrating silver eels in turbines of hydroelectric power plants, ocean current variations in the Sargasso Sea, pollution and parasites e.g. the nematode (*Anguillicoloides crassus*) found in freshwater (Knights, 2003; Wirth and Bernatchez, 2003; International Council for the Exploration of the Sea, 2006; Baltazar-Soares *et al.*, 2014). Anthropogenic noise impacts could also feature on this list.

European eels are known to detect sound frequencies below 300 Hz (Jerko *et al.*, 1989), which overlaps with the dominant frequencies of vessel emissions. European eel have responded to noise through both physiological and behavioural alterations (e.g. Simpson *et al.*, 2015; Bruintjes *et al.*, 2016; Purser *et al.*, 2016). Elevated stress through increased ventilation rate and altered metabolic rates, diminished antipredator behaviour due to distraction, reduced attention, impaired spatial performance, and changes in lateralisation have all been observed in European eel in response to vessel noise playback. Simpson *et al.* (2015) suggested that such impacts could "compromise life-or-death responses". Noise encountered throughout the long migration has the potential to harm the eel or alter behaviours resulting in lower migration success.

The migratory journey remains one of the big mysteries in the life history of the European eel. Many researchers have tried to identify the route taken by the eel on their 6,000 km migration. The final destination is thought to be a spawning ground in the Sargasso Sea (Schmidt, 1923), but the exact route taken to arrive at the site is still under debate. Throughout their journey the eels venture through rivers, estuaries, coastal zones, and out into deep water, heading towards the Azores before turning to cross the Atlantic. Along the route the eels encounter countless threats and pollutants that could impact on their journey success. Eels migrate seaward during their silver eel life stage which they enter after undergoing a metamorphosis from the yellow eel stage resulting in colour change, reduced size of the gut and enlargement of the eyes (Aida *et al.*, 2003). During the migration, they do not consume food and have to rely purely on fat reserves for energy (Palstra, 2006). This means that any hindrance or prolonging of their journey, or excess expenditure of energy, could prevent the eels from arriving at the spawning ground, or cause them to be in too poor a condition to reproduce successfully once there. Body condition has a significant effect on behavioural responses to vessel noise in European eels; when in poor body condition eel were less likely to startle when exposed to additional noise (Purser *et al.*, 2016), meaning predator avoidance could be less successful during the silver eel life stage. The noise levels the eels encounter on their journey may be significant enough to impact the eel physiologically or/and behaviourally, and thus affect their reproductive success.

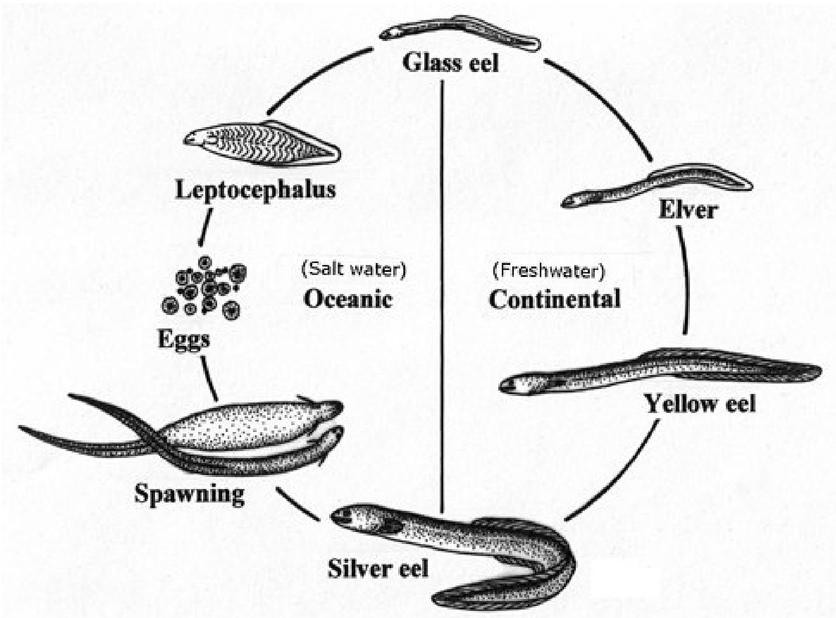


Figure 5.2. Life cycle of the European eel (Bevacqua *et al.*, 2009).

The European eel is a panmictic species (all individuals can potentially mate) that travels over 6,000 km to create a single, randomly mating population in the Sargasso Sea. From the Sargasso Sea, larvae (*leptocephali*) are transported (most likely via currents) to the coastal and freshwater foraging areas in Europe (Tesch, 2003). After metamorphosing into glass eels, they recruit to continental waters, which they inhabit during their growing phase (yellow eel). When they reach sexual maturation size, adult eels undergo a second metamorphosis into silver eels and migrate back to the spawning area, where they reproduce before dying (Tesch, 2003) (Figure 5.2).

5.2.2 Atlantic salmon

Many salmon stocks are already below sustainable numbers and conservation limits (Legault, 2005; Windsor *et al.*, 2012). Atlantic salmon are currently listed as lower risk/least concern on the IUCN Red List (World Conservation Monitoring Centre, 1996). However, their status has not been updated since 1996 and other studies have noted a large decline in salmon populations. Atlantic salmon populations have decreased by 90 % in the waters surrounding the United States due to overharvest, passage barriers, habitat destruction, and other factors (Ardren *et al.*, 2015). Atlantic salmon is also listed as Annex II and V on the Habitats Directive (92/43/EEC), meaning that the species requires the designation of Special Areas of Conservation, and whose removal from the wild and exploitation may be the subject of management measures (Joint Nature Conservation Committee, 2015). In the 1970s there was an estimated 2 million adult salmon returning to American rivers, compared to only ~750,000 in recent years (Atlantic Salmon and Sea-run Fish Restoration in Maine, 2015). It has been suggested that the decline of Atlantic salmon in recent years has been caused by increased marine mortality (Hansen *et al.*, 2002). Marine survival, calculated by smolt return rates – the ratio of the number of adult returns produced by a smolt cohort to the number of out-migrating smolts – is poor throughout the Atlantic Ocean (ICES 2008), with low marine survival being one of the top three factors, potentially the most important factor, for the continued low population number for Atlantic salmon throughout the North Atlantic (Atlantic Salmon and Sea-run Fish Restoration in Maine, 2015). A decline from 12 % to 2 % marine survival has been recorded in the last decade (Parrish *et al.*, 1998; Drenner *et al.*, 2012; Thorstad *et al.*, 2012).

Salmon have a life cycle opposite to that of the European eel (Figure 5.4). Salmonid fishes are typically anadromous, meaning that they spend most of their life in the sea, and only migrate into rivers to reproduce. However, the same challenges and noise barriers still apply. Anadromous fishes tend to exhibit a high diversity of life histories and migratory strategies that involve the adoption of different migratory behaviours and conditional responses to the environment (Lacroix, 2013). Adult salmon travel thousands of kilometres in the open ocean and return to their natal freshwater streams to spawn (Cooke *et al.*, 2004). During late spring and summer, Atlantic salmon smolts leave fresh water, and the post-smolts start their migration to their feeding areas in the ocean (European Commission, 2007b). Anthropogenic noise in coastal and estuarine areas has caused concern amongst researchers as movement from the ocean into rivers could be delayed, or even completely prevented, which results in reduced spawning activity in the rivers (Harding *et al.*, 2016). After hatching in freshwater, the salmon move down the river to estuaries where they mature into smolts and enter the open sea. The length of estuarine residence depends on the size, shape, and productivity of the estuary, as well as on water flow patterns and velocities, salinity and temperature, and the individual species and size of the salmon (Thorpe, 1994).

Prior to and during the downstream movement, the salmon display many complex physiological, behavioural, and morphological changes that make them more apt for seaward migration (McCormick *et al.*, 1998). Behavioural changes that have been observed are: increased negative rheotaxis (i.e., downstream orientation), differences in schooling behaviour, decreased agonistic and territorial behaviour, and increased salinity preference (Iwata, 1995). As with the eel, during their metamorphosis for migration it is likely the hearing apparatus will adapt too, though audiogram studies would be needed to confirm this. Smolt development is adversely affected by acidity, pollutants, and improper rearing conditions, and is often more sensitive than other life stages. Unfortunately, the migration corridors of smolts (main-stems of rivers and estuaries) are heavily impacted by pollution, dams, and other anthropogenic activities that may be directly lethal, or increase mortality by delaying or inhibiting smolt migration (McCormick *et al.*, 1998). Estuaries are particularly noisy and so the smolts are exposed to increased noise during a period when they are already sensitive to harm. In some species, immature forms may also move from the sea back into fresh water over winter, so have to pass through estuaries at least twice during their life cycle, but species do differ in the degree to which they are resident or transient there (Thorpe, 1994). There are both advantages and disadvantages of the salmon remaining longer in the estuarine environment, which is considered to be dependent on location and estuary type (Thorpe, 1994). With increasing noise levels in estuaries the disadvantages may begin to outweigh the advantages. For example, increasingly evidence is coming to light that shipping noise – the provident noise source in estuaries – has the ability to mask fish communications and acoustic cues. As predation of salmon by large pelagic fish is the single largest mortality factor at sea identified to date (Lacroix, 2014), noise may impact populations through the masking of predator sounds. Behavioural changes affecting predator avoidance results in even higher predation related mortality levels. Mortality rates in estuaries range from 0.6 % per km to 36 % per km (Thorstad *et al.*, 2012).

If noise has an impact on survival or the ability to metamorphose successfully then migration could be impacted according to some studies; Baggerman (1960) suggested that migration could only occur when the animals are in the proper physiological condition (migration disposition) at the same time as being influenced by the appropriate external stimuli to trigger the event. Migration of Atlantic salmon has the potential to be trans-Atlantic according to studies using isotope analyses and so the likelihood of exposure to noise sources is high in both the smolt and adult life stages (Spares *et al.*, 2007). If noise acts as a barrier at any point on the migration route then the fish could be damaged or delayed, which in turn could make them more susceptible to other threats such as predation. There is a need to understand post-smolt migratory behaviour and the processes that drive and affect it, and evaluate the environment in migration corridors and feeding areas (Lacroix, 2013). Once in the open ocean, Atlantic salmon remain there for at least one winter, although they can remain in their marine phase for a number of years (Lothian *et al.*, 2017).

Also still unclear is the mechanism by which the adults 'home' back to their natal rivers. It has been suggested that pollutants can impact the development of olfactory imprinting of juveniles and their ability to express the trait as adults (McCormick *et al.*, 1998). In some cases, salmon do not return to their natal stream and whilst the reason for this is still speculative, noise barriers have not yet been ruled out as a possible cause.

Movement among habitats is very important in Atlantic salmon populations but there is no single sequence of movements that characterises all populations from fry to smolts. Throughout the salmon lifecycle the fish travel between redd (spawning) sites, summer feeding territories, winter habitats, nursery streams, lower reaches rivers, and the ocean (McCormick *et al.*, 1998). Failure to protect these habitats and the capacity of fish to move freely among them may have detrimental effects on many populations. An interesting point to note is that salmon are able to successfully complete their life cycle without migrating into the ocean (Nilsen *et al.*, 2002), and so could successfully reproduce without moving through noisy areas if they faced a barrier. Such a useful behaviour has not been seen or reported in eels thus far.

The European Commission (2007b) identified biological issues related to the decline in maritime stage Atlantic salmon (Figure 5.3). Although there is no specific mention of noise impacts, two of the top factors identified were predation and growth, both of which can be negatively affected by noise pollution (e.g. Nedelec *et al.*, 2015; Simpson *et al.*, 2016). Several other factors identified have also been observed to be negatively impacted by anthropogenic noise pollution, such as increased disease (Anderson *et al.*, 2011), and higher occurrences of food handling error (Purser and Radford, 2011). However, the impacts of noise on the issues listed by the European Commission (2007) have not been studied specifically for salmon. There were no studies of noise impacts on Atlantic salmon identified during the Chapter One literature search. In areas lacking knowledge, predictive modelling can help to identify potential impacts through applying theory, and providing a rationale for future research to occur.

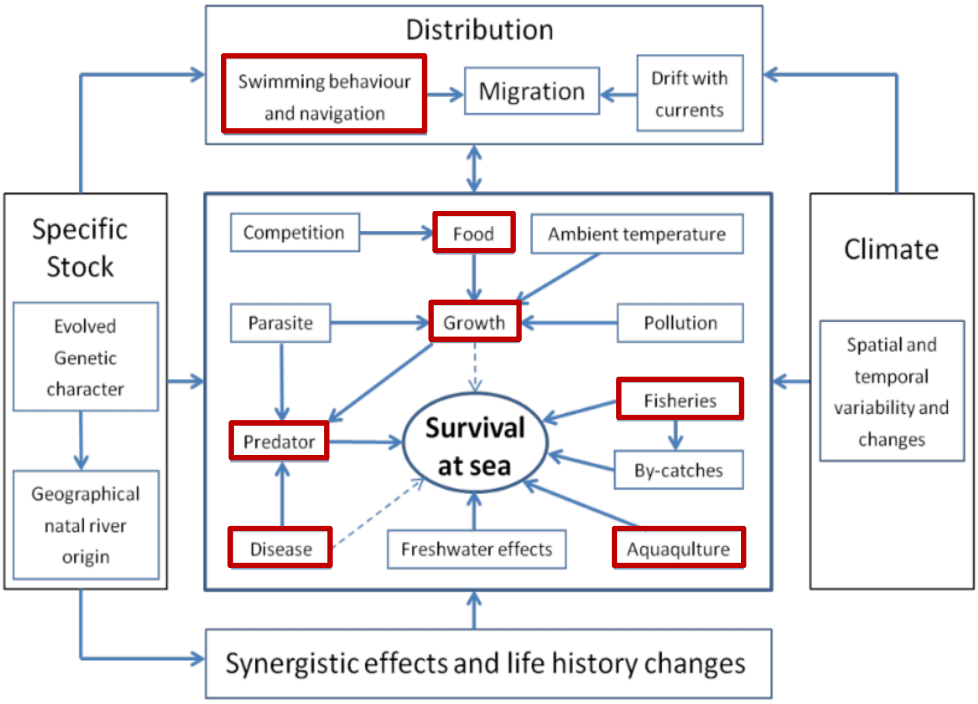


Figure 5.3. An adaptation of the SALSEA-Merge conceptual ecological model of salmon survival at sea (European Commission, 2007c). Each of the boxes is an issue related to the decline of salmon at sea. Although there is no specific mention of noise pollution in the model, the red outlines represent issues that have been observed in at least one study to be negatively influenced by anthropogenic noise pollution.

Long-distance migration of Atlantic salmon is known to result in high levels of mortality (Lothian *et al.*, 2017). It has been stated that significant increases in freshwater and marine survival are needed to improve population numbers, and that small increases in marine survival have a much greater influence on population growth than corresponding changes in freshwater survival (Atlantic Salmon and Sea-run Fish Restoration in Maine, 2015). In the same article, it was concluded that, "Without significant increases in marine survival, recovery of the Gulf of Maine Atlantic salmon is unlikely". For a species experiencing global population decline, a better understanding of the migration process is required for effective management (Lothian *et al.*, 2017).

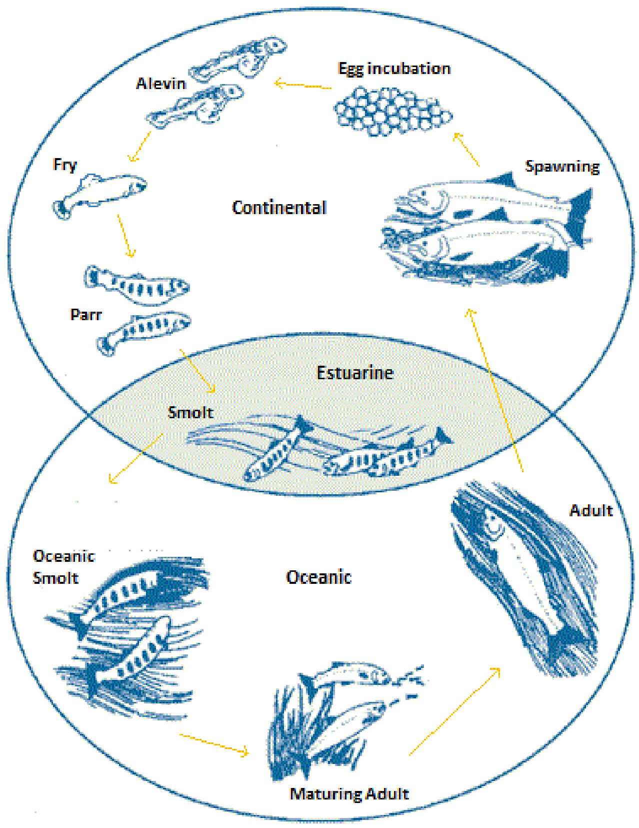


Figure 5.4. Life cycle of Atlantic salmon. Image adapted from Washington State Department of Ecology, 1998.

5.3 Method

5.3.1 European eel

The vessel noise map created in Chapter 3 (from satellite and terrestrial AIS data provided by Orbcomm, covering all vessel types within latitudes 40°N & 60°N and longitudes 20°W & 12°E) was recreated for February 2013 using a frequency of 80 Hz, the eels optimal hearing threshold (Jerko *et al.*, 1989). The model was set to produce SPLcum noise data. February was used in this study as eels have been recorded starting their migration during that month (van den Thillart *et al.*, 2008). To produce a visual output of the results a map was created from the data using an Inverse Weighted Distance interpolation, showing noise exposure levels from low (represented by blue shading) to high (represented by red shading) exposure. To investigate the coverage of the study area that was above the Marine Strategy Framework Directive indicator threshold of 100 dB re 1 μPa rms, the model was re-run to produce SPLav noise data. The same method was used to produce a visual map representation of the data, and the interpolated map was coloured using two classes of classification, below and above 100 dB to show the areas of the ocean that are above the recommended noise level. The map output shows only February, but the yearly

SPLav was also produced as it is yearly baseline levels that the MSFD refers to. A map of eel migration route data (Figure 5.5), provided by the Eeliad Project (an international project to map the migration route of the European eel using pop-up satellite tags) was overlaid onto the modelled vessel noise emissions map. In addition, a map of various possible routes for the eel to use on the outward journey was created. All maps were produced using ArcMap 10.4.1.

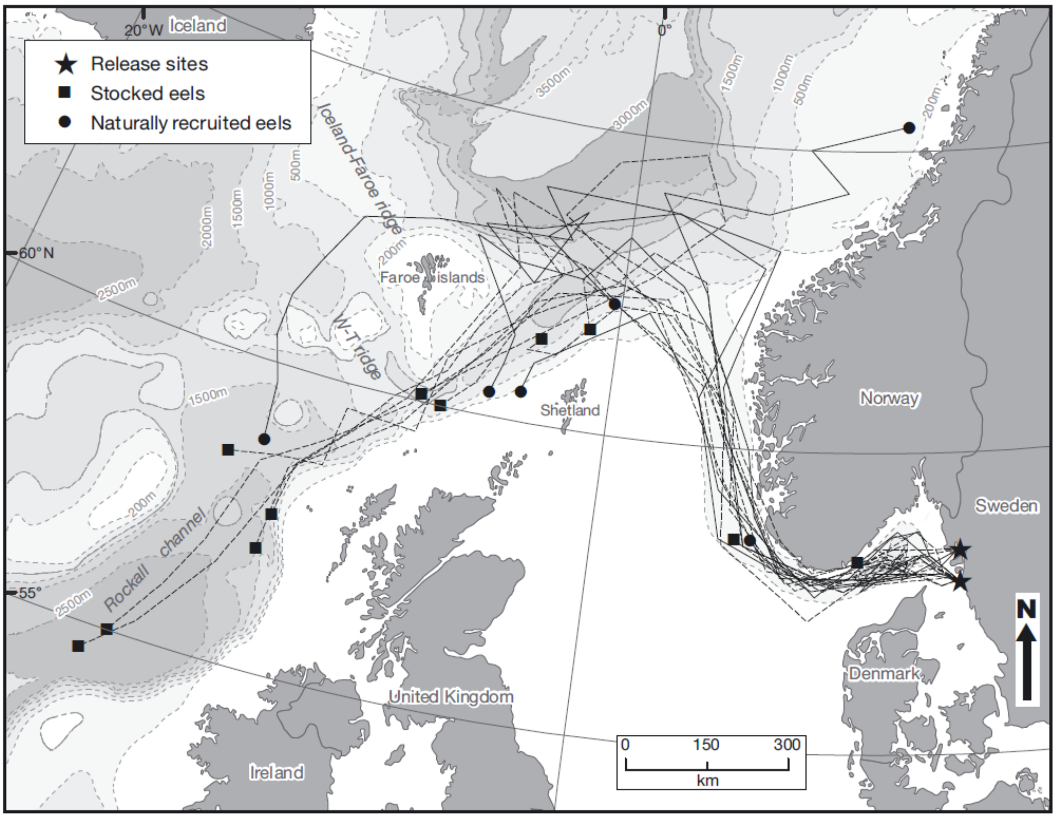


Figure 5.5. Eeliad data showing the routes tagged eels took during their migration, taken from Westerberg *et al.* (2014). The study area for this work encompasses the study area of the Eeliad project. The full area of study reaches from latitudes 40 °N & 60 °N and longitudes 20 °W & 12 °E as shown in Figure 5.6.

5.3.2 Atlantic salmon

A literature search was conducted to identify the migration route of Atlantic salmon, focusing on the North Sea. The resulting publications (European Commission, 2007c; Booker *et al.*, 2008; Dadswell *et al.*, 2010) were used to identify potential routes used by salmon, and a map of a likely route to a feeding ground in the North Sea was created. A vessel noise map, as created in Chapter 3, was produced (from satellite and terrestrial AIS data provided by Orbcomm covering all vessel types within latitudes 40 °N & 60 °N and longitudes 20 °W & 12 °E for 2013) using a frequency of 200 Hz, which salmon are sensitive to (Hawkins and Johnstone, 1978; Harding *et al.*, 2016). The vessel emission map was smoothed and coloured using inverse weighted distance interpolation to produce a useful visual to allow quick identification of noisy areas. As with the European eel data, the model was re-run to calculate the SPLav of 2013 to identify whether noise levels are above or below the threshold levels that the MSFD refers to. To provide a visual output, a map of May was produced, with the interpolated map coloured using two classes of classification, below and above 100 dB re 1 rms to show the area of the ocean. May was used as salmon migrate as post-smolts to feeding areas during late spring and summer (Thorpe, 1988; Mills, 1989), and in late April, early May are found in UK waters and moving away from the Baltic Sea (European Commission, 2007b)

5.4 Results

5.4.1 European eel

There are several areas of high exposure risk for European eels to vessel noise. The areas with the highest received noise levels from the AIS model were the English Channel (peak RL of 166 dB re 1 $\mu Pa^2 s$) and the northern-most point of Denmark – the coastline of Hirtshals and Skagen – (peak RL of 162 dB re 1 $\mu Pa^2 s$). The quietest recorded areas were the Irish Margin in the North Atlantic Ocean, peak RL of 128 dB re 1 $\mu Pa^2 s$. The noise exposure levels produced from the model range from 56 to 171 dB re 1 $\mu Pa^2 s$ (Figure 5.6), however these extremes were not common, with the majority of the map showing levels around 110 – 130 dB re 1 $\mu Pa^2 s$ (as indicated by yellow colouring).

The shortest pathway from the Baltic to the Sargasso Sea would be via the English Channel. The eels in the Eeliad study did not take the most direct route to their destination, through the English Channel (Figure 5.7). The route through the English Channel is by far the shortest distance of the potential routes analysed; route A (Figure 5.7) covered 1,920 km. whereas the route actually observed (route C) using the pop-up tags was 2,530 km.

However, all 17 eels released from the Baltic in the Eeliad project all took the same route via Norway and all but one travelled between the Shetland and Faroe Islands (Figure 5.8). The one eel who diverted around the top of the Faroe Islands remained within the quieter recorded area. Once through the Skagerrak Strait the eels seem to avoid travelling through waters with noise intensity of 130 dB re 1 $\mu Pa^2 s$ or louder (indicated as orange to red on the model map output); they remained in areas of < 130 dB re 1 $\mu Pa^2 s$.

For a frequency of 80 Hz, a vast amount of the ocean is estimated to be above the 100 dB threshold stated in the Marine Strategy Framework Directive (MSFD) (Figure 5.9). All potential routes identified (Figure 5.7) have areas over 100 dB re 1 μPa^{2}s meaning noise measurements and trend monitoring are needed along the European eel migration route, as recommended by the MSFD. The average exposure level for February 2013 was 127 dB re 1 μPa rms, with a 2013 yearly average of 126 dB re 1 μPa rms, which is above the yearly threshold state. The annual average noise exposure reached levels of 129 dB re 1 μPa rms in the English Channel.

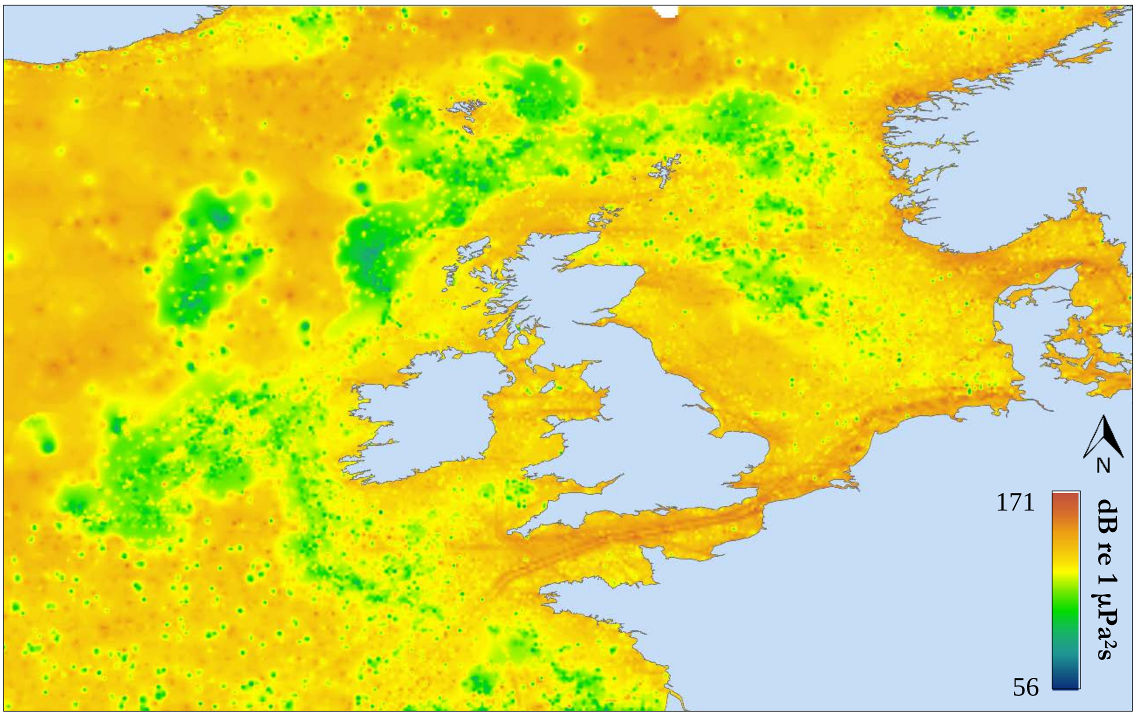


Figure 5.6. Monthly cumulative received level noise emission (dB re 1 μPa^{2}s) from vessels (where frequency is 80 Hz) during February 2013 which can be used to identify mitigation recommendations and priorities for European eel. Noise levels range from 56 dB re 1 μPa^{2}s (blue) to 171 dB re 1 μPa^{2}s (red). Cumulative received levels were calculated for each 1 km grid square, with inverse distance weighting applied to create a smooth map output. The frame stretches in longitude from 20 °W to 12 °E and in latitude from 44 °N to 65 °N.

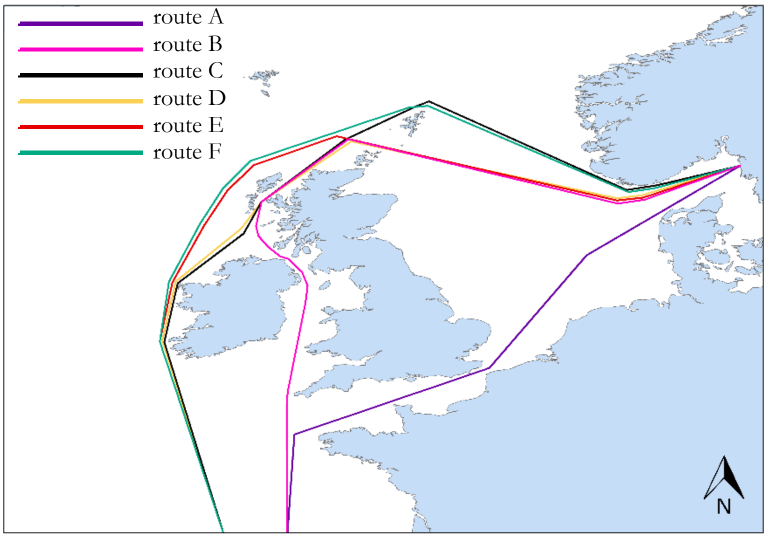


Table 5.1. Length of each route in Figure 5.7 in kilometres. Lengths were measured using Google Earth.

Route	Length (kilometres)
A	1,920
B	2,410
C	2,590
D	2,350
E	2,470
F	2,530

Figure 5.7. Potential alternative routes that could be used by the European eel on their outward migration to the Sargasso Sea. These routes are suggestions to show that there are several options; this is not an exhaustive list. The starting point of the route was the release site from Westerberg *et al.* (2014) to remain consistent with the data used in the study. Lengths of the routes in kilometres shown are given in Table 5.1 to allow comparisons of routes.

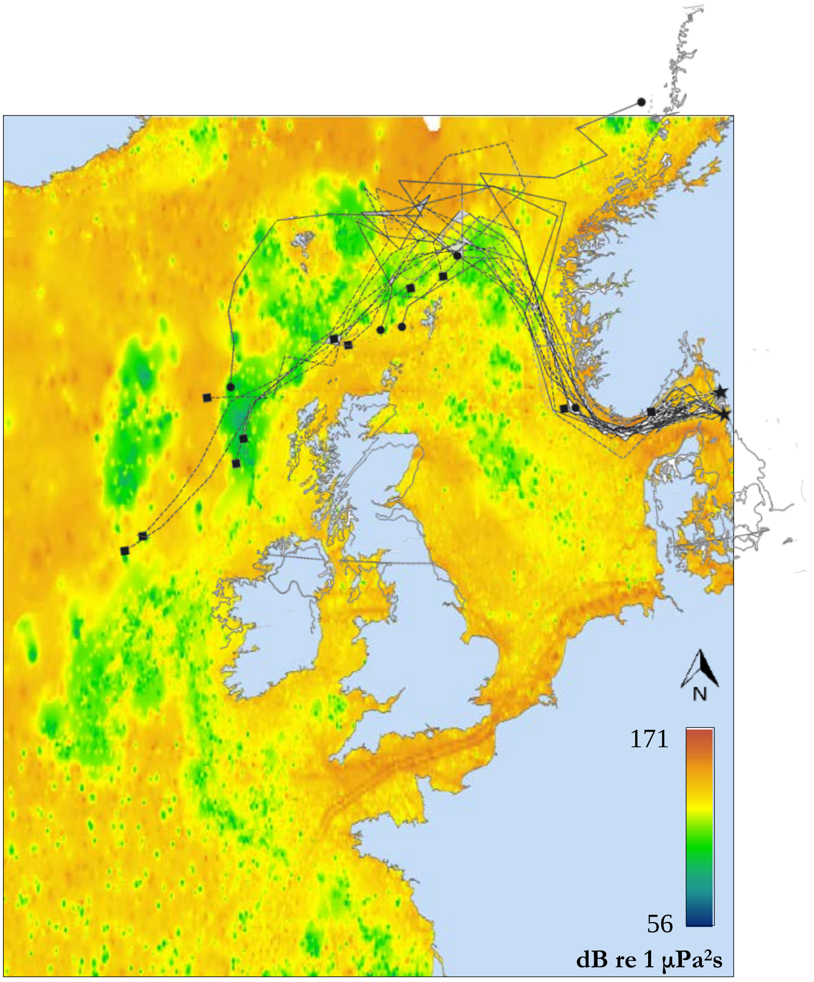


Figure 5.8. The noise map produced from the model overlaid with the eel migration data from the Eeliad Project. The quieter areas follow the same pattern as the eel migration route shown in Figure 5.5. Noise levels range from 56 dB re 1 μPa^{2}s (blue) to 171 dB re 1 μPa^{2}s (red). Cumulative received levels were calculated for each 1 km grid square, with inverse distance weighting applied to create a smooth map output. The frame stretches in longitude from 20 ºW to 12 ºE and in latitude from 44 ºN to 65 ºN. Due to different coordinate projection systems, the two maps did not overlay perfectly. A grey outline of the landmasses shows where the maps did not fit together

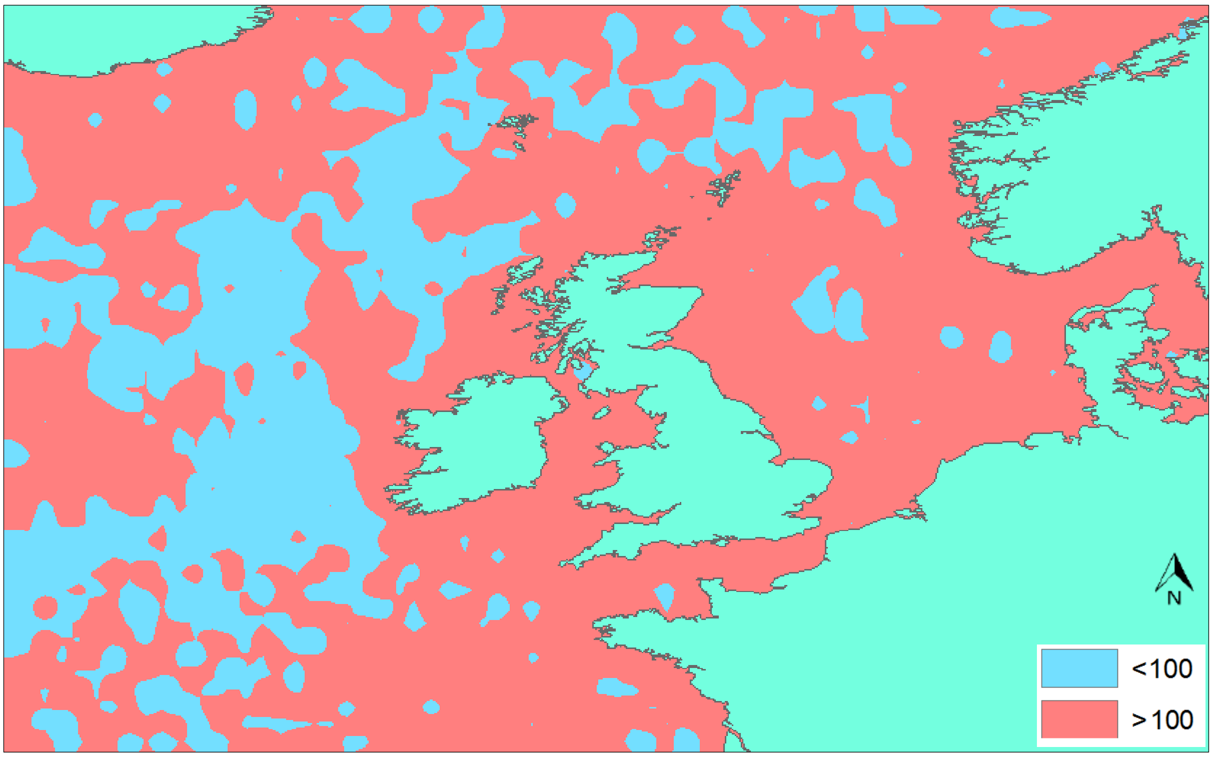


Figure 5.9. Vessel emission noise map of 2013 to identify the areas exceeding the 100 dB re 1 μPa rms threshold levels stated in the Marine Strategy Framework Directive (2008/56/EC). The red areas represent received levels greater than 100 dB re 1 μPa rms. The majority of the ocean in the study area had noise levels above 100 dB re 1 μPa rms, and therefore, above the recommended average noise level over a year (yearly average was 126 dB re 1 μPa rms).

5.4.2 Atlantic salmon

The simulated and observed routes of salmon migrations (Figure 5.10) do not seem to pass through any areas with noise intensity higher than 150 dB re 1 μPa^2s (Figure 5.11). The loudest point on the map that the salmon cross, at an intensity of 147 dB re 1 μPa^2s, is a small area on a route simulated by the European Commission (2007b) (Figure 5.12). Most of their journey is spent in noise levels of 100 – 130 dB re 1 μPa^2s, which has been observed to cause significant negative reactions in 22 studies, and also shown not to impact fish in 4 studies (*see* Table A5.1 for full list of references (although not many studies have been conducted on Atlantic salmon specifically)).

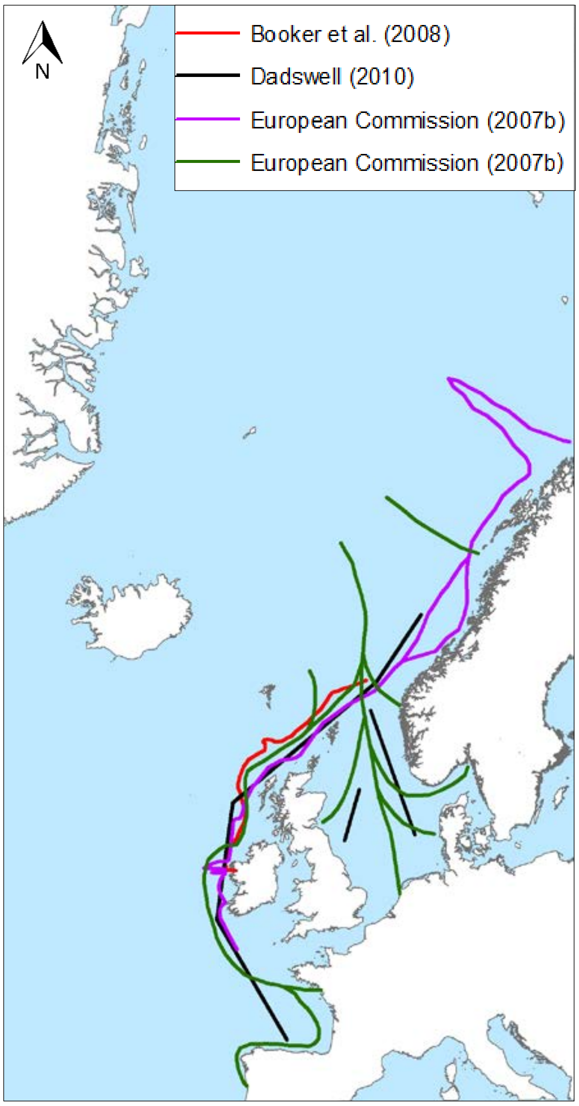


Figure 5.10. Map of possible Atlantic salmon migration routes to a feeding ground in the North Sea, produced using data taken from several publications (European Commission, 2007c; Booker *et al.*, 2008; Dadswell *et al.*, 2010). For original maps, see Appendix 5. The frame stretches in longitude from 30 ºW to 20 ºE and in latitude from 40 ºN to 80 ºN. The WGS 1984 geographical coordinate system was used so that the image could be compared to the previous publications listed as data sources.

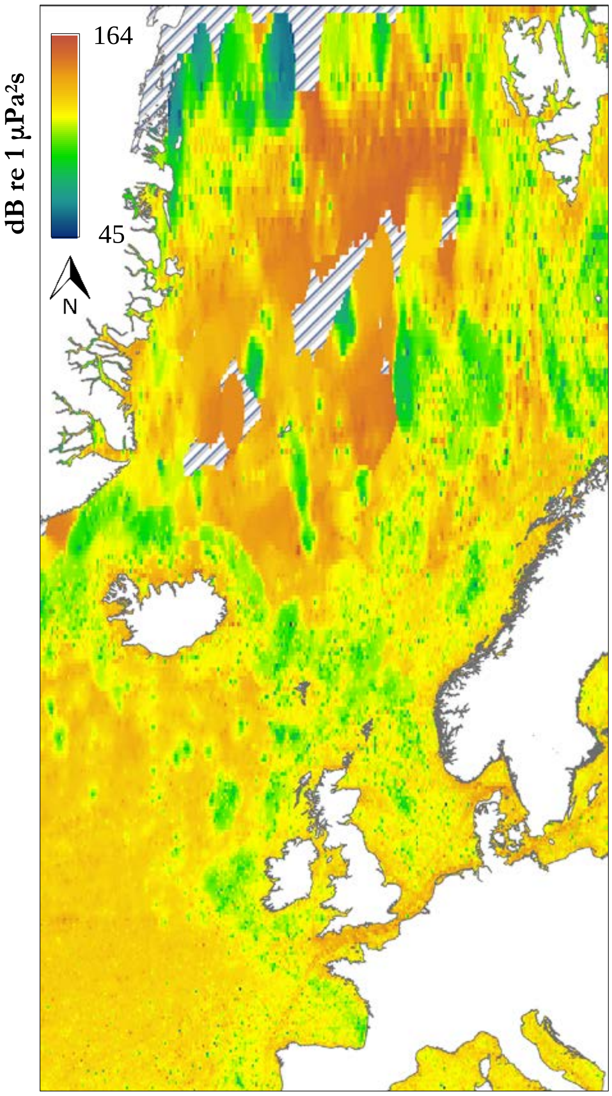


Figure 5.11 Cumulative received level noise emission (dB re 1 $\mu Pa^2 s$) from shipping (where frequency is 200 Hz) during May 2013. Noise levels range from 45 dB (blue) to 164 dB (red). Cumulative received levels were calculated for each 1 km grid square, with inverse distance weighting applied to create a smooth map output. The hatched area represents no available data. The frame stretches in longitude from 30 ºW to 20 ºE and in latitude from 40 ºN to 80 ºN. The WGS 1984 geographical coordinate system was used so that the image could be compared to Figure 5.9.

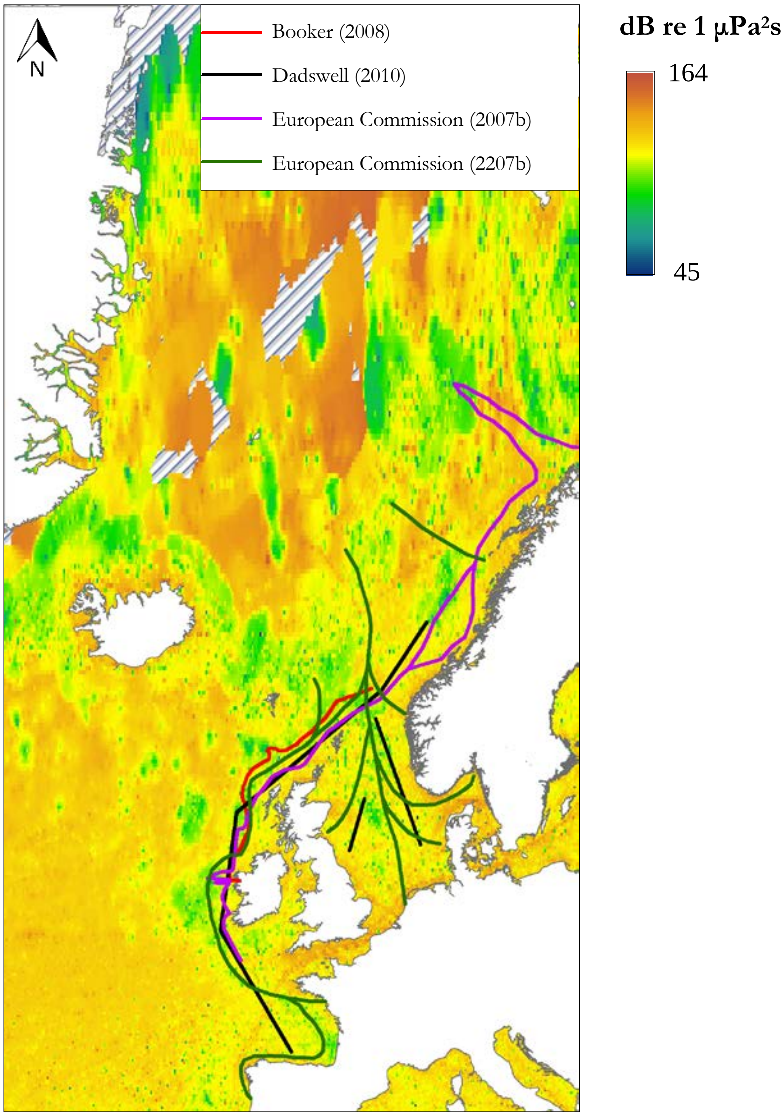


Figure 5.12. Overlay of the cumulative received level noise emission (dB re 1 μPa^{2}s) from shipping (where frequency is 200 Hz) during May 2013, and the possible migration routes taken by Salmon in the North Sea (European Commission, 2007c; Booker *et al.*, 2008; Dadswell *et al.*, 2010). The frame stretches in longitude from 30 °W to 20 °E and in latitude from 40 °N to 80 °N. The WGS 1984 geographical coordinate system was used so that the image could be compared to Figure 5.9. Hatched areas represent no available data.

5.5 Discussion

This chapter helps to identify noise hotspots and problem areas that two migrating species may face.

5.5.1 European eel

European eel have to travel several thousand kilometres from their home river to the spawning ground in the Sargasso Sea, and are subject to a number of dangers along the way. This is evidenced by eels from the Eeliad Project being predated upon during the study; of the eels who migrated for longer than one week, 41 (~50 %) suffered predation, 10 of which (~25 %) occurred when in oceanic waters (Westerberg *et al.*, 2014; Righton *et al.*, 2016). Predation is a natural occurrence and obstacle for the eel during their migration, however, vessel noise has been observed to hinder antipredator behaviours, leading to 50 % less chance of an eel startling in response to a predator (Simpson *et al.*, 2015). This diminished antipredator response was observed in the laboratory at 148 dB re 1 µPa rms, which is exceeded in some parts of the oceans (SELcum reached as loud as 171 dB re 1 μPa^2s during May 2013 – additionally, May is not the loudest month (*see* Chapter 3 for temporal differences)).

The eels in the Eeliad Project were released from Sweden and destined for the Sargasso Sea. There are multiple routes the eels could have taken to reach the Sargasso Sea from their release site (Figure 5.7), the most direct of which would be through the English Channel (route A, Figure 5.7). The reason eels choose a particular route is unknown. A logical argument for the eels to take the longer route around the top of the UK, instead of the direct route through the English Channel, is that the eels are using the currents to aid their movement to reduce their energy expenditure. However, the movement of water in the North Sea is against the eel's trajectory on both the route used (Figure 5.7, route C) and the shortest distance route (Figure 5.7, route A), with a larger magnitude of water volume flowing against the eel on their actual chosen route (Figure 5.13).

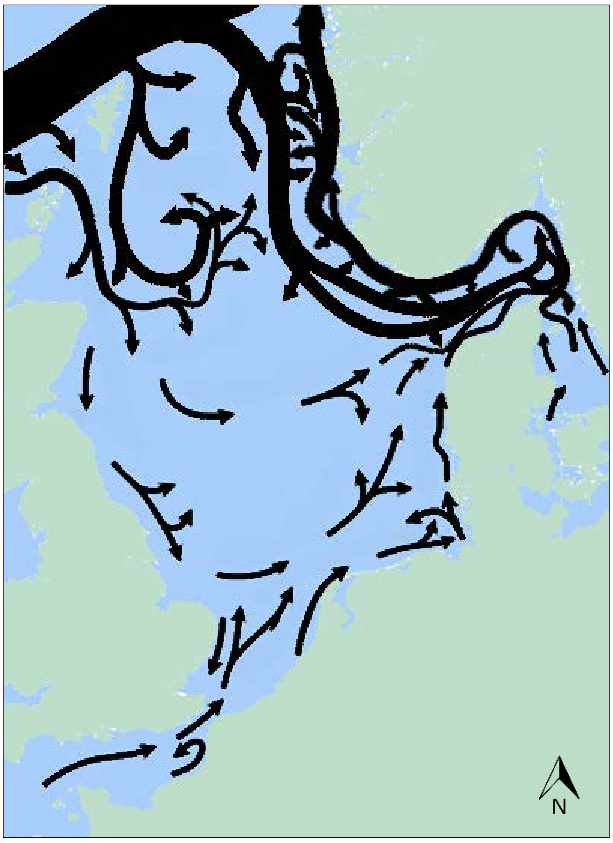


Figure 5.13. Map of North Sea currents. Arrows indicate direction of water movement. Width of the arrows indicates the magnitude of volume transport. For the eels outgoing migration, they must swim against the current whether they travel around the top of the UK or through the English Channel. They also encounter larger oncoming currents on their preferred route C (Figure 5.7) than they would on the shorter route A (Figure 5.7).

Westerberg *et al.* (2014) suggested that the eels are using the Norwegian Trench as a guide and make use of the deep channel, but this was speculation as the majority of the eels journey still remains a mystery.

There was one tag that showed an eel moving north-east away from the rest of the tagged eels. Whether this eel was alone or part of a school was not reported in the Eeliad results. Vessel noise has been reported to affect homing accuracy in fish (Sara *et al.*, 2007), although this particular adverse behaviour has not yet been studied for European eels specifically. Investigating whether vessel noise can impact homing accuracy in migrating species would provide further evidence of the need for mitigation along migration routes, if negative impacts were to be observed.

This chapter does not identify whether noise in particular is a consideration in migration route choice, but does provide information that can be useful in mitigation decisions or future research questions. The model suggests a study is needed to question whether a reduction in vessel noise in the English Channel would increase the likelihood of eels using the channel as a passageway on their migration (*see* Chapter 4 for various mitigation scenarios that could be implemented in the channel).

The map depicts one of the nosiest areas, in terms of vessel noise exposure, is unavoidable for eels migrating from the Baltic Sea (Figure 5.6). To enter the North Sea the eel must travel through the channel at the northern-most point of Denmark. This area reached a peak cumulative exposure level of 162 dB re 1 $\mu Pa^2 s$, and an annual average exposure level of 126 148 dB re 1 μPa rms. Behavioural reactions have been observed in European eel from noise exposure of 148 dB re 1 μPa rms, and so parts of their journey, especially in the Baltic Sea, could be damaging to them. The eel could become more susceptible to predation and experience physiological impacts such as altered ventilation rate (Simpson *et al.*, 2015), or show avoidance, changes in swimming direction, increased hiding behaviours or increased stress levels. All these have been observed in other fish species at noise exposure above 100 dB re 1 μPa.

The results of this chapter include only noise exposure of vessels, and no other anthropogenic noise sources. Although, AIS-based models can predict vessel noise exposure, they cannot incorporate other noise sources and so it must be recognised that the levels reported are without cumulative noise from other sources. Pile driving and seismic survey noise can propagate vast distances; Slotte *et al.* (2004) observed behavioural reactions occurring in fish up to 30 km from an air-gun noise source. The map outputs show the vessel noise that fish will encounter, not the actual noise levels in the ocean comprising of all noise sources.

Currently, there is only one published audiogram for the European eel (Jerko, 1989), yet, it is known that the eel undergoes transformations for migration which could affect the physiological and behavioural responses to noise pollution. Future work conducting audiograms on the different life stages of the eel would provide an accurate hearing threshold for each life stage. The audiograms could then be used to determine if the levels of noise shown by the noise map models will in fact have an impact on the eel during migration at different life stages. Biological information on species under study can help to increase the usefulness of mitigation models, making them more appropriate for the species in question, and more likely to produce recommendations that will benefit the species.

5.5.2 Atlantic salmon

The Atlantic salmon routes identified (Figure 5.7) all remain close to landmasses, although outside of coastal waters. This behaviour is likely linked to salmon using rheotaxis - a directed locomotor response to flow (Fraenkel and Gunn, 1961) – which was suggested by Royce *et al.* (1968) as a mechanism for salmon to find feeding grounds in the ocean. Utilising a behaviour such as rheotaxis means the salmon stay within quieter regions of the ocean, avoiding deeper water columns that allow greater propagation of noise, and busy coastal areas. Once the salmon move further into deep water towards their feeding ground they encounter louder areas. During May 2013, the noise exposure on the documented migration routes of Atlantic salmon remained below 147 dB re 1 $\mu Pa^2 s$. The migration routes identified in the literature search (Figure 5.9) tend to be in open water, meaning small patches of noise can be navigated around and avoided, with the exception of the channel joining the North Sea to the Baltic Sea. However, the salmon have to travel to their feeding grounds and it is a concern that the feeding area the salmon are migrating to seems to be a noise hotspot, with the model reporting the area as above 130 dB re 1 $\mu Pa^2 s$, and reaching peak levels of 157 dB re 1 $\mu Pa^2 s$.

The exposure risk of salmon to vessel noise, therefore, is greater once the salmon reach their feeding grounds. As no studies could be found on the impacts of vessel noise on Atlantic salmon it is difficult to know at what intensity level significant negative impacts occur. However, as mentioned in Section 4.2.2, significant negative reactions have been observed in response to vessel noise greater than 100 dB re 1 μPa in at least 22 studies (Table A1.1).

May 2013 had the loudest estimated levels for vessel noise exposure for the channel joining the Baltic Sea and North Sea. The European Commission (2007b) noted late April / early May to be the time of year when salmon start their migration from the Baltic. This model provides the recommendation that mitigation is needed in that channel during May to reduce the noise exposure level and remove potential harm or barriers to the Atlantic salmon migration. As it is May in particular when the fish pass through the noise hotspot, a temporal closure or reduction of vessel traffic during this month would reduce the noise level (*see* Chapter 4 for various mitigation scenarios that could be used).

The results from this chapter suggest that it is the feeding ground itself, and not the migration route, that would need priority focus for mitigation for Atlantic salmon. Once in the feeding area, the high level of noise could influence the fish either behaviourally or physiologically. Increased food handling errors, reduced predator avoidance, and alterations in schooling have all been observed as a response to vessel noise in other species (Sarà *et al.*, 2007; Purser and Radford, 2011; Simpson *et al.*, 2015). A recommendation to reduce risk of exposure once at the feeding ground could be to designate the area as a no vessel zone, or a Marine Protected Area, or place speed restrictions on vessels travelling through or near the area to lower the source level emissions from the vessels (*see* Chapter 4 for mitigation scenarios relating to MPA designation and speed restrictions).

5.6 Chapter conclusion

Modelling alone can prove useful (such as the Source Level modelling in Chapter 3, and the scenario modelling in Chapter 4) but by including biological information as well it makes the outcomes and recommendations of a model more pertinent. For example, creating noise maps at the octave bands recommended by the MSFD can show overall trends, but is not particularly beneficial if planning mitigation for a specific species. If a species optimum hearing sensitivity is around 80 Hz as in the case of European eel, mapping their distribution to a 125 Hz noise map would not give the most accurate information for planning and mitigation purposes. In a dolphin study, Evans et al (1992) found that quieter, faster vessels caused more disturbance for dolphins than slower, larger vessels. Noise emitted by high-speed vessels rises above ambient levels (and thus becomes detectable by the dolphin) only a short time before closest contact, thereby provoking a startle response in the animal. This is because the frequency of noise emitted from the smaller, faster vessels is higher than those emitted from larger, slower vessels. Another study on the south coast of the UK discovered that the increase in underwater noise over the summer months, caused by a dramatic increase in pleasure craft, is sufficient to impair communication between bottlenose dolphins and to reduce their echolocation performance (Wharam *et al.*, 2006). As different vessels emit difference frequencies of noise, an increase or decrease in a certain type of vessel could influence various species differently. This highlights the necessity of including any available audiogram data when modelling noise pollution impacts on marine species.

Identifying migration routes and distributions of species allows species data to be overlaid on noise model maps to show possible exposure risk. Both European eel and Atlantic salmon are subject to risk of vessel noise exposure, but at varying levels. The overlaid map helps to easily identify the noise hotspots and areas of high exposure risk for each species.

The different aspects that have high potential to impact the species under study should be considered when implementing mitigation measures. Modelling such as this provides a way of prioritising mitigation; efforts could be focused on areas of higher noise exposure, areas of high population density, or areas such as the channel leaving the Baltic Sea where no alternative route is possible. The Baltic channel and the English Channel were both identified as noise hotspots, however, if the English Channel can be avoided (as appears to be the case even if it results in a longer journey) then the species can still reach its destination and continue its life cycle. If eels avoid the noisy area in the Baltic turn back, or take too long to cross, the area could become a barrier to migration, and as such, should be rated highly in need of mitigation, and classified as a priority. When considering priorities, it is important to ensure both the noise exposure and the biological information are included, as the latter can aid decisions especially when trying to put mitigation measures in place for multiple species. For instance, Atlantic salmon have the ability to choose whether to migrate or return to the rivers, and so if they face a noise barrier they cannot pass, they can return to their river for another year. European eel does not have that option. Noise reduction mitigation should factor in biologically relevant information.

Modelling must work hand in hand with field and laboratory studies in order to test model predictions and assumptions, better parameterise and initialise the models, and strengthen the models' capabilities through repeatability (Werner *et al.*, 2007). This chapter uses just one month of data to demonstrate the usefulness of combining AIS noise model mapping with collected biological information. There is no limit to the time period that can be studied, be it several hours or several years. The method provides useful information for use in planning mitigation of noise pollution from vessels, identifying noise hotspots and species distributions, and establishing where the two overlap. Calculating quantitative data from real AIS and species data allows meaningful recommendations to be made, and priorities set.

6. Chapter Six: General discussion

6.1 Summary

According to the World Health Organization, anthropogenic noise is one of the most hazardous forms of pollution and has become omnipresent within terrestrial and aquatic ecosystems (Andrew *et al.*, 2002; World Health Organization, 2011). The impacts of anthropogenic noise can be particularly prevalent in aquatic environments, where sound travels further and faster before attenuation than in air (Williams *et al.*, 2015b). Some anthropogenic noise is now considered a global pollutant, featuring in national and international legislation (e.g. the European Commission Marine Strategy Framework Directive (2008/56/EC) and the European Union Water Framework Directive (2000/60/EC).

Species have evolved to cope with and utilise natural sounds in the marine environment, but the addition of anthropogenic noise pollution is exposing fish to increase risk of harm from noise (Simmonds *et al.*, 2014). Some fish species are no longer able to cope with the levels of noise pollution being emitted from human sources, and adverse reactions have been observed in at least 51 fish species (*see* Table A1.1). Fish are able to adjust their behaviour to suit the environment, as seen in studies on the Lombard Effect (e.g. Luczkovich *et al.*, 2012), but in cases where adaptation does not occur quickly enough, they are forced to leave the area (Badyaev, 2005; Van Buskirk, 2012).

For successful mitigation, an improved understanding of impacts is needed, which requires exposure levels and impacts of all noise-producing activities to be carefully monitored over suitable time-frames and spatial scales (Simmonds *et al.*, 2014). Climate-driven change in marine ecosystems occurs over a range of frequencies from inter-annual to century-long timescales and over a range of amplitudes (Hawkins *et al.*, 2009; Firth and Hawkins, 2011). Pollutants work in similar ways to climate change, with levels fluctuating over time and having wide-spread impacts. Sustained observations have long been recognised as important in disentangling climate-driven change from anthropogenic impacts (Edwards *et al.*, 2010), and so the same should hold true when investigating pollutants such as noise. Data can be used to establish a baseline from which all future data or predictions can be extrapolated. Climate change, for example, can use past warm and cool periods to provide probable causes for recent changes and formulate predictions for future scenarios (Philippart *et al.*, 2011). Such long-term predictive datasets have already influenced emerging UK and EU policy, and are gradually becoming more prominent in the evolution of marine spatial planning (MSP) (Hawkins *et al.*, 2013). Information on past trends is essential in order to make informed future predictions and direct policy decisions. By mapping past trends in ocean shipping noise, and the migration behaviour of various species, the overall trends can be determined.

Pollutant trends can be analysed using map-based tools; mapping the density, concentration and dispersal of pollutants over time or geographic location can identify trends. Map-based tools can be used to study the impacts of wide-spread pollutants such as vessel noise through long term measuring of noise or through Automatic Identification System (AIS) models, as shown in Chapter 3.

Previous studies have called for areas where there is a high prevalence of shipping traffic to be monitored so that any impacts occurring on marine species can be addressed and mitigated for (Merchant *et al.*, 2012; Williams *et al.*, 2015b). There is, therefore, a need to identify areas of high risk within the marine environment and develop methods to assess the long-term noise exposure and trends (Erbe *et al.*, 2012, 2014; Merchant *et al.*, 2012). However, if our knowledge on ocean noise impacts is ever to get ahead of the curve of the rapid industrialisation of the ocean, we are going to have to predict impacts before they occur, and guide effective mitigation for the most vulnerable populations (Williams *et al.*, 2015b).

AIS modelling has the ability to predict future scenarios and guide mitigation to reduce impacts of noise pollution on fish. Such models can identify the potential benefits of source level mitigation such as noise-reducing technologies. Alongside reducing noise levels at source, impacts on sensitive species can also be reduced by temporal or spatial separation. Predictive models can direct mitigation focus by identifying areas of high noise exposure risk and priority habitats. Modelling combined with field research will continue to help in the identification of concentrations of noise-sensitive species. Biological information collected from experimental research, such as hearing sensitivity data (audiograms) and distribution patterns, can add another dimension to AIS models to make predictions species specific. Such research should be prioritised, as should the identification of small populations and species whose diminishing population would adversely affect ecosystems as a whole.

6.2 Research impacts

The overall aim of this thesis was to address three key questions associated with marine noise pollution research: which fish species and associated habitats should be given priority status; where should research, mitigation efforts and legislation be focused for maximum benefit; and can researchers overcome challenges in noise mapping and AIS data modelling to create useful tools for predicting future noise mitigation strategies.

Chapter 1: Literature review

Aim: to undertake a comprehensive literature review to identify gaps in knowledge and pinpoint areas for targeted research.

A thorough literature review was undertaken to collect and collate all relevant literature pertaining to the impacts of anthropogenic noise emissions on fish. A meta-analysis identified the need for new methods to aid mitigation decision-making, particularly with regards to long-term trend modelling and prioritisation of research and mitigation. Two forms of mitigation were identified; the first involved the use of biological information, and the second applied changes to the sound source to minimise effects. With so many different aspects of noise pollution research identified as a priority, and with no universally acknowledged metrics or generalizable research available, it was clear that quantitative or semi-quantitative methods were needed to help direct mitigation focus to the most vulnerable areas. The importance of modelling in noise pollution research was recognised as a powerful tool that can be used effectively to aid in the collection of data, dissemination of information, and for analytical purposes.

Output:

- Publication: Neenan STV, Piper R, White PR, Kemp P, Leighton TG, Shaw PJ. 2016. Does masking matter? Shipping noise and fish vocalizations. *Advances in Experimental Medicine and Biology* **875**: 747-753.

Chapter 2: Establishing priorities for noise pollution impact research on marine fish: a value- and susceptibility-based framework

Aim: to create a method for prioritising fish species vulnerable to noise pollution impacts to guide research and mitigation.

Understanding the dynamics of ecosystems and how threats can influence those dynamics can aid in prioritisation of the most important or most manageable threats. Identifying the value of a species to ecosystem dynamics, and understanding the consequences if that species were to suffer adverse reactions from noise pollution, is an important aspect of prioritising mitigation and research. The first form of mitigation, that of using biological information, was used in this chapter to create a prioritisation model. The model assessed species likely exposure to noise and severity of harm should exposure occur. Data was assimilated using online databases and previous publications. Having a prioritisation tool that can quantify such information and pinpoint species, habitats and noise sources which are most likely to influence marine ecosystems allows research to be focused and justified. The model was run on a UK case study and identified Atlantic cod as being a priority species. During the literature search and meta-analysis in Chapter 1, only two studies on the impacts of noise pollution on Atlantic cod were found (Engas *et al.*, 1995; Nedelec *et al.*, 2015). This Chapter identified a large gap in experimental data and emphasises the need for prioritisation models as important species may be overlooked and not studied without such tools to guide and focus research.

Output:

- Submitted manuscript: Neenan, S.T.V., White, P.R., Leighton, T.G., Saunders, J. Shaw, P.J. Establishing priorities for noise pollution impact research on marine fish: a value- and susceptibility-based framework.

Chapter 3: Creation of an ocean noise map: using AIS data to model shipping noise emissions

Aim: to create a method for modelling vessel source level noise with a visual output (mapping) of vessel emissions.

To investigate the second form of mitigation, the manipulation of the noise source level, a model was created that uses AIS data to map cumulative Sound Exposure Levels of vessels on the ocean. The output from the model can be easily interpreted and used to identify ocean vessel noise exposure trends over time. The model can be used for any AIS data transmitted from any location and is easily repeatable. Large datasets (hundreds of millions of transmissions) can be quickly and accurately interpreted into useful information than can inform mitigation and research. To evidence the ability of the model a UK case study was run through the model and reported in this chapter. The applications of such a model are many-fold, be it analysing historical or real-time trends, or predicting future scenarios.

Output:

- Poster presentation at the 4th International Conference on the Effect of Noise on Aquatic Life – Dublin – 10-16 July 2013.

- Oral presentation at the 4th International Conference on the Effect of Noise on Aquatic Life – Dublin – 10-16 July 2016.

- Publication: Neenan, S.T.V., White, P.R., Leighton, T.G., Shaw, P.J., 2016. Modeling vessel noise emissions through the accumulation and propagation of automatic identification system data. *Proceedings of Meetings on Acoustics* 27 (2016), 070017. http://dx.doi. org/10.1121/2.0000338.

- Manuscript to be submitted: Neenan, S.T.V., White, P.R., Leighton, T.G., Shaw, P.J. A year of ocean vessel noise: using AIS data to map shipping noise emissions.

Chapter 4: Using AIS data modelling of noise emissions as a predictive tool

Aim: to show the applications of an AIS-based noise exposure model for predicting the outcomes of five possible mitigation scenarios.

Mitigation management can be challenging, especially when measures are costly to implement, disruptive to businesses, and time consuming to establish. Deciding which mitigation measures should be implemented and in what order of priority is difficult. The ability to predict the outcomes of mitigation measures helps simplify such decisions. Five scenarios were run through the AIS model to predict likely outcomes and trends of the mitigation measure under consideration. This allowed direct comparison of all the scenarios, providing an idea of priority order, and estimating the likely impact each scenario would have. Decision makers are then in a position to weigh the costs against any potential beneficial outcome of the measure under scrutiny and make a much more informed decision. After investigation of the 5 scenarios, the highest priority mitigation measure suggested by the model was to apply speed restrictions on vessels in areas of ecological importance. The influence that speed restrictions have on cumulative vessel noise exposure levels were considerably greater than any other explored method of mitigation; its effects are immediate and can be done voluntarily until regulations are put in place, and are of particular value in protected areas and surrounds.

Output:

- Manuscript to be submitted: Neenan, S.T.V., White, P.R., Leighton, T.G., Shaw, P.J. AIS data modelling as a predictive tool for prioritising vessel noise mitigation methods.

Chapter 5: Combining prioritisation, historical and predictive modelling as a method for mitigation

Aim: to combine prioritisation, biological and source level modelling to predict impacts on species migration.

The previous chapters have shown how the two separate forms of mitigation can be used in modelling to help prioritise and focus research and mitigation efforts. The combination of both methods provides powerful mitigation tools to help identify trends and prioritise species specific mitigation. Biological information of fish species was combined with AIS vessel noise exposure modelling to identify areas of high noise exposure risk and predict potential impacts of vessel noise on populations. Migratory species travel large distances and come into contact with varying levels of noise exposure. The model presented in this chapter allows researchers and mitigation decision makers to identify likely hotspots of noise that may act as barriers to important behaviours such as migration and feeding. The vessel noise exposure levels in the Baltic sea are high enough to cause avoidance behaviour in migratory species, potentially even blocking the migratory route, and so should be considered a high priority area for mitigation.

Output:

- Manuscript to be submitted: Neenan, S.T.V., White, P.R., Leighton, T.G., Shaw, P.J. Mitigating vessel noise impacts on migration: a combination of prioritisation, historical and predictive modelling

Chapter 6: Thesis discussion

Aim: to discuss the relevance and part played by each method and model designed and explained in previous chapters and its potential role in noise mitigation

Individually, both forms of mitigation, using biological information and reducing source level emissions, have a role to play in alleviating the known impacts of vessel noise pollution on marine fish species and populations. A combination of both approaches allows even more information to be collected and utilised to produce more accurate and useful predictive models for decision makers. The models created in this thesis identify priority species and habitats for research and mitigation purposes.

The research in this thesis will be of interest to researchers investigating the impacts of noise pollution on all marine species, not just fish species. The AIS-based model can also be used for vessel noise mitigation for marine mammals. Those involved with mitigation decision-making can use the models to help guide their focus and priorities whilst environmental consultancies can utilise the models for use in Impact Assessments. Williams *et al.* (2015) suggested that precautionary measures be put in place for the quietest areas of the oceans, to create acoustic refuges, and experimental control sites, that will help to improve our understanding of the ecological cost of increasing anthropogenic noise. AIS models can be used to identify those areas that remain quiet enough to be considered control locations for future research. The Prioritisation Index has wider applications in that it can be adapted for threats other than noise pollution, and used for both marine and terrestrial systems. The publications produced as a result of this thesis will demonstrate the models using UK case studies and provide information on temporal changes in vessel noise exposure throughout the year. Information on vessel noise exposure risk on two migratory species will contribute to the growing research on their migratory journeys which are still widely unknown; knowledge on the Atlantic salmon and European eel marine phases are particularly lacking.

6.3 Future research

All research work undertaken to answer specific questions inevitably leads to even more questions being raised. There are a number of particular areas that require further investigation and also recommendations for future work.

One of the key knowledge gaps identified in Chapter 1 was the lack of audiogram data for fish species. Audiograms were found for only 77 species of the near 32,000 extant fish species. Biological information such as audiograms can help to guide mitigation decisions and provide additional information for ocean noise maps. Fish have shown a response to both sound pressure levels and particle motion (Hastings *et al.*, 1996; Horodysky *et al.*, 2008; Wysocki *et al.*, 2009), and so any legislation developed must consider both these metrics of a sound field. More information is needed on the different auditory systems of fish - how a noise (both pressure and particle velocity) might affect the auditory system, and how noise sources differ from one another - before any extrapolation of impacts can be made for populations of species (Popper and Hastings, 2009b). There is limited knowledge of the levels of vibration and particle velocity that would affect fish species, as well as the background levels that exist in the ocean. This is because the techniques and sensors for measuring vibration and particle velocity (both in the water column and along the seafloor) are still under-developed, and information about calibration standards is lacking (Robinson *et al.*, 2014). Continuing research on particle velocity will provide increased knowledge and understanding, and new findings should then be incorporated into both new and existing legislation. Chapter 5 produced noise exposure maps at the optimal hearing sensitivity of the species under concern (80 Hz for European eel (Jerko *et al.*, 1989) and 200 Hz for Atlantic salmon (Harding *et al.*, 2016)) to provide more accurate information on potential noise hotspots and barriers to migration. The maps produced from the specific frequencies in Chapter 5 differed from the maps produced at 63 Hz and 125 Hz in Chapter 3 which emphasises the importance of tailoring the models to the specific species being investigated. The vessel exposure maps are still useful for species without known audiogram data or optimum hearing sensitivity but knowing such information would increase the accuracy of the model, allowing more informed mitigation decisions to be made. Continuing work would also include looking at the migration routes of other species, and investigating known spawning grounds such as the Sargasso Sea to see if noise could be impacting reproduction at those sites, and whether mitigation is needed.

One of the main challenges moving forwards is to deliver to regulators evidence that can be directly useful to aid management and policy decisions. An issue environmental consultants face is that the people who are making decisions are not always as well informed on the topic as those who conduct the research and write the reports. A caveat with noise mapping research is that the methods are often not stated clearly. Several papers have shown noise maps of the ocean with no specific explanation of how the map was produced, the calculations performed, or how the authors came to the conclusions that they stated (e.g. Erbe *et al.*, 2012). Additionally, even if scientific publications were easy to understand by the layman they are not always accessible (unless open access and easily found through a search engine) to the people involved in the decision-making processes for mitigation. It is for this reason that modelling can be so beneficial in mitigation management. Models do not have to stay as calculations in a publication, they can be made into online tools and software developed that can be freely disseminated and downloaded by anyone with an interest in the topic. All the models produced in this thesis can be converted into online tools. It is important for models to be made readily available and accessible so that there is more potential for use by any interested party. It is, therefore, a recommendation that AIS-based modelling of noise exposure be made into an online tool to allow for uploading of AIS data to be transformed into visual map outputs of ocean vessel noise.

Java is a widely-used programming language. Java programmes are capable of being made into software or online applications. The AIS-based model devised in Chapter 3 provided a method for modelling cumulative vessel noise exposure through Java programming with this in mind. The model is open to the possibility of becoming an online tool that can be used by anyone with an interest in ocean vessel noise monitoring, whether they be researchers, mitigation decision-makers, or general public. There could even be potential to create a website with real-time vessel noise exposure monitoring using AIS data as soon as it is made available.

Another area of future research would be to combine the AIS-based model with models of other noise sources such as pile-driving (Rossington *et al.*, 2013) to produce a map of all cumulative noise in the oceans. Pile driving noise can influence species as far as 1.3 km away from the source (Roberts *et al.*, 2016). Seismic air-gun arrays have caused significant behavioural reactions in fish up to 30 km away from the source (Slotte *et al.*, 2004). Atlantic salmon have been seen to react to wind farm operational noise 1 km away from the source (Thomsen *et al.*, 2006), and wind farms off the coast of the UK can cover as much as 122 km^2.The model created in Chapter 3 focuses only on vessel noise exposure, and ignores other anthropogenic noise sources which may be adding further noise in the same location. When combining biological information such as species distributions with the AIS-based noise exposure model, although it provides useful information in terms of vessel noise mitigation, it does not provide an entire picture without all other anthropogenic noise sources being included.

6.4 Concluding remarks

This thesis addressed the key questions of which species and habitats should be prioritised first, where should research, mitigation efforts and legislation be focused, and how can researchers overcome challenges in noise mapping and apply the models to direct mitigation. A Prioritisation Index was created to aid in species prioritisation in terms of noise pollution. This index is not limited to noise pollution, however, and has applications for other threats and species too. In terms of vessel noise pollution, AIS-based modelling can help address the issue of where research and mitigation efforts should be directed. Mapping ocean noise can identify noise hotspots and areas where noise levels are likely to be above harmful levels, and manipulation of historical data can provide prediction of future scenarios so that mitigation can be pre-emptive as well as reactive. In order that ocean noise maps can be trusted, the methods involved in creating the map must be solid and provide accurate results. Challenges of vessel noise mapping have included lack of availability of large datasets and inaccuracies in the source level estimation of the vessels. Both of these challenges have been overcome in this thesis. The yearlong dataset covering the entirety of the UK has proved that large scale ocean noise modelling is possible, and that it does not take exhaustive computational power. Combining the AIS modelling with individual ship source level equations has added a new dimension to vessel noise mapping, so that source levels are no longer estimated using vessel density or one sound level for every vessel of a certain type.

The models and methods from this thesis will contribute to the growing field of underwater marine noise modelling and help direct future research and mitigation to priority areas.

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8. Appendix 1

8.1 Literature Review meta-analysis data

Table A1.1. Details of experiments discovered during the literature review. Information has been added to each column where possible. When there is no mention of the data in the article 'Not stated' was used. Extra information extracted that seemed relevant was added in parenthesis. In the 'Field / Lab' column, (c) means that a study used controlled (fish has restricted range or movement) or caged conditions, and (w) means the study was done in open water with no restrictions on the fish. For durations seconds is represented by 's', minutes by 'min' and hours by 'hrs'. For sample size, the number in parentheses specifies that the sample size was for the control condition; no parentheses implies the sample size is for the experimental condition. For articles presenting more than one experiment, multiple data are separated by a semi-colon. *articles are taken from conference proceedings from the 3rd International Conference on the Effects of Noise on Aquatic Life 2013. **articles are taken from conference proceedings from the 4th International Conference on the Effects of Noise on Aquatic Life 2016.

Author	Year	Species	Noise type (specifically)	Field/ lab	Distance from source (m)	Noise level	Frequency (Hz)	Signal depth	Duration of signal	Acoustic metric	Sample size	Conclusions
Abbott & Bing-Sawyer (taken from review)	2002	Sacramento blackfish	Pile driving (no details given)	Field (c)	45-850	> 193 dB, < 183 dB	Not stated	Not stated	43 pile strikes with air bubble curtain followed by 45 strikes without curtain	SPL	Not stated	• Damage found in fish subjected to >193 dB • More damage closer to source
Abbott *et al.*	2005	Chinook salmon, anchovy, perch	Pile driving (0·61 m, jetted concrete piles, diesel-assisted hammer)	Field (c)	9.75 (cage depth 7.62 m)	Not stated	Not stated	Not stated	200 noise pulses over 4 min	Not stated	Not stated	• No significant impacts on physiology
Amoser & Ladich	2003	goldfish, catfish	White noise (unfiltered)	Lab	Within bucket	158 dB	200-4,000	On bottom of bucket (tank bottom 15 cm)	12 or 24 hr	SPL	12 goldfish, 19 catfish	• The auditory sensitivity of both species diminished. • *C. auratus* recovered within three days • *P. pictus* recovered within 14 days after exposure in all but one frequency. • Hearing specialists are affected differently by noise exposure • Acoustic communication might be restricted in noisy habitats
Amoser *et al.*	2004	carp, perch, whitefish, roach	Boating (powerboat, top speed 270 km/h)	Lab	> 300 (recording mounted 1 m above test tank)	103-128 dB	100-4,000	Hydrophone recorded at 1.5 m (bottom depth 140 m)	21 tone burst per s	SPL	6 carp, 7 perch, 6 whitefish, 2 roach	• Fish can detect noise up to 400 m • Fish were disturbed by race
Anderson *et al.*	2011	Lined seahorse	Aquaria	Lab	Within tank	78-148	100-2,000	Not stated	Not stated	SPL	11	• Fish in noisy tank made more adjustments • No differences in vocalisations • Those exposed to noise declined in weight and condition • More heterophils (an immune system response) were evident in noisy tanks • Cortisol creations were greater in noisy tanks
Andersson *et al.*	2007	sticklebacks, roach	Wind turbines (monopole steel foundation, wind speed 14 $m{\cdot}s^{-1}$, 1780 rpm)	Lab	Within tank (recorded at 83 m)	80-120 dB	25-500; 25-1,000	Hydrophone recorded at 12.9 m (tank water depth 15 cm)	Sequence of 10 s with 2 minute gap	SPL	60 (10) sticklebacks, 45 roach	• Both species responded to noise • Roach showed swimming bursts of 20-40 cm • Observed: twitching, backing, vertical movement and freezing
Andersson *et al.**	2012	Silver eel	Wind farm (0.0019 m/s^2, 2.3 MW/turbine)	Field (w)	< 11,000	126-142 dB at 1 m, 81-96 dB at 1 km	2-200	From turbine	Continuous while passing through on migration	SL	264	• Eels did not shift their migration route • Did not alter swimming behaviour • Hypothesised that ship noise was masking the presence of the windfarm
Banner & Hyatt (taken from review)	1973	Sheepshead minnow, Longnose killifish	White noise (broadband noise)	Lab	Not stated	15 dB above ambient environment	100-1,000	Not stated	Not stated	Not stated	Not stated	• Minnow egg viability significantly reduced • Growth rates of fry for both species significantly less than those held at 20 dB quieter

Author	Year	Species	Noise type (specifically)	Field/ lab	Distance from source (m)	Noise level	Frequency (Hz)	Signal depth	Duration of signal	Acoustic metric	Sample size	Conclusions
Berthe & Lecchini	2016	Eagle ray	White noise & Boat (25 hp Yamaha engine, passing hydrophone at 10 m)	Lab	1	90 -120 dB re 1 µPa	40, 600, 1,000	4 m	5 min	SPL	50 (9)	• White noise had no effect on 90 % of rays • Boat noise significantly disturbed rays; 60 % showed escape behaviour, moving at least 50 m away
Blaxter & Hoss	1981	herring	White noise (no details given)	Lab	0.35-0.85	Not stated	10-1,000	35 cm above surface	0.5 s pulse	Sound Pressure	10-30 per experiment	• Fish length significantly influenced hearing threshold • Behavioural results were only observed between 70 – 200 Hz
Blaxter *et al.*	1981	herring	White noise (no details given)	Lab	0.17-0.24	Not stated	80 or 92	35 cm (water depth 75 cm)	Not stated	Sound Pressure	370; 75; 1,000 fish observed in total	• Startle response depended on number of fish • Slight increase in noise caused accelerated swimming in forward direction • Stronger noise caused abrupt turning and acceleration away from source
Bolle *et al.***	2016	Common sole, herring, European sea bass	Pile driving (4 m diameter steel monopole at 20 m)	Lab	Not stated (recording at 100 m)	188-210 dB	50-1,000	Not stated	Not stated	SPL, SEL	Per experiment: 25 sole; 30 sea bass. No. of herring unclear	• No immediate effect of noise exposure observed • In one trial mortality in the control was lower but not significantly.
Bracciali *et al.*	2012	damselfish	Boating (no details given)	Field	Throughout field site	Not stated	Not stated	Surface (mean bottom depth 15 m)	4 x 15 min sessions 3 times a day	Not stated	Not stated	• Increased boating caused decrease foraging • Low intensity boating caused increased foraging
Bruintjes & Radford	2013	cichlids	Boating (Bristol harbour)	Lab	Not stated	127 dB re 1 µPa	Not stated	Not stated (tank depth 27 cm)	2 boat passes per min (~18 s per pass) for 15 min	SPL	78	• Noise negatively affects nest digging and defence • Individuals are impacted differently depending on context • Sex specific responses observed
Bruintjes & Radford	2014	cichlids	Boat noise (playback)	Lab	Not stated (recordings taken 10-50 m from ship)	127 dB re 1 µPa	20-20,000	Not stated	1 h recordings played randomly over 13 h for one month	RL	415 eggs, 191 fry, 237 eggs, 109 fry	• No direct negative impacts on hatching success or survival • No effects on weight, length and survival
Bruintjes *et al.*	2016	European eels, European sea bass	Boat (5 m aluminium 30 hp 2 stroke outboard motor, < 10 knots, 100-400 m from hydrophone); Pile-driving (127 m from 1.2 m monopole 25 m into seabed at 6 m depth)	Lab / Field (c)	0.1; Within tank; 0.1; 1	144.4 – 148.06 dB re 1 µPa $_{rms;}$ 139.3 – 141.2 dB re 1 µPa $_{rms;}$ 165.5-167.3 dB re 1 µPa; 200.1-201.5 dB re 1 µPa	50-5,000	On bottom of tank; On bottom of tank; On bottom of tank; 0.8 m	2 min	RL, SEL	156; 195; 36; 36	• Decrease in antipredator responses in eels during noise treatments • Eels showed rapid recovery of startle responses and startle latency (42.9 % less likely to startle during noise exposure) • Rapid (but incomplete) recovery in ventilation rate in eels 2 min after exposure • After 2 min full recovery was observed • Sea bass ventilation rate was significantly impacted by noise exposure
Buscaino *et al.*	2010	European sea bass, Gilthead sea bream	White noise	Lab	5	150 dB re 1 µPa	100-1,000	1.5 m	10 min	SPL	14 sea bass, 14 sea bream	• Demonstrated a disturbance effect from noise exposure on glucose, lactate and haematocrit levels in sea bream and sea bass. • Increased swim speed in noise condition
Caltrans	2004	Shiner perch, Rainbow trout	Pile driving (steel, 500 & 1,700 kj hammer)	Field (c)	23-314	215-220 dB	Not stated	Not stated	1-20 min	SPL	Unknown	• More trauma in fish exposed to noise • 204 dB resulted in serious barotrauma injury • No barotrauma injury observed further than 440 m away or with less than 180 dB
Caltrans	2010	Steelhead salmon	Pile driving (2.2 m, cast in steel shell, diesel impact hammer	Field (c)	35 – 150 (control at 350 m)	179-194 dB, control 132-141 dB	Not stated	(bottom depth 1.3 m)	2 hrs 6 min – 4 hrs 30 min (1,100 – 3,396 strikes)	SPL, SEL	50 (10)	• No statistically significant difference • Suggests noise level must be higher than 194 dB to have adverse effects • No fish died in any cage • No significant internal or external injuries • Plasma cortisol and haematocrit levels were significantly different (considered due to fish handling)

Author	Year	Species	Noise type (specifically)	Field/ lab	Distance from source (m)	Noise level	Frequency (Hz)	Signal depth	Duration of signal	Acoustic metric	Sample size	Conclusions
Casper *et al.*	2012	Chinook Salmon	Pile driving (76.2 cm steel shell pipe, diesel hammer.	Lab	Not stated (recording taken at 10 m)	217 dB, 210 dB	Not stated	Not stated	960 strikes	SEL	175 (53)	• Fish evaluated on day 0 showed a wide range of injuries e.g. bruised organs, haemorrhaging of tissues • The higher dB level resulted in more injuries • Recovery was occurring by day 5
Casper *et al.*	2013	Striped bass, tilapia	Pile driving (playback of 76.2 cm steel shell pile with diesel hammer)	Lab	10	210-216 dB	Not stated	Not stated	960 strikes	SEL	123 (32) bass, 14 (14) tilapia	• All bass and 1 tilapia had significant numbers of damaged hair cells at 216 dB • No cell damage was observed at 210 dB • Suggests pile driving has more impact on swim bladders than inner ears
Celi *et al.*	2016	Gilthead sea bream	Boat (7 recreational boats, hydrofoil, fishing boat and ferry)	Lab	Within tank (recordings made at 30-50 m from source)	113.9-141.2 dB re 1 µPa	44-22,720	Not stated	10 days	SPL	40	• No significant differences in biometric and plasma parameters • No significant difference in weight and fork length • Significantly increased cortisol, glucose, lactate, haematocrit, Hsp70, cholesterol, triglycerides, osmolality and Adrenocorticotropic hormone.
Codarin *et al.*	2009	Red Mouthed goby, Mediterranean damselfish, Brown meagre	Boating (cabin cruiser, 8.5 m, 163 HP inboard diesel engine, max speed 6 knots)	Lab	0.5 (above surface)	132 dB (continuous) 138 dB (peak)	300-10,000	Hydrophone recorded at 10 m (bottom depth 18 m)	60 s	SPL, SEL	6 goby, 6 damselfish, 6 meagre	• Auditory thresholds increased by up to 10 dB, 20 dB & 35 dB respectively
Cott *et al.**	2012	chub, pike, whitefish	Air gun (730 in^3 array, see Popper *et al.* 2005)	Field (c)	2 – 15	205-209 dB	Not stated	Not stated	5 or 20 shots	SPL	Not stated	• No acute associated mortality • Stunning occurred in those 2 m from source • Temporary Threshold Shift (TTS) occurred in chub, with recovery after 18 hrs • TTS occurred in pike, recovery was after 24 hrs (but no TTS in juveniles) • Whitefish showed no TTS • No evidence of herding, swimming or startle behaviours • No damage to the inner ear
Davidson *et al.*	2009	Rainbow trout	Aquaculture	Lab	Not stated	149 dB re 1 µPa	Not stated	Not stated	24 hours a day over 5 months	SPL	200	• Initial alarm reaction, scattering through tank and swimming erratically. • Normal swimming behaviour returned within a few hours • Fish in noise treatment were smaller than the control but not significantly • No significant difference in body condition, feed conversion rates or survival. • No long-term effect of exposure reported
De Robertis *et al.*	2008	Walleye pollock	Boating (NOAA Oscar Dyson noise reduced vessel; NOAA Miller Freeman; 11 knots)	Field (w)	Passed within 10 m of buoy	Not stated	<2,000	From boat on surface (bottom depth 60 – 700 m)	2 boats took turns passing buoy at 15 min intervals	Not stated	Not stated	• In shallower water there was significant difference in avoidance behaviour at night but not during the day. • No differences in deep water (400 – 700 m) • Deepening of fish layer in response to the Miller Freeman vessel
Debusschere *et al.*	2014	Sea bass	Pile driving (2.5 m depth monopole, hydraulic hammer)	Field (c) / Lab	43, 500	181-188 dB re 1 µPa^2 s $_{ss}$, 210 dB re 1 µPa $_{z\text{-}p}$, 215 up to 222 dB re 1 µPa^2 s $_{cum}$	125-200 (peak)	Not stated	1,739 – 3,067 strikes over 1.5 hrs	SEL	528 (264)	• Smaller juveniles had high mortality rate • No significant increase in mortality during first 14 days after exposure • Significant differences observed for whole-body lactate levels • Clear reduction in oxygen consumption (49-55 % reduction) • No significant differences in growth and condition between treatments • No skeletal abnormalities observed

Author	Year	Species	Noise type (specifically)	Field/ lab	Distance from source (m)	Noise level	Frequency (Hz)	Signal depth	Duration of signal	Acoustic metric	Sample size	Conclusions
Doksaeter *et al.*	2009	herring	Sonar	Field (w)	<400	127-197 dB, 139-209 dB	1,000-2,000, 67,000	35 m	20 s pulses of 1 s signals	SEL	Not stated	• No obvious difference • No escape reactions seen • There was an effect observed when a boat with towed gear navigated past the experiment. Response lasted a few min
Doksaeter *et al.**	2012	herring	Sonar (1,000 – 1,600 Hz hyperbolic FM upsweep; 1,000 weighted continuous wave signal)	Field (c)	500 – 1,609	168 dB (maximum)	1,000-1,600	Near surface	Gradual from 1 mile, sudden at 500 m	RL	Not stated	• Minor startle responses in 3 of 14 'sudden' exposures • Engine noise caused strong schooling, increased school density and rapid downward movement (this behaviour was stronger in winter than in summer)
Engas *et al.*	1996	Cod, haddock	Air gun (3 x 6 gun array, 13,784 kPa, 2,000 psi, 82,132 cm^3 total volume)	Field (w)	< 50,000	248.7 dB (maximum)	10-150	Towed at 6 m (bottom depth 250 – 280 m)	Every 10 s or 25 m	SPL	33,000 tonnes cod, 6,000 tonnes haddock	• Acoustic density reduced from 129.8 to 72.0 after air gun blasts • Cod catches in shooting area decreased by 69 % • Catch did not increase again in the 5 days after shooting • Haddock catches in shooting area decreased by 68 % • Longline catches reduced by 45 % • Larger fish disappeared quicker
Engas *et al.*	1995	Atlantic cod, Atlantic herring	Boat (factory trawler, 56.9 m 3,000 hp, recorded at 45-200 m)	Field (c)	11 (at closest point)	90 dB re 1 μPa	600-3,000	2 m	Four 130 s clips with 20 min gap	SPL	34 cod, 18 herring	• Cod and herring reacted in 73 % and 75 % of experiments • Avoidance observed • No alarm reactions • Cod increased group cohesion and moved down the water column • Herring school became denser and schooled in a diagonal path
Enger (taken from review)	1981	cod	Not stated	Not stated	Not stated	180 dB	50-400	Not stated	Several hours	Not stated	Not stated	• Ciliary bundles on sensory cells of ears destroyed
Everley *et al.**	2013	Sea bass	Pile driving	Lab	Not stated	160.5 dB	0-2,000	Not stated (tank 35 cm deep)	Not stated	RL	18, 18 (c)	• Antipredator behaviour was impaired • Did not startle to predator presence when noise was playing
Filiciotto *et al.*	2013	Gilthead sea bream	Aquaculture	Lab	Not stated	112 – 146 dB re 1 μPa	25-1,000	Not stated	4,500 noise files on loop for 120 days	SPL	400	• Observed higher levels of serum cortisol, glucose, red blood cell counts, haematocrit values and haemoglobin content and lower levels of white blood cells in fish exposed to onshore aquaculture system noise compared with noise from offshore aquaculture systems. • Offshore fish showed higher growth • Sea soundscape positively influence growth performance and reduced stress
Fisher-Pool *et al.**	2012	Hawaiian damselfish	Boating (twin 205 hp, 8 cylinder diesel engine, only 1 engine used)	Field	10 m	Not stated	100-1,000	From boat on surface	30 min	Not stated	Not stated	• Calling rates did not differ
Gerlotto *et al.*	2004	anchovy, sardine	Boat noise (43.6 m trawler, 3.5 knots)	Field (c)	<40	Not stated	Not stated	Near surface	Continuous whilst boat passes	Not stated	Not stated	• No evidence of vertical avoidance • No evidence of horizontal avoidance • Diving of 5 m observed by suggested avoidance of the hull not the noise
Govoni *et al.*	2003	pinfish, spot	Explosions	Field (c)	3.6, 7.5, 17.0	2.0 – 8.7 Pa	Not stated	Not stated	12.8 min	Sound pressure	Not stated	• Many fish dead, stunned or terminal • Internal haemorrhaging was evident in 9 fish at 3.6 m and 1 pinfish at 17 m • Haematuria found in exposed fish kidneys • Pancreas ruptured in exposed fish

Author	Year	Species	Noise type (specifically)	Field/ lab	Distance from source (m)	Noise level	Frequency (Hz)	Signal depth	Duration of signal	Acoustic metric	Sample size	Conclusions
Graham & Cooke	2008	Largemouth bass	Recreational boating (9.9 hp operating in neutral, electric, Combustion engine, trolling motor, canoe paddle)	Lab	1.5	Not stated	Not stated	On surface of tank	60 s	Not stated	9	• All treatments resulted in increased cardiac output in all fish – increased heart rate and decreased stroke volume • Canoe had the shortest recovery time (~15 min), combustion engine had the longest (~40 min)
Halvorsen *et al.*	2011	Chinook salmon	Pile driving (steel shell, diesel hammer)	Lab	<0.45 (Recorded 10 m from pile)	215 dB	0-1,200	Not stated	960 or 1,920 strikes over 24 or 48 min	SEL	356, 140 (c)	• As energy level of pile driving increases, damage increases • High energy levels caused substantial physiological costs • Less severe exposures caused mild injuries • Only mild injuries could be recovered from • Those without a swim bladder were at much lower risk to barotrauma than those with a swim bladder
Halvorsen *et al.*	2012	Chinook salmon	Pile driving (steel shell, diesel hammer)	Lab	<0.45	204 – 220 dB re 1 μPa^2 s	0-1,200	Not stated	960 or 1,920 strikes over 24 or 48 min	SEL	356 (140)	• Barotrauma injuries ranged from mild to mortal • Severity of fish injury was a function of the energy in SELss, SELcum, and the number of impulsive sounds. • Higher energy exposures caused mortal injuries such as organ haemorrhages
Halvorsen *et al.*	2012	Lake sturgeon, tilapia, hogchocker	Pile driving (76.2 cm steel shell pile, diesel hammer)	Lab	<0.45 (Recorded 10 m from pile)	204 – 216 dB re 1 μPa^2 s cum	Not stated	Not stated	960 strikes	SEL	125 (32) tilapia, 57 (10) hogchoker: 141 (32) sturgeon	• Presence and type of swim bladder correlated with injury at higher sound levels • No external injuries in any fish • No internal injuries in the hogchoker • Tilapia showed moderate to mortal internal injuries • Sturgeon showed mild to moderate internal injuries
Handegard *et al.***	2016	herring	Echosounder & multibeam sonar	Lab	11	155-175 dB at source, 131-147 dB at receiver	160, 320, 500	5 m	10 tones of 2 s & 2 sweeps of 5 s	SPL & RL	Not stated	• Weak/non-existent overall reaction • Stronger response to louder sources and when in smaller group
Hastings & Miksis-Olds*	2012	Pinecone soldierfish, Sabre squirrelfish, Bluestripe snapper, Green damselfish	Air gun (array, 2,055 in^3)	Field (c)	45 – 2,743 (cage at 5 m depth)	190 dB	1,000-2,400	Not stated (water depth 24 – 45 m)	Single or double pass over 5 days (22 – 50 ms pulse tones)	SEL	20 soldierfish, 10 squirrelfish, 47 snapper, 51 damselfish	• No temporary threshold shift observed
Hastings *et al.*	1996	oscar	White noise (pure tones)	Lab	3.8	100, 140, 180 dB	60-300	Not stated	Continuous or every 3 s	SPL	59 (5)	• 4 of 5 fish were damaged after 1 hrs of continuous noise • No fish were damaged in intermitted noise condition • No damage was observed after 1 day (it took 4 days for damage to show)
Hawkins *et al.*	2014	sprat, mackerel	Pile driving	Field (w)	5	185 dB re 1 µPa at 1 m p-p, 171 dB re 1 µPa p-p	50-600	Near surface	10 strikes with 2 second gap	SPL, SEL	321 (222 were subject to noise)	• Sprat were more likely to disperse after noise • Mackerel were more likely to change depth after noise • Responses observed at 163.2 & 163.3 dB re 1 µPa p-p, 135 & 142 dB re 1 μPa^2 s • There was a sudden dent in the zooplankton layer at start of noise. This did not persist. Occurred from 155.8 dB re 1 µPa p-p, • Lateral dispersal observed with schools recombining at a greater depth • Rapid depth changes observed

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Herbert-Read *et al.*	2017	European sea bass	Pile driving (1.2 m diameter monopole driven ~25 m into seabed at 6.5 m depth)	Lab	Recordings made 77 m and 200 m from source	Not stated	Not stated	8 cm	5 min loop of 10-30 s playbacks with 1.7 s pile driving rate	Not stated	60 ambient, 60 pile driving treatment	• Significant shoal dispersion observed • Position in shoal changed during playback • Angular difference in heading was significantly different in noise treatment • Reductions in speed observed in presence of noise • Change in coordinated movement • Significant impact on social interactions
Holles *et al.*	2013	Sabre-spined Cardinal-fish	Boat noise (playback)	Field (c)	10	77.71 peak ambient noise	200-3,000	5 m	Five 1 m replicates	SPL	Up to 16 per treatment	• 69 % of fish swam towards Reef playback • 56 % during Reef+Boat playback • 8 % of fish larvae moved away from Reef playback • 44 % of fish larvae moved away from Reef+Boat playback
Holmes *et al.*	2017	Ambon damselfish	Boat (5 m aluminium dinghy, 30 hp 2-stroke engine, max speed ~10 knots)	Field (c)	30-80	Not stated	100-1,000 peak	Not stated (2-3 m depth)	20 min	Not stated	Not stated	• There was an immediate decrease in the boldness and relative distance moved • Behaviour of newly-settled fish was significantly affected by the operation of a small motor boat nearby. • Noise from small boats operating within 30–80 m of the fish affected the activity and boldness of individuals. However, after 20 min of continuous exposure to boat noise, behaviour returned to pre-exposure levels, suggesting fish become desensitised.
Holt & Johnston	2014	Blacktail shiner	White noise	Lab	0.2	10.2-16.93 dB above control level	80	15 cm	17 min – 2.5 hours	Not stated	19	• Lombard effect can occur • Duration of calls decreased in noisy conditions • Rate of call increased in noisy conditions • Intensity of call increased in noisy conditions
Houghton *et al.*	2010	Coho salmon	Pile driving (steel sheet, 20 in, 3 x small impact hammer (BSP SL-60), 8 x larger J&M 115 hammer)	Field (c)	1-50 (1 – 2.5 m cage depth)	170-195 dB peak, 145-189 dB exposure	Not stated	Not stated	354 – 2,781 strikes over 13 – 51 min	SPL, SEL	16	• No mortalities or tissue damage reported up to 48 hrs after exposure • Only applicable to negatively buoyant fish
Iafrate *et al.***	2016	Sheepshead and Gray snapper	Pile driving (polymetric fibreglass reinforced concrete piles at 10 m)	Field (w)	10-370	182 dB at source, 136-158 dB at receiver	Not stated	Not stated	Over 82 days	SPL	Sheepshead: 15, 12 (c) Gray: 10, 3 (c)	• Presence of snapper decreased • No injury or mortality observed
Johansson *et al.***	2016	perch, roach	Boat noise (aluminium hull, 70 hp 4 stroke outboard motor run at 2,000 rpm)	Field (c)	10-13	147 dB	150-5,000	Near surface	30 min	SPL	Perch: 15 Roach: 15	• Perch had significantly higher cortisol secretion rates • More cortisol measured in short term exposure trials
Jorgensen *et al.*	2004	capelin	Boating (64.4 m 910 BRT; 47.5 m 493 BRT, 11 knots,	Field (w)	5-1,000	90-150 dB	10-10,000	3 – 10 m (Bottom depth 53 – 167 m)	Not stated	SL	Not stated	• No significant influence in spawning or feeding ground
Jorgensen *et al.**	2012	zebrafish	White noise (Gaussian filtered by a 1st order Biquad filter)	Lab	No stated (speaker outside tank)	130-150 dB	550-1,450	Not stated	Not stated	SPL	50, 50(c)	• ABR showed threshold shift from 95 dB (control) to 105 dB (exposed) • No behavioural differences when exposed to predators
Jung & Swearer	2011	11 reef species (larvae)	Boating (powerboat, 28 knots, 6 m rigid hull inflatable, 70 hp 2-stroke engine)	Field (w)	30 (bottom depth 8 m)	107-111 dB	8,000	Bottom depth at hydrophone was 4 m	14 s	RL	Not stated	• No avoidance reaction

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Kastelein *et al.*	2008	Sea bass, mullet, pout, cod, eel, pollock, mackerel, herring	White noise (pure tones)	Lab	Within tank (4 m long tank)	Not stated	100-64,000	Not stated	30 s	SPL	Sea bass: 17 Mullet: 11 Pout: 9 Cod:5 Eel: 10 Pollock: 4 Mackerel: 13 Herring: 10	• Increased swim speed • Often made tight turns • Startle responses observed
Kastelein *et al.*	2017	European sea bass	Pile-driving (4.2 m pile, recorded 800 m from source)	Lab	3.5	166 dB re 1 µPa	200-20,000	On bottom of tank (2 m depth)	920 strikes over 20 min	SPL, SEL, PVL	68	• Changes in swim speed and direction • Tighter school cohesion observed on onset of noise • Changes in body posture and startle responses observed • Small fish responded to quieter sounds (~ 10 dB quieter) that larger fish • Small fish school cohesion significantly different from large fish school cohesion.
Klimley & Beavers	1998	rockfish	ATOC signal (playback)	Field (c)	< 15	109.5-153 dB	75 (with harmonics of 150 & 225)	~1 m (bottom depth up to 2 m)	25 min	SPL	11	• ATOC had no effect on fish • Fish stayed in high noise zones • No behavioural difference between silent and noise conditions
Konagaya (English summary)	1980	Not stated	Construction noise	Not stated	160	90 dB	500-600	Not stated	continuous	SPL	Not stated	• Not stated
Krebs *et al.***	2016	sturgeon	Pile driving (2.4 & 3 m, impact hammer)	Field (w)	< 500	187 dB continuous, 206 dB peak, 150 dB at receiver	Not stated	Not stated	10 min events over ~ 4 weeks	RL, SPL SEL	155	• Unlikely the sturgeon will be close enough to the pile for long enough to be impacted
Luczkovich *et al.**	2012	Atlantic croaker	Boating (large tug boats & ferryboat)	Field (w)	Not stated	Not stated	200-8,000	Not stated	Continuous boat traffic for 8 months	Not stated	Not stated	• Fish vocalisations were less common when large vessels with low mid-frequencies passed by the recorder • Vessel noise had limited effect on vocal production
Luczkovich *et al.***	2016	Oyster toadfish	Boat noise	Field (w)	Not stated	Not stated	20-20,000	7.5 m above floor	Not stated	SPL	25-50 % occupancy of 96 dens	• Call rates occurred in both quiet and noisy treatments • More calls occur in noisy site • Lombard observed • Call power increase by up to 19 dB • No change in frequency
Magnhagen *et al.*	2017	Eurasian perch, roach	Boat (aluminium hull, 5 m long, 70 hp, 4-stroke outboard motor, 2,000 rpm)	Field (c)	8	126 dB	50-2,500	On surface	30 min a day for 4 days	SPL	36	• Feeding performance of both species was negatively affected
McCauley *et al.*	2003	Pink snapper	Air gun (0.33 L Bolt PAR 600 B, 10 MPa, 330 cc, 20 in^{3}, single gun)	Field (c)	5-800	222.6 dB re 1 mPa at 1 m	20-1,000	5 m (bottom depth 9 m)	4 approaches over 65 min, 72 min break, then 3 approaches over 36 min (6 pulses per minute)	SL	Not stated	• Causes severe damage to sensory hair cells of the saccule of the ear • Damage increased for at least 58 days post exposure • 18 hours after exposure there were localised dense patches of holes on the epithelia • Number of holes were significantly greater after 58 days
Meier & Horseman	1977	tilapia	White noise	Lab	Within tank (tank: 76 x 38 x 30 cm high)	40 dB	Not stated	Not stated (tank depth 20 cm)	20 min per day for 25 days	Not stated	105	• Influenced fat stores, growth and several reproductive indices • Fish subject to the daily stimulus gained weight • Greatest disturbance occurred when sound played at dawn • Increase in growth depended on time of treatment and diminished in treatment groups
Miller *et al.*	2016	flounder	Pile driving	Model	Not stated	Not stated	Not stated	Not stated	Not stated	Not stated	Not stated	• Conservative criterion proposed for swim bladder fish – effects are limited to 250 m from the pile driving for 960 strikes • More strikes will likely cause fish to leave the area

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Mitson & Knudsen	2003	herring	Boating	Field (w)	Directly above	Up to 165 dB	18,000, 120,000, 200,000	Not stated	Continuous	SL	Not stated	• Fish avoidance occurs • Reduced noise vessels may help
Nedelec *et al.*	2015	Atlantic cod	Boat	Lab	0.1	Not stated	Not stated	15 cm	Six 15-minute ship recordings over 6 hours (over 24 hours a day for 16 days); Two 15 minute recordings with a 2 minute break; 3 minute recording	Not stated	7,000 eggs per tank; 32; 40	• Reduced growth observed • Noise affected larval cod behaviour, growth and development. • Short-term exposure caused startle responses in newly hatched larvae. • Two days of additional noise of both regular and random regimes reduced growth • Larvae that had a lower body width–length ratio were easier to catch in a predator-avoidance experiment • Regular noise was more disturbing than random noise. • Larvae exposed to regular noise used their yolk sacs faster after 2 days of exposure and had a lower body width–length ratio after 16 days post hatch compared with those raised in ambient or random noise.
Nedelec *et al.*	2016	Three spot damselfish	Boat (5 m long aluminium outboard motorboats with 25 horse power Suzuki engines)	Field (c)	1	Not stated	Not stated	Not stated	45 s every 5 min for 12 hours a day (for 14 days)	Not stated	288	• Significantly more hiding observed in presence of boat noise (0.83 more fish hid in noise condition) • Short-term exposure significantly increases opercular beat rate. This significantly lessens in long-term exposure. • No significant effect on growth or body condition • No significant effect observed in cortisol concentration • Short term exposure was worse than long term exposure • Found evidence for behavioural and physiological attenuation
Nedelec *et al.*	2017	Bluestreak cleaner wrasse	Boat (25 hp outboard motor, 4 – 10 knots)	Field	10-100	80-150 dB re 1 µPa	1,000-30,000	At surface	20 min	SPL	24	• No significant difference in composition of clientele at cleaner station • Significantly more time was taken inspecting clients during noise treatment • Normal behaviour resumed when noise stopped • Clients jolted significantly more to contact during noise treatment, and often after, but this did not impact overall cleaning.
Nedelec *et al.*	2017	reef fish	Boat	Field	Not stated	128 +- 2 dB re 1 µPa	0-2,000	Not stated	6 disturbances per hour 12 hrs a day	SPL	38 nests	• Significant effect of noise on defensive brood-guarding • Males in noise treatment spent 25 % less time feeding • Offspring glancing was not significantly impacted • Complete brood mortality was more likely in noise treatment • No significant impact on juvenile growth • Noise did not affect the growth of developing offspring • Noise did reduced the likelihood of offspring survival • There was a significant effect of sound treatment on defensive acts made by brood-guarding • Boat treatment males made on average twice as many defensive acts (chasing/making aggressive strikes at other fish) compared to males exposed to ambient-sound playback • Males at Boat nests also spent 25 % less time feeding
Nedwell *et al.*	2003	Brown trout	Pile driving (PVE2316 VM driver; BSP357/9 hydraulic drop hammer, 200 pa)	Field (c)	25-400	~134-194 dB	Not stated	Source noise measured at 2.5 m	10 piles driven,	SPL	Not stated	• No damage observed in fish 400 m from source • No reaction to vibropiling for fish as close as 25 m
Nedwell *et al.*	2006	Brown trout	Pile driving (4 x 914 mm, 6 x 508 mm	Field (c)	25 – 400 (recorded at 2.5 m depth)	189-198 dB	Not stated	Source noise measured at 2.5 m	Up to 200 min	SL	Not stated	• No negative effects • No increase in activity or startle response was seen to vibropiling • Published report of above paper

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Neo *et al.*	2014	European sea bass	White noise	Lab	3.5	134-172 dB	250-1,600	On bottom of tank	Continuous, 1 s pulses, alternate 1 & 9 s pulses, alternate 1 – 17 s pulses	SPL	48	• Strong initial response • Recovery to normal swimming occurred quicker in continuous condition • Diving observed within first 5 min of exposure • Group cohesion increased with individuals moving closer together • Increased swimming speed after exposure
Neo *et al.*	2015	zebrafish	White noise	Lab	<1	112 dB	500-1,500	In air	Continuous, 1 s pulses, alternate 1 & 9 s pulses, alternate 1 – 17 s pulses	SPL	200	• Exposure to moderate noise levels (112 dB re 1 µPa) can affect the swimming behaviour of fish by changing group cohesion, swimming speed and swimming height • Effects were brief for both continuous and intermittent noise treatments • Initial startle responses • Diving response • No long term spatial avoidance or noise-dependent tank choice
Neo *et al.*	2016	European sea bass	Filtered brown noise	Lab	7.8 (minimum)	169-169 dB re 1 µPa for continuous, 180-192 dB re 1 µPa for intermittent	200-1,000	2.2 m	Continuous or 0.1 s pulses for 60 min	SPL, SVL	64	• Swim speed, depth and distance from the speaker significantly increased on initial exposure • Group cohesion was stronger in regular impulsive treatment • Within 30 min of exposure fish resumed normal behaviour
Neo *et al.*	2015	European sea bass	Filtered brown noise	Lab	2.4-7	157 dB re 1 µPa $_{z\text{-}p,}$ 170-179	600	On bottom of tank	Continuous or 0.1 s pulses for 60 min	SPL, SEL	48	• When exposed to noise fish dived significantly deeper in tighter shoals and swam significantly faster • Observed startle response • Fish habituated within an hour
Nichols *et al.*	2015	Giant kelpfish	Boat (175 hp outboard Yamaha engine)	Lab	4, 6, 8, 10, 15, 20	126.1 – 141.9 dB re 1 µPa	Peak between 50 & 400	Not stated	60 min of 5-12 s boat sounds with a 1 – 120 s gap	SPL	72 (assuming 4 treatments were used)	• Acute stress responses to intermittent noise but not continuous • Random intermittent noise exhibited significantly higher cortisol concentrations • Regular intermittent noise showed higher but not significantly higher levels of cortisol than control
Oestman & Earle*	2012	Steelhead trout	Pile driving (2.2 m, cast-in-steel shell, Pileco D225 diesel impact hammer)	Field (c)	35 – 150 (control at 350 m)	178-194 dB	Not stated	From pile	20 – 24 min sessions	SPL, SEL	159, 156 (c)	• No significant damage observed during histopathology or necropsy • Damage was observed but believed to be natural levels and not caused by noise impacts
Ona & Godo	1990	haddock	Boating (60-m combined purse seiner/trawler, 3 knots, 460 rpm, 1,000 hp)	Field (w)	5-2,000	Not stated	Not stated	(bottom depth 500 m)	Not stated	Not stated	Not stated	• Strong, downward avoidance reactions • At depths greater than 100 m the reaction pattern was weak and irregular • Fish avoidance occurs between the surface and 200 m depth • A slight reaction was seen even in the pre-vessel zone, and a substantial diving reaction occurred just after propeller passage • Original pattern of distribution was re-established by the time the vessel was 400 m away, after 4 or 5 min.
Pearson	1992	rockfish	Air gun (diesel driven compressor)	Field (c)	12	137-205 dB	Not stated	Gun at 6 m (bottom depth 14 m)	6 pulses per min for 10 min	SPL	54	• Change in swimming behaviour at 154-168 dB • Alarm behaviour exhibited at 178-207 dB • Startle responses seen between 200-205 dB • Vertical movement and freezing observed • Behaviour affected from 161 dB • Alarm response threshold 180 dB

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Picciulin *et al.*	2010	Red Mouthed goby, damselfish	Boating (26 m tourist ferry, inboard diesel engine, 6 knots; 5 m fibreglass boat, 40 HP outboard engine, 15 knots)	Field (c)	0.5 (3-6 m depth)	162, 142 dB	1,033; 602	Hydrophone recorded at 4 m (bottom depth 8 m)	5 min	SPL	Goby: 20 Damselfish: 20	• No significant swimming behaviours observed • Time spent in shelter increased • Decrease in time spent caring for nests
Picciulin *et al.*	2012	Brown meagre	Boat (8.5-m CC with a 163-hp inboard diesel engine operating at 6 kn; a 5-m FB with a 40-hp outboard engine moving at 15 kn; and a 7-m INF with a 130-hp in-outboard engine operating at 20 kn)	Field (w)	Not stated	135 (INF), 134 (FB), and 133 (CC) dB re 1 µPa with a maximum instantaneous SPL of 150 (INF), 150 (FB), and 144 (CC) dB re 1 µPa	160 (INF), 123 (FB), 130 (CC)	Near surface	6 boat passes, 10 min apart	SPL	Not stated	• Mean pulse rate increased over multiple boat passages in the experimental condition but not in the control condition
Popper *et al.*	2005	Broad whitefish, Northern pike, Lake chub	Air gun (8 SGI & SGII type sleeve guns, 730 in^3, 12,000 cc, 2.6 m x 1.22 m, 19,000 psi)	Field (c)	13 – 17 (recorded 1 m below surface)	205-210 dB	2-10,000	1.8 m and pointed towards fish (water depth 1.9 m)	5 or 20 shots	SPL	Not stated	• No effect on broad whitefish • 10-25 dB TTS in pike and chub
Popper *et al.*	2007	Rainbow trout	Sonar (LFA)	Field (c)	7.9	193 dB	170-320	23.8 m (bottom depth 140.2 m)	108 s of signal, 9 min of silence, repeated 3 times	SPL, SEL	Not stated	• No mortality • Behavioural effects observed • Rapid burst of swimming after onset.
Popper *et al.***	2016	Pallid sturgeon, paddlefish	Seismic air gun (4 guns totalling 10,060 cm^3)	Field (c)	0 – 150	206-231 dB at source, 199-225 dB at cages, 187-205 dB exposure	Not stated	3 m	Not stated	SPL, RL, SEL	Not stated	• No mortality recorded • No significant tissue damage
Poulton *et al.*	2017	European Sea bass	Pile driving	Lab	Within tank (recording at 120 m from source)	161.3 dB	Not stated	Not stated	5 min (21 strikes, pulse length = 100 ms)	SPL	96	• There was a significant difference in the change in ventilation rate after the track change between noise treatments (two-way repeated measures • 67 % more fish startling when exposed to 1,000 µatm/pile-driving noise compared with 400 µatm/ambient noise • There was no interaction effect between elevated CO_2 and pile-driving noise on anti-predator behaviour or ventilation rate.
Purser *et al.*	2016	European eel	Boat (5 m aluminium 30 hp 2 stroke outboard motor, < 10 knots, 100-400 m from hydrophone)	Lab	Not stated	148 dB dB re 1 µPa $_{rms}$	Not stated	2 min	Not stated	SPL	130	• Opercular beat rate increase in noise treatment, and was higher in eels with bad body condition compared to those with good body condition • Startle responses reduce in noise treatment, correlated again to body condition
Purser & Radford	2011	Three spined sticklebacks	White noise (playback comparable to peak SPLs recorded at the shoreline of lakes where recreational speedboats are active)	Lab	Within tank	>48, <150 dB	100-1,000	Not stated	10 or 300 s	SPL	24	• Startle responses observed in noise conditions • Foraging errors increase in both noise conditions • Food vs non-food errors increased • Negative impact on foraging activity

Author	Year	Species	Noise type (specifically)	Field/ lab	Distance from source (m)	Noise level	Frequency (Hz)	Signal depth	Duration of signal	Acoustic metric	Sample size	Conclusions
Radford *et al.*	2016	European sea bass	Pile-driving (127 m from 1.2 m monopole 25 m into seabed at 6 m depth), seismic surveys (4,450 in^3), boat (5 m aluminium 30 hp 2 stroke outboard motor, < 10 knots, 100-400 m from hydrophone)	Lab	0.3	156 dB dB re 1 μPa at 1 m, 130-140 dB re 1 μPa	<1,000	Not stated	5 min; Boat: 4 hours, airgun: 4 hours, pile-driving: 16 hours. For 12 weeks	SPL	90 in initial experiment, 270 from post 12 week experiment; 1,350 from long term study	• Lessened response after repeated exposure, likely due to increased tolerance or change in hearing threshold • Habituated after 12 weeks • Noise significantly impacted opercular beat rate • Beat rate significantly varied between noise types
Ramcharitar & Popper	2004	Black drum, Atlantic croaker	White noise (pure tones)	Lab	0.35	124.3, 136.2	100-1,500	Bottom of bucket (40cm depth)	10 ms x 5	SPL	Drum: 15 Croaker: 15	• At 124 dB both species changed auditory sensitivity due to masking
Roberts *et al.***	2016	sprat, mackerel	Pile-driving (recording)	Field (c)	1287.5	103.9-148.6 dB received, 127.9-171.3 dB over 20 s period, 164-173 dB peak-peak 5 m from recorder	50-600	0.5 m	20 s (10 strikes)	SEL	236	• Dispersal, density changes and depth changes observed
Ruggerone *et al.*	2008	Coho salmon	Pile driving (hollow steel pipe piles, 0.51 m diameter, 1.3 cm wall thickness, 17.7-18.3 m long, open-end diesel impact hammer)	Field (c)	1.8-15 (cage 30 cm below surface)	207 dB	23-442	(water depth 4.3-5.2 m)	1,627 strikes over 4.3 hours	SPL	700	• No adverse effects observed • Behavioural responses subtle and not consistent • Startle responses occurred near to pile • No habituation was observed • Startle responses to crew walking past cage • Only applicable to negatively buoyant fish • Visual stimuli caused greater startle effect
Sabet *et al.*	2015	zebrafish, water flea	White noise	Lab	1.5 (Speaker outside tank)	122 dB re 1 μPa	Band passed between 300 – 1,500	Level with bottom of tank	9 min per treatment	SPL, PVL	14 zebrafish, 100 water flea	• Significant increase in swim speed in noise condition • No significant change in swim depth • Intermittent sounds delayed acceleration response to prey and increased handling errors • Intermittent sound cause more behavioural change
Sara *et al.*	2007	Bluefin tuna	Boating (hydrofoil ferries, two 2,000 hp engines; small boats, outboard 75-100 hp motors; large car ferry)	Field (caged)	200-800	100-135 dB	70-20,000; 4,000-6,000	Bottom depth 30 m	>20 min	SPL	~100	• Behavioural response observed • Increased vertical movements, abandoning middle layer • Dispersion of school • Restless, speed changes and turning behaviour
Scholik & Yan	2001	Fathead minnow	White noise	Lab	1	142 dB	300-4,000	1 m above surface (tank depth 5.5 cm)	1-24 hours	SPL	24	• Intense white noise significantly elevated auditory threshold • Effects were dependent on duration of exposure • Longer duration had more impact • Recovery seen within 6 days if exposed for 2 hours, but no recovery seen even after 14 days to those exposed for 24 hours
Scholik & Yan	2002	Bluegill sunfish	White noise (no details given)	Lab	1	142 dB	300-2,000	1 m above surface (tank depth 5.5 cm)	2-24 hours	SPL	24	• No significant evaluation in auditory threshold • Showed species differ when comparing results with Scholik & Yan's (2001) paper • Sunfish threshold is affected but minimally
Sebastianutto *et al.*	2011	Red mouthed goby	Boating (field recorded diesel engine, 5 m fibreglass, 40 hp, 15 knots)	Lab	0.7	161 dB exposure, 165.9 dB peak	0-800	Within 60 cm deep tank	25 s looped for entire territorial display	SPL, SEL	5	• Noise was detectable between 70 – 400 Hz • Resident fish was less successful at defence in noisy conditions • Boat noise had negative impact

Author	Year	Species	Noise type (specifically)	Field/ lab	Distance from source (m)	Noise level	Frequency (Hz)	Signal depth	Duration of signal	Acoustic metric	Sample size	Conclusions
Simpson *et al.*	2015	European eel	Boat (5 m aluminium 30 hp 2 stroke outboard motor, < 10 knots, 100-400 m from hydrophone)	Lab	Not stated	148 dB re 1 µPa $_{rms}$	Not stated	Not stated (recorded at 1 m)	2 min; For 10 turns made by the eel	SPL	48; 24; 48; 48	• Antipredator performance was impacted (50 % less likely to startle in noise condition) • Eels in noise condition were caught twice as quickly • Observed changes in spatial behaviour and physiological state • Significant increase in opercular beat rate • Antipredator performance of juvenile eels affected in experimental ambush and pursuit predation paradigms • Changes in spatial behaviour • Changes in physiological state • Decrease in lateralisation
Simpson *et al.*	2005	Saddle red clownfish	White noise (no details given)	Lab	1 (above dish)	80-150 dB	100-1,200	Above petri dish	20 ms tone bursts for 72 s	SL	Not stated	• Heart rates significantly increased • Noticeable 3 to 5 days post fertilisation
Simpson *et al.*	2016	damselfish	Boat noise (5 m aluminium 30 hp 2 stroke outboard motor)	Field (c) / Lab	Not stated	Not stated	0-3,000	Not stated	72 hours; 30 min; Not stated; 5 min; 15 min	Power Spectral Density	80; 58; 36; 30; 36; 220	• Significant negative effect on survival (only 27 % survived in the boat condition; 79 % survived control) • Fish used 20 % more oxygen in noise treatment than in control in lab • Fish used 33 % more oxygen in noise treatment than in control in field • Significant mortality due to predation in noise condition
Simpson *et al.***	2016	reef fish	Boat noise (5 m aluminium 30 hp 2 stroke outboard motor)	Field (w)	10-100	136.1 dB	Not stated	1-4 m	Continual overnight	RL	Not stated	• Reduced settlement of young fish in presence of boat noise • 61 % of fish chose the quieter reef.
Skalski *et al.*	1992	rockfish	Air gun (diesel driven air compressor used; 33.5 m industrial vessel	Field (w)	Not stated	180-200 dB	Up to 1,500	6.1 m	Single blast	SPL	Not stated	• 52 % decline in catch rate • Decline in catch rate significant in 3 of 5 rockfish species • Alarm behaviour observed
Slotte *et al.*	2004	herring, Blue whiting	Air gun (two 20 guns, 2,000 psi, 3,090 in^3)	Field (w)	Up to 30,000	Not stated	Not stated	8 m (bottom depth ~200 – 2,600 m)	Every 25 m along 51 525 m long transects	Not stated	301,000 tonnes whiting, 155,000 tonnes herring	• No short term scaring effects observed • During shooting fish moved to greater depths • Fish density was lowest in shooting area and highest 20 nautical miles from shooting area
Smith.	2004	goldfish	White noise	Lab	Source within 76/19 litre tank	160-170 dB	100-10,000	On bottom of tank	10 min-21 days	SPL	42	• Spike in cortisol secretion • Loss of hearing • Shift in threshold after just 10 min of exposure • Shifts increased linearly up to 28 dB after 24 hrs of exposure • After 21 days of exposure, it required 14 days for recovery
Smith *et al.*	2004	tilapia, goldfish	White noise	Lab	Within bucket	130-170 dB	0.1-10,000	Not stated	1-28 days	SPL	Not stated	• Tilapia showed little or no hearing loss even after exposure of 28 days • Goldfish showed considerable threshold shifts of up to 25 dB
Smith *et al.*	2006	goldfish	White noise	Lab	Within tank	170 dB re 1 µPa $_{rms}$	200-2,000	Not stated	48 hours	SPL	36	• Significant loss of hair bundles • Hearing had significantly recovered after 7 days • Developed scar formations and intact cuticular plates with missing hair bundles were occasionally observed

Author	Year	Species	Noise type (specifically)	Field/ lab	Distance from source (m)	Noise level	Frequency (Hz)	Signal depth	Duration of signal	Acoustic metric	Sample size	Conclusions
Spiga *et al.**	2012 b	Red drum, Sea trout	Boat noise (sport fishing vessel)	Lab	Not stated	120, 175 dB	Not stated	Not stated	Continuous ambient, continuous boat, intermittent (30 min periodically) For 8 weeks	SPL	Not stated	• Fish reacted to initial playback by swimming tight circles and accelerating • This was no longer observed after 1 week of exposure • Protein to energy ratio declined in noise exposed fish • Higher feed conversion ratio values were shown by fish subjected to continuous noise in red drum; but opposite for sea trout • No influence on growth or survival
Spiga *et al.**	2012a	Red drum, Sea trout	Boat noise (sport fishing vessel)	Lab	Not stated	180 dB; 175 dB	Not stated	Not stated	0, 15, 30, 60 min; Continuous ambient, continuous boat, intermittent (30 min periodically) For 8 weeks	SPL	100	Short term study: • In the 30 min condition there were significantly elevated plasma cortisol levels • Recovery after 60 min Long term study: • No significant results • Observed that after 30 min whole body cortisol level had increased, but reduced again after 60 min of exposure • Behavioural changes at the onset of noise, including startling and swimming in tight circles with increased acceleration. This response diminished after 1 week.
Suzuki *et al.*	1980	anchovy	Boating (fishing vessel, diesel engine)	Lab	Within tank (min 0.4)	120-140 dB	Not stated	3 m (bottom depth 45 cm)	Not stated	SPL	Not stated	• Vessel noise causes avoidance • 130 to 140 dB can frighten fish and make them dive down several metres just after the noise is produced • This level of noise might be experienced within 400 m of a large tanker or within 10 m of a small purse-seiner • Within 30 m there might be large behaviour changes • Fish moved away from speaker when turned on
Suzuki**	2016	Plainfin midshipman	Boat noise (14 ft aluminium boat, 9.9 hp engine)	Field (w)	1-30	Not stated	Not stated	Near surface	16 min	Not stated	15 nests	• Indicated possible change in visitation patterns by certain species of predators • Significant avoidance observed for predators • Overlap of noise and calls apparent at 75-800 Hz • No significant difference in nest care, but reactions were observed
Thomsen *et al.**	2012	cod, sole	Pile driving (playback)	Field (c)	Not stated	140-161, 144-156 dB	Not stated	Not stated (water depth 10 – 15 m)	Not stated	SPL		• Sole significantly increased swim speed during playback • Cod showed freezing response
Voellmy *et al.*	2014a	Three-spined stickleback, European minnow	Boat noise (playback)	Lab	Within tank (recordings taken at 100-400 m)	5 dB under original ship noise	<5,000	Recorded at 1 m	5 min	SEL	Stickleback: 30 Minnow: 28	• Showed startle responses • Consumed less food • Stickleback maintained foraging effort but made more mistakes • Minnows reduced foraging effort and became more social
Voellmy *et al.*	2014 b	Three-spined stickleback, European minnow	Boat noise (playback)	Lab	Within tank (recordings taken at 100-400 m)	5 dB under original ship noise	100-3,000	On bottom of tank (recorded at 1 m)	1 min minimum	SEL	Stickleback: 35 Minnow: 27	• Sticklebacks were significantly more likely than minnows to respond to a visual predatory stimulus
Wardle *et al.*	2001	pollack, cod, saithe	Air gun (150 in^3, 3 guns, 56 kg, 60 cm x 29 cm, 2,000 psi, firing < 1 ms)	Field (w)	Up to 109	206 dB	Up to 250 (mean of 100.1)	5 m or seabed at 14 m (max bottom depth 20 m)	295 shots over 4 days	SPL	5 pollock, 5 cod, 5 saithe	• Shots did not cause fish to leave area • Shots did cause startle responses • Involuntary sideways movement when shot fired
Watwood *et al.***	2016	Sheepshead & Gray snapper	Sonar	Field (w)	Not stated	Not stated	Not stated	Not stated	Not stated	Not stated	Sheepshead: 25 Gray: 28	• No mortality recorded • Short-term declines in residency observed in the area

Author	Year	Species	Noise type (specifically)	Field/ lab	Distance from source (m)	Noise level	Frequency (Hz)	Signal depth	Duration of signal	Acoustic metric	Sample size	Conclusions
Wilson *et al.*	2008	Allis shad	Ultrasound	Lab	Not stated	157, 161, 167, 173, 179, 185 dB	70,000, 120,000	Not stated	12 stimulations, 0.4 or 0.7 s pulse with a 5 min gap	RL	6 groups of 2 – 5 fish	• Intensity graded response between 161 & 167 dB re 1 µPa for both frequencies • Change in swimming behaviours observed • Change did not persist when noise stopped • Significant correlation between RL and swimming speed
Wysocki & Ladich	2005	goldfish, Raphael catfish, Pumpkinseed sunfish	White noise (pure tone bursts)	Lab	1	110-130 dB	100-4,000	1 m above surface	21 /s, 2 cycles to 8 cycles	SPL	6 goldfish, 6 catfish, 7 sunfish	• Masking caused increased hearing thresholds
Wysocki *et al.*	2006	European perch, Common carp, Common gudgeon	White noise (generated by a noise generator), Boating (boat passage recorded in Danube River and the lakes Mondsee and Traunsee)	Lab	Within bucket	156, 153 dB	Up to 5,000	Bottom of bucket (water depth 12 cm)	30 min	SPL	6 carp, 7 goby, 7 perch	• Boat noise causes cortisol secretion • White noise had no effect
Wysocki *et al.*	2007	Rainbow trout	Aquaculture noise (9.1 m x 2.4 m round fiberglass aquaculture tank within a recirculating system)	Lab	Not stated	115, 130, 150 dB	2-20,000	Not stated (water depth 0.8 m)	For first 9 months of life	SPL	2,100	• No stress response observed • No TTS • Hearing sensitivity, growth, survival, stress, and disease susceptibility were not negatively impacted by noise levels common to recirculating aquaculture systems
Harding *et al.*	2016	Atlantic salmon	Pile-driving	Lab	2	122.17-164.33 dB	1-1,000	0.5 m; 0.07 m	4 hours	SPL, PAL	40; 26	• No significant difference in oxygen consumption or activity between treatments • No mortality or injury observed • Study suggests that additional piling noise did not cause any of the differences between treatment • No clear evidence of startle responses • Salmon used in the study did not perceive pile driving noise as a stressor

8.2 Audiogram data retrieved from previous research

The following diagrams (Figure A1.1 and Figure A1.2) depict the hearing thresholds found in all known audiograms studies. 77 species in total have been investigated. The hearing thresholds have been shown on a scale with the anthropogenic noise sources present in the marine environment. Both the full hearing range and the optimal frequencies for the species are shown.

Key:

Best hearing range

Total hearing range

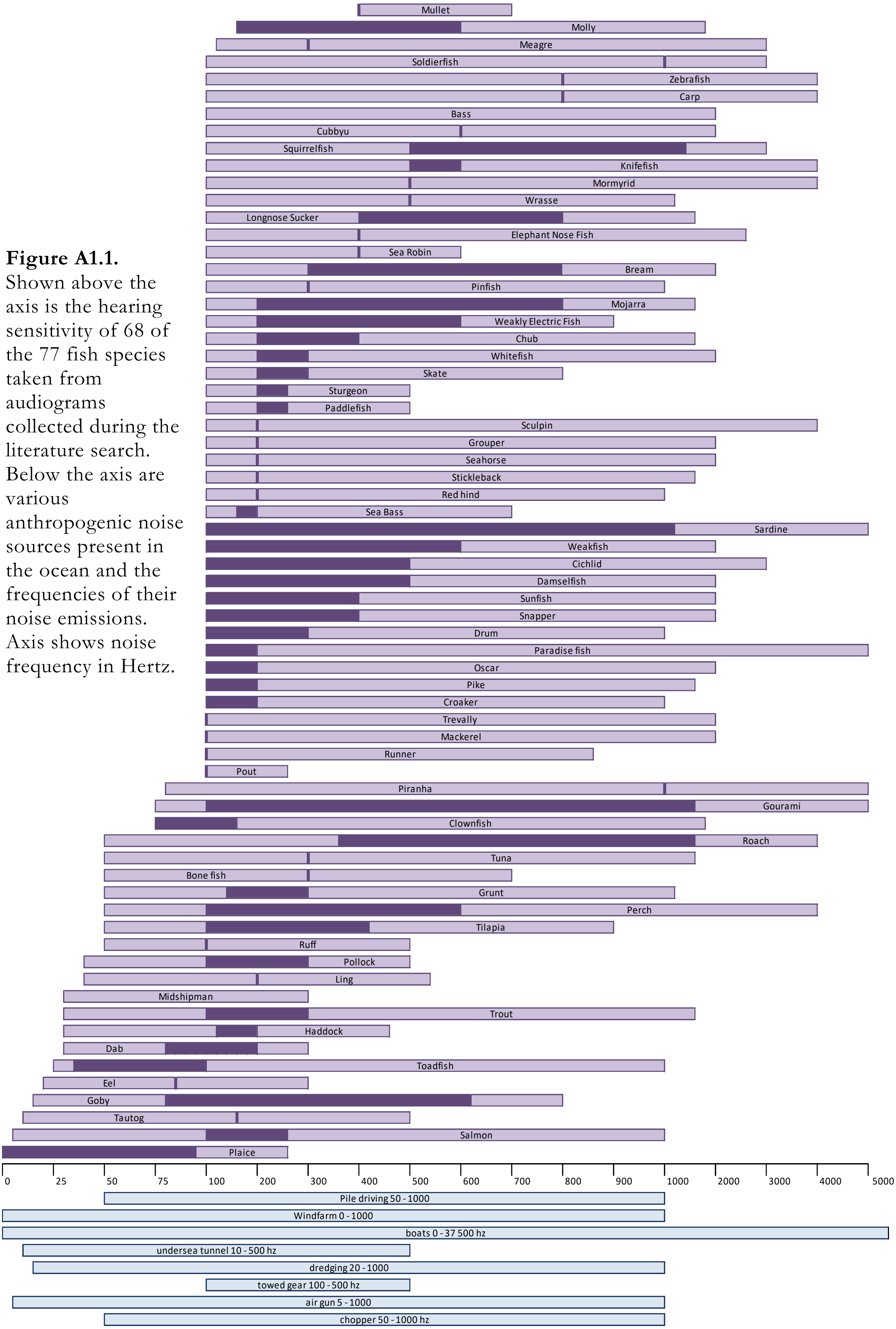


Figure A1.1. Shown above the axis is the hearing sensitivity of 68 of the 77 fish species taken from audiograms collected during the literature search. Below the axis are various anthropogenic noise sources present in the ocean and the frequencies of their noise emissions. Axis shows noise frequency in Hertz.

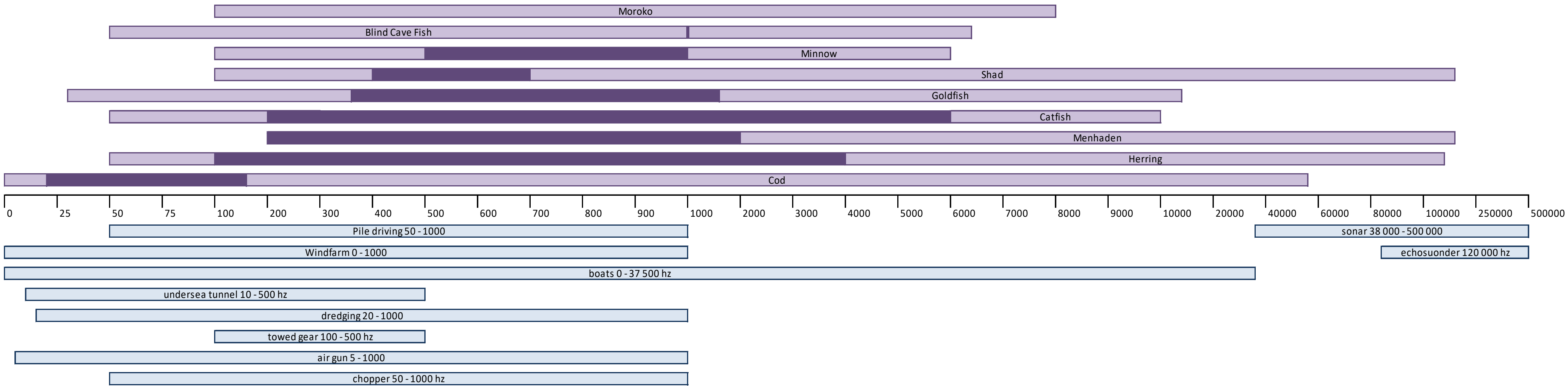


Figure A1.2. Shown above the axis is the hearing sensitivity of the remaining 9 of the 77 fish species taken from audiograms collected during the literature search. Below the axis are various anthropogenic noise sources present in the ocean and the frequencies of their noise emissions. Axis shows noise frequency in Hertz (Hz).

9. Appendix 2

9.1 Reason for linear scale in Population Condition category

A. Population size reduction. Population reduction (measured over the longer of 10 years or 3 generations) based on any of A1 to A4

	Critically Endangered	Endangered	Vulnerable
A1	≥ 90%	≥ 70%	≥ 50%
A2, A3 & A4	≥ 80%	≥ 50%	≥ 30%

A1 Population reduction observed, estimated, inferred, or suspected in the past where the causes of the reduction are clearly reversible AND understood AND have ceased.

A2 Population reduction observed, estimated, inferred, or suspected in the past where the causes of reduction may not have ceased OR may not be understood OR may not be reversible.

A3 Population reduction projected, inferred or suspected to be met in the future (up to a maximum of 100 years) *[(a) cannot be used for A3]*.

A4 An observed, estimated, inferred, projected or suspected population reduction where the time period must include both the past and the future (up to a max. of 100 years in future), and where the causes of reduction may not have ceased OR may not be understood OR may not be reversible.

based on any of the following:

(a) direct observation *[except A3]*
(b) an index of abundance appropriate to the taxon
(c) a decline in area of occupancy (AOO), extent of occurrence (EOO) and/or habitat quality
(d) actual or potential levels of exploitation
(e) effects of introduced taxa, hybridization, pathogens, pollutants, competitors or parasites.

B. Geographic range in the form of either B1 (extent of occurrence) AND/OR B2 (area of occupancy)

	Critically Endangered	Endangered	Vulnerable
B1. Extent of occurrence (EOO)	< 100 km²	< 5,000 km²	< 20,000 km²
B2. Area of occupancy (AOO)	< 10 km²	< 500 km²	< 2,000 km²
AND at least 2 of the following 3 conditions:			
(a) Severely fragmented **OR** Number of locations	= 1	≤ 5	≤ 10

(b) Continuing decline observed, estimated, inferred or projected in any of: **(i)** extent of occurrence; **(ii)** area of occupancy; **(iii)** area, extent and/or quality of habitat; **(iv)** number of locations or subpopulations; **(v)** number of mature individuals

(c) Extreme fluctuations in any of: **(i)** extent of occurrence; **(ii)** area of occupancy; **(iii)** number of locations or subpopulations; **(iv)** number of mature individuals

C. Small population size and decline

	Critically Endangered	Endangered	Vulnerable
Number of mature individuals	< 250	< 2,500	< 10,000
AND at least one of C1 or C2			
C1. An observed, estimated or projected continuing decline of at least (up to a max. of 100 years in future):	25% in 3 years or 1 generation (whichever is longer)	20% in 5 years or 2 generations (whichever is longer)	10% in 10 years or 3 generations (whichever is longer)
C2. An observed, estimated, projected or inferred continuing decline AND at least 1 of the following 3 conditions:			
(a) (i) Number of mature individuals in each subpopulation	≤ 50	≤ 250	≤ 1,000
(a) (ii) % of mature individuals in one subpopulation =	90–100%	95–100%	100%
(b) Extreme fluctuations in the number of mature individuals			

D. Very small or restricted population

	Critically Endangered	Endangered	Vulnerable
D. Number of mature individuals	< 50	< 250	D1. < 1,000
D2. *Only applies to the VU category* Restricted area of occupancy or number of locations with a plausible future threat that could drive the taxon to CR or EX in a very short time.	-	-	D2. typically: AOO < 20 km² or number of locations ≤ 5

E. Quantitative Analysis

	Critically Endangered	Endangered	Vulnerable
Indicating the probability of extinction in the wild to be:	≥ 50% in 10 years or 3 generations, whichever is longer (100 years max.)	≥ 20% in 20 years or 5 generations, whichever is longer (100 years max.)	≥ 10% in 100 years

Figure A2.1. Summary of the five criteria used to evaluate if a taxon belongs in an IUCN Red List threatened category. For Least concern and Near Threatened there is no specific criteria given. Image taken from: http://s3.amazonaws.com/iucnredlist-newcms/staging/public/attachments/3110/2001catscrit_summary_en.pdf.

9.2 Further detail on ecosystem service scores

Species Names	Scientific Name	Goods/Benefits												Ecosystem services provision score
		from Provisioning services				from Regulating services				from Cultural services				
		Fish feed (wild, farmed, bait)	Fertiliser and biofuels	Ornaments and aquaria	Medicines and blue biotechnology	Healthy climate	Prevention of coastal erosion	Sea defence	Waste burial / removal / neutralisation	Tourism and nature watching	Spiritual and cultural well-being	Aesthetic benefits	Education	
Albacore tuna	*Thunnus alalunga*		1			1	1	1	1				2	7
Angel shark	*Squatina squatina*					1	1	1	1					4
Atlantic Bluefin tuna	*Thunnus thynnus*		1			1	1	1	1				2	7
Atlantic cod	*Gadus morhua*		1			1	1	1	1		3		3	11
Atlantic halibut	*Hippoglossus hippoglossus*		1			1	1	1	1					5
Basking shark	*Cetorhinus maximus*		1	1	1	1	1	1	1	4	4	4	4	23
Blue skate	*Dipturus batis*				1	1	1	1	1			4		9
Bottlenose skate	*Rostroraja alba*				1	1	1	1	1			4		9
European eel	*Anguilla anguilla*		1	2		1	1	1	1	2	2	1	2	14
Flounder	*Platichthys flesus*					1	1	1	1					4
Haddock	*Melanogrammus aeglefinus*		1			1	1	1	1					5
ling	*Molva dypterygia*		1	1	1	1	1	1	1	2	3	3	3	18
Mackerel	*Scomber scombrus*	4	1			1	1	1	1	4				13
Meagre	*Sciaena umbra*					1	1	1	1					4
Nursehound	*Scyliorhinus stellaris*					1	1	1	1			1		5
Sandy ray	*Leucoraja naevus*		1			1	1	1	1	2	2			9
Seabream	*Sparus aurata*					1	1	1	1					4
seahorse	*Hippocampus hippocampus*		1	3	2	1	1	1		3	4		3	19
Shortfin mako	*Isurus oxyrinchus*					1	1	1	1			4		8
Smalleyed ray	*Raja microocellata*					1	1	1	1					4
Smooth hammerhead	*Sphyrna zygaena*					1	1	1	1	2		4	2	12
Smoothhound	*Mustelus mustelus*					1	1	1	1					4
Sunfish	*Mola mola*					1	1	1	1					4
Thornback skate	*Raja clavata*		1			1	1	1	1	2	2			9
Undulate ray	*Raja undulata*		1			1	1	1	1				2	7
Whithound	*Galeorhinus galeus*					1	1	1	1					4
Yellowfin tuna	*Thunnus albacares*		1			1	1	1	1				2	7

Scale of ecosystem service supplied reative to other features	
4	Significant contribution
3	Moderate contribution
2	Low contribution
1	negligible contribution
0	No contribution

Confidence in evidence
UK-related, peer-reviewed iterature
Grey or overseas literature
Expert opinion or Obvious
Not assessed

Figure A2.2. Ecosystem services provision scoring based on the method by Potts *et al.* (2014).

9.3 Supplementary material for Prioritisation Index

Table A2.1. Full species list used to demonstrate the Prioritisation Index (PI). Species with population condition scores of 1 or below were deemed low priority and removed from the case study to keep the number of species manageable, allowing easier demonstration of the Index. There is no limit for the number of species that can be included in the PI.

Species	Population condition score
Herring	0
Skipjack tuna	0
Atlantic Pollock	1
Bullet tuna	1
Catfish	1
Dab	1
Frigate tuna	1
Garfish	1
Goby	1
Hake	1
Lamprey	1
Mullet	1
Plaice	1
Pout	1
Runner	1
Salmon	1
Sand eel	1
Sardine/anchovy	1
Seabass	1
Sole	1
Solenette	1
Sprat	1
Stickleback	1
Whiting	1
Wrasse	1
Flounder	2
Mackerel	2
Nursehound	2
Albacore tuna	3
Atlantic cod	3
Haddock	3
Ling	3
Smalleyed ray	3
Thornback skate	3
Yellowfin tuna	3
Atlantic halibut	4
Basking shark	4
Meagre	4
Sandy ray	4
Seabream	4
Seahorse	4
Shortfin mako	4
Smooth hammerhead	4
Smoothhound	4
Sunfish	4
Whithound	4
Atlantic Bluefin tuna	5
Bottlenose skate	5
Undulate Ray	5
Angel shark	6
Blue skate	6
European eel	6

10. Appendix 3

10.1 Vessel noise map technical build information

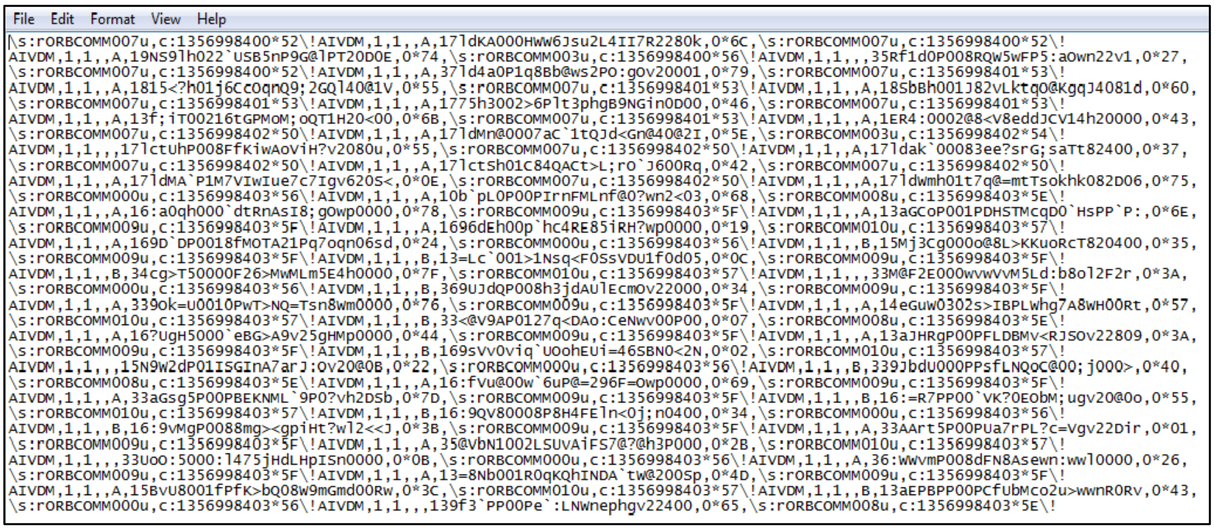
The software application built in Chapter 3 is written in the Java programming language using Eclipse IDE Mars.1(4.5.1), with a PostgreSQL object-relational database management system, PostGIS (an extension for spatial and geographic objects for PostgreSQL), and web development languages (HTML5, CSS, Javascript and PHP).

All the software selected for the model development are freely available and allow extensive, reusable code to be written and run anywhere. Java programs are now considered on par with the speed and performance of other languages such as C++, and is considered good for high performance computing.

PostgreSQL was chosen above other more popular database management systems such as MySQL because it is the most advanced system and allows geographic extensions like PostGIS to be used for geographical data.

The web development languages were chosen to make the tool easily accessible online allowing more users and a greater reach of audience.

10.2 Vessel noise map AIS data



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AIVDM,1,1,,A,169D`DP0018fMOTA21Pq7oqn06sd,0*24,\s:rORBCOMM000u,c:1356998403*56\!AIVDM,1,1,,B,15Mj3Cg000o@8L>KKuoRcT820400,0*35,
\s:rORBCOMM009u,c:1356998403*5F\!AIVDM,1,1,,B,13=Lc`001>1Nsq<F0SsVDU1f0d05,0*0C,\s:rORBCOMM009u,c:1356998403*5F\!
AIVDM,1,1,,B,34cg>T50000F26>MwMLm5E4h0000,0*7F,\s:rORBCOMM010u,c:1356998403*57\!AIVDM,1,1,,,33M@F2E000wvwVvM5Ld:b8ol2F2r,0*3A,
\s:rORBCOMM000u,c:1356998403*56\!AIVDM,1,1,,B,369UJdQP008h3jdAUlEcmOv22000,0*34,\s:rORBCOMM009u,c:1356998403*5F\!
AIVDM,1,1,,A,339ok=U0010PwT>NQ=Tsn8Wm0000,0*76,\s:rORBCOMM009u,c:1356998403*5F\!AIVDM,1,1,,A,14eGuW0302s>IBPLWhg7A8WH00Rt,0*57,
\s:rORBCOMM010u,c:1356998403*57\!AIVDM,1,1,,B,33<@V9AP0127q<DAo:CeNwv00P00,0*07,\s:rORBCOMM008u,c:1356998403*5E\!
AIVDM,1,1,,A,16?UgH5000`eBG>A9v25gHMp0000,0*44,\s:rORBCOMM009u,c:1356998403*5F\!AIVDM,1,1,,A,13aJHRgP00PFLDBMv<RJSOv22809,0*3A,
\s:rORBCOMM009u,c:1356998403*5F\!AIVDM,1,1,,B,169sVv0viq`UOohEUi=46SBN0<2N,0*02,\s:rORBCOMM010u,c:1356998403*57\!
AIVDM,1,1,,,15N9W2dP01ISGInA7arJ:Ov20@0B,0*22,\s:rORBCOMM000u,c:1356998403*56\!AIVDM,1,1,,B,339JbdU000PPsfLNQoC@00;j000>,0*40,
\s:rORBCOMM008u,c:1356998403*5E\!AIVDM,1,1,,A,16:fVu@00w`6uP@=296F=Owp0000,0*69,\s:rORBCOMM009u,c:1356998403*5F\!
AIVDM,1,1,,A,33aGsg5P00PBEKNML`9P0?vh2DSb,0*7D,\s:rORBCOMM009u,c:1356998403*5F\!AIVDM,1,1,,B,16:=R7PP00`VK?0EObM;ugv20@0o,0*55,
\s:rORBCOMM010u,c:1356998403*57\!AIVDM,1,1,,B,16:9QV80008P8H4FEln<0j;n0400,0*34,\s:rORBCOMM000u,c:1356998403*56\!
AIVDM,1,1,,B,16:9vMgP0088mg><gpiHt?wl2<<J,0*3B,\s:rORBCOMM009u,c:1356998403*5F\!AIVDM,1,1,,A,33AArt5P00PUa7rPL?c=Vgv22Dir,0*01,
\s:rORBCOMM009u,c:1356998403*5F\!AIVDM,1,1,,A,35@VbN1002LSUvAiFS7@?@h3P000,0*2B,\s:rORBCOMM010u,c:1356998403*57\!
AIVDM,1,1,,,33Uoo:5000:l475jHdLHpISn0000,0*0B,\s:rORBCOMM000u,c:1356998403*56\!AIVDM,1,1,,A,36:WWvmP008dFN8Asewn:wwl0000,0*26,
\s:rORBCOMM009u,c:1356998403*5F\!AIVDM,1,1,,A,13=8Nb001R0qKQhINDA`tW@200Sp,0*4D,\s:rORBCOMM009u,c:1356998403*5F\!
AIVDM,1,1,,A,15BvU8001fPfK>bQ08W9mGmd00Rw,0*3C,\s:rORBCOMM010u,c:1356998403*57\!AIVDM,1,1,,B,13aEPBPP00PCfUbMco2u>wwnR0Rv,0*43,
\s:rORBCOMM000u,c:1356998403*56\!AIVDM,1,1,,,139f3`PP00Pe`:LNWnephgv22400,0*65,\s:rORBCOMM008u,c:1356998403*5E\!
```


Figure A3.1. Raw AIS data from Orbcomm for position messages. Each AIS transmission starts with “\S:”, the timestamp starts at “c:”, and the position data starts at “\!AIVDM”. The timestamp is decoded using specially written Java code, and the position data is decoded based on the codes in Table A3.1.

```
$ head -10 apr_alltypes.txt
\g:1-2-8891,s:rORBCOMM008,c:1364774400*5f\!AIVDM,2,1,5,A,5;rt8jh00003SSG00033SSG
0000000000000000E0000040000000000000000,0*60
\g:2-2-8891*55\!AIVDM,2,2,5,A,00000000008,2*29
\g:1-2-8923,s:rORBCOMM010,c:1364774400*5e\!AIVDM,2,1,3,B,56:DA1000000000000001`V0`
TpN0`UF0P4v22220t2qCA350Tj9j0BH0C`?;K,0*7C
\g:2-2-8923*5D\!AIVDM,2,2,3,B,OR0BH2kmH80,2*47
\g:1-2-8863,s:rORBCOMM000,c:1364774400*5a\!AIVDM,2,1,8,A,5342l602>;;LtpLR220huLh
4pA>0u04j22222216DHKD<4w`0?hPC52CClQH,0*29
\g:2-2-8863*58\!AIVDM,2,2,8,A,88888888880,2*2C
\g:1-2-8893,s:rORBCOMM008,c:1364774400*5d\!AIVDM,2,1,6,A,5=JDv<P000000000000PPP0
0000000000000000N0000040000000000000000,0*25
\g:2-2-8893*57\!AIVDM,2,2,6,A,00000000008,2*2A
\g:1-2-8925,s:rORBCOMM010,c:1364774400*58\!AIVDM,2,1,4,A,56:V7u@00007OP00001`PEP
T4u<P4pQDw;?O3H164H:2500007LLP0000000,0*3E
\g:2-2-8925*5B\!AIVDM,2,2,4,A,00000000000,2*20
```

Figure A3.2. Raw AIS data from Orbcomm for ship type messages. Each AIS transmission starts with "\g:", the timestamp starts at "c:", and the ship data starts at "\!AIVDM". The timestamp is decoded using specially written Java code. The encoded ship type messages contain information about the ship dimensions which are used to estimate its maximum speed when no data was available in static vessel databases online.

Table A3.1. AIS data parameters and descriptions. Parameters used in the AIS model included: User ID, Navigation status, Speed over ground, longitude, latitude, and timestamp.

Parameter	Description
Message ID	Identifier for this message 1, 2 or 3
Repeat indicator	Used by the repeater to indicate how many times a message has been repeated. 0 = default, 3 = do not repeat any more
User ID	MMSI number
Navigational status	0 = under way using engine, 1 = at anchor, 2 = not under command, 3 = restricted manoeuvrability, 4 = constrained by her draught, 5 = moored, 6 = aground, 7 = engaged in fishing, 8 = under way sailing, 9 = reserved for future amendment of navigational status for ships carrying DG, HS, or MP, or IMO hazard or pollutant category C, high speed craft (HSC), 10 = reserved for future amendment of navigational status for ships carrying dangerous goods (DG), harmful substances (HS) or marine pollutants (MP), or IMO hazard or pollutant category A, wing in ground (WIG); 11 = power-driven vessel towing astern (regional use); 12 = power-driven vessel pushing ahead or towing alongside (regional use); 13 = reserved for future use, 14 = AIS-SART (active), MOB-AIS, EPIRB-AIS 15 = undefined = default (also used by AIS-SART, MOB-AIS and EPIRB-AIS under test)
Rate of turn	0 to +126 = turning right at up to 708 deg per min or higher 0 to -126 = turning left at up to 708 deg per min or higher Values between 0 and 708 deg per min coded by ROTAIS = 4.733 SQRT(ROTsensor) degrees per min where ROTsensor is the Rate of Turn as input by an external Rate of Turn Indicator (TI). ROTAIS is rounded to the nearest integer value. +127 = turning right at more than 5 deg per 30 s (No TI available) -127 = turning left at more than 5 deg per 30 s (No TI available) -128 (80 hex) indicates no turn information available (default). ROT data should not be derived from COG information.
Speed over ground	Speed over ground in 1/10 knot steps (0-102.2 knots) 1 023 = not available, 1 022 = 102.2 knots or higher
Position accuracy	1 = high (<= 10 m) 0 = low (> 10 m) (default)
Longitude	Longitude in 1/10 000 min

	+/-180 deg, East = positive (as per 2's complement), West = negative (as per 2's complement) 181 deg (6791AC0h) = not available (default)
Latitude	Latitude in 1/10 000 min +/-90 deg, North = positive (as per 2's complement), South = negative (as per 2's complement). 91 deg (3412140h) = not available (default)
Course over ground	Course over ground in 1/10 = (0-3599). 3600 (E10h) = not available (default) 3 601-4 095 should not be used
True heading	Degrees (0-359) 511 = not available (default)
Second	UTC second when the report was generated by the electronic position system (EPFS) 0-59, 60 = not available (default) 61 = operating with manual input mode 62 = operating in estimated (dead reckoning) mode 63 = system inoperative
Special manoeuvre indicator	0 = not available (default) 1 = not engaged in special manoeuvre 2 = engaged in special manoeuvre (i.e.: regional passing arrangement on Inland Waterway)
Spare	Not used. Should be set to zero. Reserved for future use
Raim flag	Receiver autonomous integrity monitoring (RAIM) flag of electronic position fixing device; 0 = RAIM not in use (default) 1 = RAIM in use
Timestamp	Unix time stamp

10.3 Vessel types included in the AIS model

The following is a list of vessel types included in this project:

- Cargo
- Tanker
- Fishing
- Pilot
- Towing
- Rescue
- Law enforcement
- Medical transport
- Antipollution
- Military operations
- Tug
- High speed craft
- Passenger
- Sail
- Spare local vessel1
- Spare local vessel2
- Pleasure craft
- Port tender
- Wing-in-ground hazardous B
- Dredging or underwater operations
- Ship according to RR resolution No18
- Diving operations
- Other type
- No information available