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ASSESSING THE EXPOSURE AND BEHAVIOURAL RESPONSE OF GREY SEALS (*HALICHOERUS GRYPUS*) TO SHIPPING NOISE

by

LEAH TRIGG

A thesis submitted to the University of Plymouth in partial fulfilment for the degree of

DOCTOR OF PHILOSOPHY

School of Biological and Marine Sciences

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“Soon silence will have passed into legend...”

- Hans Jean Arp
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Author’s Declaration

At no time during the registration for the degree of Doctor of Philosophy has the author been registered for any other University award without prior agreement of the Doctoral College Quality Sub-Committee.

Work submitted for this research degree at the University of Plymouth has not formed part of any other degree either at the University of Plymouth or at any other establishment.

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Chapter 1, Chapter 2 and Chapter 6 were prepared and written by the author, and comments on the drafts of each chapter were provided by the supervisory team.

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Publications and presentations arising from the thesis are outlined on pages 5 and 6. The author also attended specialist training and workshops outlined on page 7 as part of ongoing professional development.

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Signed ..................................................
Summary of Research Outputs

Publications


Conference Presentations


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Internal and External Training Courses

- Preparing to submit on PEARL. University of Plymouth. January 2019.

Professional Development Activities

- Committee Member Marine Biological Association Postgraduate Conference. May 2017 - May 2018.
- Committee member UK and Ireland Regional Student Chapter of the Society for Marine Mammalogy. August 2016 - January 2018.
Abstract

Marine mammals use sound to drive vital life functions such as communication, foraging and navigation, but the underwater soundscape upon which they rely is changing. Growth in global trade and manufacturing has driven a dramatic increase in the number and size of ships in the commercial fleet. These ships generate chronic underwater noise which has been associated with a number of negative ecological effects such as auditory damage, changes in behaviour and stress. As a result, regulatory bodies have recognised underwater noise as a pollutant. However, there is still a lack of data relating to the noise levels experienced by marine life, and how exposure to shipping noise will impact certain species, frustrating efforts to set targets for acceptable levels of noise from shipping.

The grey seal (*Halichoerus grypus*) is a protected species with hearing sensitive to the dominant low frequencies of shipping noise. UK waters are home to approximately 38% of the global grey seal population but are also traversed by some of the world's busiest shipping lanes. As a result, there is high spatial overlap between grey seals and shipping traffic. However, knowledge of the impact of shipping noise on grey seals while at-sea is sparse. Consequently, this thesis aims to investigate the exposure and behavioural response of grey seals to underwater noise from shipping, and improve the efficiency with which predictions of shipping noise for this task can be calculated.

Using an acoustic modelling approach, ship noise levels were reconstructed along the GPS location and dive tracks of grey seals in the English Channel and Celtic Sea. The m-weighted 24-hr cumulative sound exposure levels of seals ranged from 121 to 170 dB re $1\mu Pa^2 s$. The exposure of seals was influenced by the maximum source level, number and closest point of approach of ships in relation to the location of the seal, and noise predictions varied with depth as seals moved throughout the water column. Main findings indicate that while the exposure of seals to shipping noise was not high enough to cause damage to
auditory systems, it was great enough to result in changes in the diving behaviour of grey seals in both regions. Adult seals increased the ascent rate of benthic and shallow dives, and seal pups decreased the descent rate of pelagic dives as a result of exposure to shipping noise at median sound pressure levels of 122 and 111 dB re 1 µPa respectively. The efficiency with which predictions of exposure along the tracks of seals was calculated improved 5 fold by developing a grid with an adaptive cell size to aggregate ships.

These findings contribute to our understanding of the potential risk that shipping noise poses to grey seals, facilitates more efficient and accurate assessment of underwater noise and informs future policy that seeks to protect marine ecosystems from shipping noise.
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List of Abbreviations

AIC ......... Akaike’s Information Criterion
AIS ......... Automatic Identification System
ANSI ....... American National Standards Institute
ASA ......... American Standards Association
ATOC ....... Acoustic Thermometry of the Ocean Climate
CPA ......... Closest point of approach of a ship in 15 minute period
$cSEL_{24}$ .... M-weighted 24-hr cumulative sound exposure level (dB re $1\mu Pa^2s$)
$cSEL_{eq}$ .... M-weighted 24-hr cumulative sound exposure level above
effective quiet (dB re $1\mu Pa^2s$)
$cSEL_{15}$ .... M-weighted 15-min cumulative sound exposure level (dB re $1\mu Pa^2s$)
DTAG ....... Digital Acoustic Recording Tag
EMODnet .... European Marine Observation and Data Network
EU ......... European Union
GAMM ....... Generalised Additive Mixed Model
GES ......... Good Environmental Status
GPS ......... Global Positioning System
GSM ......... Global System for Mobile Communications
HELCOM ...... Baltic Marine Environment Protection Commission - Helsinki Commission
IMO ......... International Maritime Organisation
ISO ......... International Organization for Standardization
IQR ......... Inter-Quartile Range
MMSI ....... Maritime Mobile Service Identity
ML . . . . . . Maximum Likelihood

NEMO . . . . Nucleus for European Modelling of the Ocean

NOAA . . . . National Oceanic and Atmospheric Administration

NUM . . . . Number of ships in 15 minute period

OSPAR . . . . Convention for the Protection of the Marine Environment of the
               North-East Atlantic (derived from Oslo and Paris conventions)

PCAD . . . . . Population Consequences of Acoustic Disturbance

PCoD . . . . . Populations Consequence of Disturbance

PL . . . . . . . Propagation Loss

POLCOMS . . Proudman Oceanographic Laboratory Coastal Ocean
               Modelling System

PTS . . . . . . Permanent Threshold Shift

RANDI . . . . Research Ambient Noise Directionality

REML . . . . . Restricted Maximum Likelihood

RL . . . . . . . Received Level (dB re 1$\mu$Pa)

SL . . . . . . . Source Level (dB re 1$\mu$Pa)

$SL_{max}$ . . . . Maximum source level in a 15 minute period (dB re 1$\mu$Pa)

SONIC . . . . Suppression of Underwater Noise Induced by Cavitation

SPL . . . . . . Sound Pressure Level (dB re 1$\mu$Pa rms)

$SPL_{max}$ . . . . Maximum sound pressure level in a dive (dB re 1$\mu$Pa rms)

SOLAS . . . . Safety of Life at Sea

TTS . . . . . . Temporary Threshold Shift

UNCTAD . . . United Nations Conference on Trade and Development

VHF . . . . . . Very High Frequency

VIF . . . . . . Variance Inflation Factor
1 | General Introduction
1.1 Project rationale and aims

Global commercial shipping underpins trade and economic development, and with the globalisation of manufacturing and financial markets, shipping has increased dramatically since the start of the 20th century (Hoffmann and Kumar 2010). In 1970, the world commercial fleet had a carrying capacity of 320 million deadweight tonnes but this has increased to 1.92 billion deadweight tonnes carried by more than 94,000 commercial ships in 2018\(^1\) (UNCTAD 2018). The ships joining the fleet in the last 10 years are also, on average, 7 times larger than those built over 20 years ago (UNCTAD 2018). Commercial ships emit low frequency underwater noise from propeller cavitation, machinery onboard the ship and the flow of water past the vessel (Urick 1983). Acoustic intensity is generally greatest between 10 and 1000 Hz (Arveson and Vendittis 2000; McKenna et al. 2012) but noise can extend to frequencies beyond 96 kHz (Hermannsen et al. 2014; Veirs et al. 2015). The acoustic footprint of increasing maritime trade has been linked to a 3.3 dB per decade increase in underwater ambient noise levels between 1950 and 2007 (Frisk 2012). Specifically, measurements of ambient noise in the north-east Pacific increased by 10 dB between the 1960s and 1990s (Andrew et al. 2002). International seaborne trade saw its fastest growth in 5 years during 2017, increasing by 4%, and it is projected that compound annual growth in shipping between 2018 and 2023 will be as much as 3.8% (UNCTAD 2018). Consequently, if not addressed, underwater noise from shipping will continue to rise in the coming years.

Sound is central to the life strategy of many marine animals because it travels more efficiently in the ocean than light (Au and Hastings 2008). Shipping noise is one of many anthropogenic noise sources that are altering the underwater soundscape within which marine animals communicate, navigate and forage (Tyack 2008; Madsen et al. 2006). There are very few regions of the ocean that

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\(^1\)UNCTAD define commercial fleet to include propelled sea-going merchant vessels of 100 gross tonnes or above excluding inland waterways, fishing vessels, military vessels, yachts, ocean platforms and barges.
are not traversed by ships, and the low frequency noise can travel very long
distances in the ocean (Porter and Henderson 2013). Consequently, shipping
noise has become a pervasive and chronic presence in the oceans, and
contributes to an increase in overall low frequency ambient noise (Ross 2005),
as well as more acute exposures above ambient noise levels when animals and
ships are are close together in space (Mikkelsen et al. 2019). As a result,
shipping noise has been recognised internationally as a widespread
environmental pollutant (IMO 2014; Vancouver Fraser Port Authority 2017;
European Commission 2008).

An increasing weight of evidence demonstrates the potentially detrimental
impact of shipping noise on marine animals. Impacts include the masking of
calls between conspecifics (Vasconcelos et al. 2007; Jensen et al. 2009; Hatch
et al. 2012), potential damage to the structures in the ear (Finneran 2015; Jones
et al. 2017), changes in behaviour such as reduced antipredator responses,
reduced foraging or habitat exclusion (Simpson et al. 2015; Dyndo et al. 2015;
Blair et al. 2016) and physiological changes such as an increase in stress
(Wysocki et al. 2006; Rolland et al. 2012; Celi et al. 2015). If such impacts affect
the ability of the animal to undertake important life functions such as foraging,
breeding and escaping from predators, noise may have population as well as
individual level effects (New et al. 2013; Pirotta et al. 2018). This is particularly
important for marine ecosystems that are already under the strain of great
environmental change and pollution (Cosgrove et al. 2016; Tulloch et al. 2019).

As such, regulatory bodies recognise the need to characterise and mitigate
against the risks posed by shipping noise. International organisations such as
the United Nations (UN 2018) and International Maritime Organisation (IMO
2014) have taken steps to include the issue on their agenda, but specifically, the
European Union (EU) has included noise pollution within the Marine Strategy
Framework Directive, which aims to ensure the ‘Good Environmental Status’ of
EU waters by 2020 (European Commission 2008; European Commission 2010;
European Commission 2017). This focus on underwater noise as a pollutant has
been mirrored by regional conventions such as OSPAR and HELCOM that have undertaken to monitor underwater ambient noise in the north-east Atlantic and Baltic Sea areas respectively (OSPAR Commission 1992; Baltic Marine Environmental Commission 1992). However, it is difficult to set targets for which the shipping industry can strive without an understanding of current noise levels and its impacts on marine ecosystems against which to track trends in noise and assess the efficacy of policy interventions (Van der Graaf et al. 2012; Merchant 2019). Existing research, while highlighting the potential negative impacts, is limited in the number of species, regions and types of impact it addresses. Historically, there has been a focus on marine mammals because they utilise sound for important functions such as foraging and communication (Tyack 2008). The mysticetes, particularly, are thought to have low frequency hearing which overlaps with the shipping noise spectra, attracting research in the area (Richardson et al. 1995; Tyack 2008; Rolland et al. 2012). However, pinnipeds are a group of marine mammals that remain poorly studied with respect to shipping noise (Jones et al. 2017; Mikkelsen et al. 2019). Specifically, there is a limited understanding of the exposure and behavioural response of pinnipeds to this noise source, despite an ability to hear in the frequency range of shipping noise, and their spatial overlap with shipping traffic in coastal zones (National Marine Fisheries Service 2018; Jones et al. 2017).

Pinnipeds, specifically grey seals (*Halichoerus grypus*), are amphibious marine mammals that undertake trips to sea between land-based haul-outs where they pup, breed and moult (Thompson et al. 1991; McConnell et al. 1992; Pomeroy et al. 1994; McConnell et al. 1999). Approximately 38% of the world’s population of grey seals breed at locations around the UK coastline, and estimates of the UK population suggest it grew by approximately 1.8% per year between 2012 and 2017 (SCOS 2018). This was driven primarily by an increase in pup production at colonies in the North Sea (SCOS 2018). Despite this, they face a number of potential threats to their continued success including bycatch (Cosgrove et al. 2016), disease (Yon et al. 2019), marine renewable energy
installations (Russell et al. 2016; Hastie et al. 2018), culling (Thompson et al. 2007) and climate change (SCOS 2018). Shipping noise has the potential to exert further pressure on such challenged seal populations. Their at-sea distribution and the location of haul-outs in the coastal zone places them at risk of encountering high levels of shipping traffic. Co-occurrence of seals and shipping in the UK is high (≥ 100 daily co-occurrences in 5 km grid cell) within 50 km of the coast and particularly near haul-out sites (Jones et al. 2017).

Furthermore, the functional hearing range of phocid seals (50 Hz - 86 kHz) demonstrates they are sensitive to the dominant low frequency range of shipping noise (National Marine Fisheries Service 2018). Sound plays an important role in underwater communication and foraging in seals. Grey seals emit low frequency vocalisations such as growls (100 - 500 Hz), rups (100 - 3000 Hz) and clicks (3000 Hz), primarily thought to be associated with social activity and breeding (Asselin et al. 1993). These social calls overlap in frequency with recorded underwater noise from ships, suggesting that shipping noise may reduce the ability of grey seals to communicate (Bagočius 2014). For foraging seals, passive listening is an important tool (Schusterman et al. 2000). It allows the localisation of prey, conspecifics and predators, as well as having a role in spatial orientation and navigation (Schusterman et al. 2000). This was highlighted during a simulated foraging experiment with captive seals (Stansbury et al. 2015). Seals were required to locate fish in two of 20 foraging boxes. One box contained a fish with an acoustic tag; a device which emits an auditory signal, and another box contained a fish with no tag. The box with a tagged fish was found after significantly fewer visits to the empty boxes, suggesting the ability of seals to detect and localise acoustic fish tags as a signal for the presence of food (Stansbury et al. 2015). In addition, captive and wild seals have demonstrated a sensitivity to a number of noise sources. A behavioural response has been detected for phocid seals in response to acoustic deterrent devices (Götz and Janik 2013), impulsive noise from pile driving for the installation of wind farms (Russell et al. 2016), tidal turbine noise (Hastie et al.
2018) and sonar playbacks (Hastie et al. 2014).

However, the research on the relationship of phocid seals, and specifically grey seals, to shipping noise is more sparse. Hauled-out seals react to approaching boats and ships by flushing into the sea, displaying alert behaviours such as head-raising, undertaking orienting behaviour and increases in aggressive interactions with conspecifics (Jansen et al. 2010; Tripovich et al. 2012; Andersen et al. 2012; Niemi et al. 2013; Blundell and Pendleton 2015). However, this often may be related to disturbance by the presence of boats and tourists rather than in-air shipping noise. There is much less information regarding the at-sea exposure and behavioural response of seals to underwater shipping noise. This distinction is important given the differences in shipping noise characteristics underwater and in-air (Dahl et al. 2007; Badino et al. 2012).

A study of the at-sea exposure of harbour seals (*Phoca vitulina*) in the Moray Firth predicted, using a modelling approach, that seals were exposed to mean m-weighted 24-hr cumulative sound exposure levels from shipping noise of 176 dB re $1 \mu P a ^2 s$ (Jones et al. 2017). The upper limits of 95% confidence intervals were above estimated thresholds for auditory damage in seals (Jones et al. 2017). This is of concern for policy that must ensure the ‘favourable conservation status’ of this protected species under Annex II of the EU Habitats Directive (European Commission 1992). However, this study predicted noise at a single uniform depth, and hence neglected a key characteristic of seal behaviour. As air-breathing benthic foragers, their at-sea behaviour is characterised by dives throughout the water column, where they primarily forage on the seafloor at depths over 100 m, before returning to the surface to breathe (Boyd 1997; Thompson et al. 1991; SCOS 2018). Stratification in water column properties such as temperature and salinity influences sound speed and propagation in the ocean. In the Celtic Sea between south-west UK and southern Ireland, the presence of a summer thermocline can result in an average step change of 10 dB (10 - 1000 Hz) in exposure for seals as they dive from one depth to another (Chen et al. 2017). However, the cumulative exposure of seals with respect to
their horizontal and vertical movement in the water column is yet to be considered.

Preliminary observations of the diving behaviour of grey seals using tags that record acoustic and movement data (DTAGs) in the North Sea, gave examples of seals changing their dive behaviour in response to a nearby ship (Mikkelsen et al. 2019). However, these tags were only operational for approximately 12 days, and it can be difficult to analyse the acoustic data they record due to noise from the flow of water past the device (Benda-Beckmann et al. 2016). Hence, there is a need for a more detailed analysis of changes in the at-sea diving behaviour of seals in response to shipping noise. Seals are an excellent model species for studying the impact of noise on marine mammals at different depths because there are alternative well developed long-term (2 - 15 months) telemetry tags that can record the at-sea horizontal and vertical movement of seals over many months (Carter et al. 2016). However, these long-term tags, unlike the DTAG, do not record noise levels. Therefore, shipping noise at the location of the seal must be determined by another method. Until long-term DTAGs move beyond proof-of-concept (Mikkelsen et al. 2019), this can be achieved through modelling noise.

Models of shipping noise are useful for such tasks because they allow the calculation of noise levels at different spatial scales, including at the location of individual animals, and they are relatively efficient to transfer to new regions compared to hydrophone deployments (Dekeling et al. 2014). However, modelling can be computationally expensive and time consuming. This is particularly true in areas with a high number of ships, each of which must be modelled. In response, the most sophisticated, accurate and time consuming models are often overlooked in favour of very fast simple geometric spreading laws that assume a logarithmic decay in acoustic intensity with range from the source (Etter 2013; Marine Management Organisation 2015). However, these laws do not account for changes in sound speed driven by environmental variation, and consequently, can result in large errors in predicted noise levels in
environmentally dynamic regions such as shallow shelf seas (Farcas et al. 2016). Such errors could result in failures to protect sensitive marine species or excessive limitations on marine activities (Merchant 2019). However, this could be addressed by improving the efficiency of the most accurate modelling solutions allowing users to implement more realistic modelling workflows in a wider range of scenarios. This is particularly relevant for predicting the exposure of pinnipeds to shipping noise. Characterising fine-scale changes in noise levels horizontally and vertically requires sophisticated models with high computational costs (Etter 2013; Wang et al. 2014). Therefore, an improvement in the ship noise modelling methodology is required to make such activities more achievable in policy and research settings.

To address the current shortcomings in our understanding of the impacts of shipping noise on seals, this thesis aims to investigate the exposure and behavioural response of diving grey seals to underwater noise from shipping. It also aims to improve the methodology available to model underwater noise from shipping in order to make accurate exposure calculations for marine species more accessible in policy settings. Specifically, the subsequent chapters of this thesis address the following objectives.

1. Review the literature on underwater noise from shipping, measuring and modelling shipping noise, and its impact on marine mammals.

2. Predict the exposure of grey seals to underwater noise from shipping and assess the implications of this exposure with respect to auditory damage in seals.

3. Investigate the influence of shipping traffic on the exposure of grey seals to shipping noise.

4. Investigate the impact of shipping noise on the diving behaviour of grey seals.

5. Improve the efficiency of modelling underwater noise from shipping.

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6. Evaluate the results of the thesis in the wider context of regulatory planning and the management of underwater noise.

1.2 Thesis structure

This chapter (Chapter 1) has given a general introduction to the thesis and outlined the aims and objectives that will be addressed in the following chapters. This is complemented by Chapter 2 which provides a detailed review of the current literature on underwater noise from shipping, how it is measured and modelled, the impacts of shipping noise on marine mammals, possible solutions and future research. The chapter addresses Objective 1 and will help set the remaining chapters in the wider context of shipping noise research on marine mammals and the modelling of underwater noise from shipping.

Chapter 3 presents predictions of the exposure of grey seals to shipping noise, and gives an assessment of the factors that are driving the predicted shipping noise exposure. The chapter addresses the work laid out in Objective 2 and Objective 3, and will assess exposure with respect to auditory damage in seals. It aims to broaden our understanding of seal exposure by studying exposure levels with respect to the horizontal and vertical at-sea movement of seals, and includes seals at different life stages, particularly, adults and pups. The results will have implications for the management of shipping noise for seal conservation.

Chapter 4 addresses Objective 4 by presenting analysis of seal diving behaviour in response to shipping noise exposure. At present, there is very little information regarding the impact of shipping noise on the diving behaviour of seals. The results will help address this gap and builds on Chapter 3 by utilising the predicted exposure levels calculated in that chapter in addition to dive metrics from telemetry tags. As a result, the thesis presents an assessment of both the potential auditory damage and behavioural responses of grey seals to shipping noise.
Chapter 5 is presented in support of Objective 5 and complements Chapters 3 and 4 by allowing the methodology utilised in those chapters to be more easily implemented in new settings, and in locations with heavy shipping traffic. It aims to provide a gain in efficiency without loss in accuracy, and can help the assessment of underwater noise exposure for marine animals in a regulatory context. The chapter presents an adaptive grid to aggregate ships when modelling shipping noise. It aims to reduce the number of times the model must be executed and hence reduce overall execution time.

Chapter 6 concludes the thesis by addressing Objective 6. It will discuss the implications of the results from Chapters 3 to 5 for grey seal conservation, and given the context set out in the literature review (Chapter 2), it will address the significance and possible applications of the key findings to shaping future regulations and the management of underwater noise.
A review of marine mammals and shipping noise: current knowledge, challenges and solutions

2.1 Introduction

Ships are highly mobile sources of underwater noise pollution. As global economic development has driven growth in commercial shipping, underwater noise from the fleet has become increasingly ubiquitous in the world’s oceans (Fig. 2.1) and low frequency (10 - 1000 Hz) (Ross 1976; Urick 1983) underwater noise from ships is an ever larger component of the underwater ambient noise budget (Andrew et al. 2002; Ross 2005; McDonald et al. 2006; McDonald et al. 2008; Andrew et al. 2011; Chapman and Price 2011). In response, legitimate concern about how growing levels of underwater noise from shipping impacts marine ecosystems has emerged, and resulting studies have generated a growing body of evidence to suggest its deleterious impact on marine life.

Marine mammals are of specific concern with respect to shipping noise because of the central place sound plays in their life strategies (Au and Hastings 2008). Sound waves travel very efficiently in the ocean and as a result, it is most effective for marine mammals to use sound production and perception to drive important activities such as communication, navigation and prey detection (Richardson et al. 1995). This makes them particularly vulnerable to changes in the underwater soundscape because it could impact their ability to carry out these activities through mechanisms such as communication masking, habitat abandonment and chronic stress (Erbe et al. 2015; Nowacek et al. 2007; Southall et al. 2007; Rolland et al. 2012), which may ultimately determine individual and population level success (Nabe-Nielsen et al. 2018). This vulnerability to noise has catalysed research into the impact of all forms of underwater noise on marine mammals and this field has been reviewed as a complete body of work by Richardson et al. (1995), Weilgart (2007b) and Tyack (2008). In addition, there are a number of specific reviews that focus on; the behavioural responses of marine mammals to noise (Wartzok and Popper 2003; Gomez et al. 2016), the threat of noise from wind turbines (Madsen et al. 2006), communication masking (Erbe et al. 2015) and the effects of anthropogenic
noise specifically to cetaceans (Nowacek et al. 2007; Weilgart 2007a). In contrast to impulsive (short duration) noise sources such as pile driving and seismic airguns, shipping noise is a low frequency continuous noise source. As a result of these different characteristics, each noise type is subject to independent scientific and regulatory consideration (Van der Graaf et al. 2012). For example, EU regulation on noise (Marine Strategy Framework Directive) has separate indicators of ‘Good Environmental Status’ for impulsive and continuous sounds (European Commission 2010; European Commission 2017). However, at the time of writing, the literature that deals with shipping noise is yet to receive a specific review¹. In response, the aim of this review is to analyse the current understanding of shipping noise with respect to marine mammals in order to inform future marine mammal conservation, policy and research. Specifically, it will examine the characteristics of and trends in shipping noise to provide context for future research, it will review how shipping noise is measured and modelled, and the known impacts on marine mammals in order to identify data gaps and suggest future research to inform subsequent regulation of shipping noise.

2.2 What is shipping noise?

Shipping noise is a low frequency broadband noise source with peak spectral power between 10 - 1000 Hz (Urick 1983). It also has tonal components and can extend to frequencies of 96 kHz and higher (Hermannsen et al. 2014; Veirs et al. 2015). There are three main sources of radiated noise from a commercial ship; propeller noise, machinery noise and hydrodynamic noise (Urick 1983). The properties of these noise sources are summarised in Boxes 2.1, 2.2 and 2.3 respectively, and are discussed in detail by Ross (1976) and Urick (1983). The noise spectrum emitted by a ship is dependent on a number of factors such as the type of ship, the propulsion system, the speed of the ship, the time of year, propeller type and size (McKenna et al. 2013; Simard et al. 2016). For example, ship noise tends to increase with ship speed (Arveson and Vendittis 2000), but a

Figure 2.1: Global Shipping Density from January to March 2010 on a 1° × 1° grid. Shipping density is given by average number of vessels in a grid cell for 10 orbits of an AIS satellite. The map shows the near ubiquitous spatial coverage of modern commercial shipping and highlights the particularly high density of ships in productive coastal regions and shelf areas. These areas are particularly important for marine mammals. Image attributed to: PASTA MARE project European Commission.

A higher number of propeller blades has been shown to reduce noise by decreasing the size of air bubbles which collapse to cause cavitation noise (Ebrahimi et al. 2019). Shipping noise is emitted into the water at shallow depths (1-7 m) and interacts with the water surface (McKenna et al. 2012; Gassmann et al. 2017; Jansen and Jong 2015). As a result, radiation tends to be directed vertically downwards (Wittekind 2014). Noise is also radiated from different locations on the ship with machinery noise located towards midship and propeller noise to the aft (Abrahamsen 2012). This impacts the pattern of radiated noise in the environment.

Our understanding of the characteristics of ship noise are primarily derived from measurements of ships during World War II (Ross 1976; Urick 1983). The design of ships, their size and propulsion systems have changed dramatically since that era (Richardson et al. 1995). Economic and environmental pressures have resulted in a drive for fuel efficiency (Ross 2005). Consequently, more efficient and quieter slow drive diesel electric propulsion systems have become more popular (Ross 2005). In contrast, the flexibility and power generation from controllable pitch propellers is increasingly popular and their sale now holds a
As described by Urick (1983), propeller noise is produced outside the hull, propagates directly into the water and is generally amplitude modulated with the propeller beats. The hull and wake restrict propagation to the fore and aft respectively. Generally, noise associated with the propeller is generated by cavitation. This is the major source of noise associated with shipping. Propeller cavitation arises due to the differential pressures experienced at the surface and tips of the propeller blades during its motion (Abrahamsen 2012). Areas of low pressure cause water to vaporise and form small bubbles in the water. These bubbles collapse violently when they move away from the low pressure area into the turbulent wake or as they impact against the propeller blades themselves (Abrahamsen 2012). Bubble collapse causes a loud noise due to the creation of momentary high pressures as the contents of the bubble are compressed. Propeller cavitation results in broadband noise that peaks at the higher end of the ship noise spectrum between 100 and 1000 Hz (Arveson and Vendittis 2000). Cavitation begins at the cavitation inception speed where blade rotation is sufficiently high to cause the pressures required for bubble formation (Urick 1983). This property is complex and dependent on a number of ship specific factors such as hull form, propeller design, extent of propeller fouling, revolutions per minute of the propeller and number of propeller blades (Arveson and Vendittis 2000; Ebrahimi et al. 2019). Propeller singing is less prevalent but nonetheless makes an important contribution to propeller noise (Urick 1983). Propeller singing is caused by the resonant vibration of the propeller at tones between 100 and 1000 Hz depending on the propeller diameter, speed of revolution and shape of the propeller (Urick 1983). Propeller singing can be easily addressed with propellers produced from high damping alloys or alterations to the shape of the propeller tips (Ebrahimi et al. 2019).

Box 2.1: Propeller Noise

35% market share (Carlton 2019). These propellers cavitate more, and therefore are noisier, than traditional fixed propellers at low speeds (Carlton 2019).

Measurements of modern ships are required to effectively understand the characteristics of shipping noise at present and how to implement effective noise reduction schemes for marine mammals.

Attempts to measure the broadband source levels of ships present varied results. Broadband sound pressure levels can be as high as 202 - 209 dB re $1\mu Pa$ (5 Hz - 1 kHz) for icebreakers and container ships (Erbe and Farmer 2000a; Gassmann et al. 2017). However, a smaller 18.9 m research vessel varied between approximately 155 and 175 dB re $1\mu Pa$ (100 Hz - 10 kHz) as
Machinery noise (1 - 5000 Hz) originates inside the ship and is transferred to the surrounding water through the vibration of the hull (Urick 1983). As a result, the level of noise produced by the mechanical components of a ship is not only dependent on the magnitude of the source, but also on the efficiency by which it is transferred to the surrounding water. The major contributors to machinery noise on-board a ship are the main propulsion system, generators, gears and propeller shafts (Urick 1983). The consistent and repetitive nature of mechanical action usually results in noise in the form of specific tones at low frequencies (Urick 1983).

Hydrodynamic noise, like propeller noise, is generated outside the hull and arises from the flow of fluid past the moving vessel. Irregular flows are generated by interaction with the hull via the wake and bow wave and excitation of the hull (Urick 1983). The pressure fluctuations that arise may be radiated as sound or may induce vibrational resonance in the hull (Urick 1983). The majority of noise is due to breaking waves at the bow and stern, and is caused by oscillating air bubbles (Urick 1983). The contribution of hydrodynamic noise generated by the hull is generally believed to be much less than that of machinery and propeller noise (IMO 2014).

Speed increased (Brooker and Humphrey 2016). Furthermore, opportunistic measurements of passing commercial ships can range from 181 - 186 dB re $1\mu Pa$ (20 Hz - 1000 Hz) (McKenna et al. 2012) and 174 - 178 dB re $1\mu Pa$ (20 Hz - 40 kHz) (Veirs et al. 2015). These measurements were up to 15 dB lower than opportunistic source level measurements by Simard et al. (2016). These variations can in part be attributed to the inconsistencies in the approach used to measure shipping noise. Particularly, the handling of surface reflections and the estimation of source levels using propagation models (Sec. 2.4.2.3). The introduction of standards for the measurement of underwater sound from ships such as ISO 18405:2017, ANSI/ASA 12.64, ISO 17208-1:2016 and ISO 17208-2:2019 should assist in the generation of future measurements that are comparable, especially given further standards such as ISO 17208-3:201X yet to be released. However, despite inconsistencies in the source level measurements, the results of these studies illustrate that ships constitute a
significant underwater noise source especially given their global distribution (Fig. 2.1) and increasing numbers (UNCTAD 2018).

2.3 Shipping noise: past, present and future

In the years between 2001 and 2012, the world saw the greatest shipbuilding cycle ever recorded (Fig. 2.2) (UNCTAD 2014; UNCTAD 2015). Data has shown a substantial increase in the number of ships and the gross tonnage of the commercial fleet since the 1980s (Fig. 2.2), driven by the globalisation of manufacturing and trade (Frisk 2012). At the height of growth in 2011 the fleet grew by nearly 10%. Following the 2008 financial crisis, growth was at its slowest pace for nearly 10 years with 4.1%, 3.5%, 3.48% increases in the world’s commercial fleet reported in 2013, 2014 and 2015 respectively (UNCTAD 2014; UNCTAD 2015; UNCTAD 2016). However, recovering economic growth in many nations saw global seaborne trade (i.e. volume of goods transported) grow by 4% in 2017; the fastest growth for 5 years. Similarly, the number of ships again increased with the commercial fleet comprising 94,171 vessels with a 1.92 billion deadweight tonne carrying capacity as of January 2018 (UNCTAD 2018).

The fate of the shipping industry is inherently uncertain. It is intimately intertwined with the global economy and is subject to any shocks felt in global markets (Hoffmann and Kumar 2010; Frisk 2012; UNCTAD 2018). There is currently global overcapacity in the fleet and it must invest in new technologies to meet resolutions for conventions on emissions and ballast water (IMO 2004; IMO 1973). Nevertheless, compound annual growth in seaborne trade is expected to be 3.8% per year until 2023 (UNCTAD 2018), and industry leaders predict continued increases in the number of vessels and total tonnage to 2030 (Lloyd's Register 2013). Therefore, the noise emitted from this fleet, and issues such as ship strike, are of growing concern for marine mammal populations.

The trends in shipping traffic have generally been reflected in increasing trends in underwater ambient noise measurements. Studies examining multi-year recordings have found increases in low frequency ambient noise (< 300 Hz) of
Figure 2.2: Growth in the world commercial shipping fleet since 1980. a) Growth in the number of ships since 2011. b) Growth in size of commercial fleet since 2011. c) Growth in carrying capacity of the commercial fleet since 1980. Demonstrates the explosive growth in shipping since the late 1990s. Data from UNCTADstat.

between 1 and 12 dB at sites in the north-east Pacific Ocean between the 1960s and late 2000s (Andrew et al. 2002; McDonald et al. 2006; Andrew et al. 2011; Chapman and Price 2011). An increase in ambient underwater noise has also been seen between 2002 and 2012 in the Indian Ocean (Miksis-Olds et al. 2013), and between 1966 and 2014 near Bermuda in the North Atlantic where an increase of 2.8 dB (44Hz) was recorded (Širović et al. 2016). However, noise levels are closely linked to the local conditions and decreases of a similar magnitude in underwater noise have been seen for some frequencies at Ascension Island and Wake Island in the South Atlantic and Equatorial Pacific respectively (Miksis-Olds and Nichols 2016), and at Cape Leeuwin in the Indian Ocean due to local changes in the soundscape (Harris et al. 2019). These locations were not dominated by shipping noise and changes in the level of seismic survey and biological activity from whales are thought to have driven the changes at Ascension and Wake Islands (Miksis-Olds and Nichols 2016), while possible explanations for decreases in sound pressure levels at Cape Leeuwin
are changes in Antarctic sea ice volume (Harris et al. 2019). McDonald et al. (2008) reported that removing peaks in noise associated with ships near a hydrophone resulted in marginal change in overall ambient noise levels between 1963 and 2006 at San Clemente Island, California. However, these peaks from local shipping were found in 89% of recordings in 2006 and only 31% in 1963, and for marine mammals, the peaks in noise could be as important or more than ambient noise increases (Ellison et al. 2012). If the relationship between noise and shipping remains unchanged, the projected rise in shipping over the coming years suggests that shipping noise will also increase in the future (Kaplan and Solomon 2016). However, this will be influenced by economic growth, legislation, ship design and even changing sea conditions as a result of climate change (Ilyina et al. 2010; Rossi et al. 2016).

2.4 Methods of studying shipping noise for marine mammal research

The ability to quantify shipping noise is crucial for monitoring long-term trends in response to policy targets, understanding ship noise spectrum characteristics, and measuring the exposure levels received by marine mammals. Primarily this can be achieved by measuring noise directly in the field using a hydrophone, computational models, or some combination of the two approaches.

2.4.1 Measuring shipping noise using a hydrophone

Deploying a hydrophone is not a simple endeavour and there are many methodological decisions involved, such as the sensitivity, frequency response, dynamic range and calibration of the hydrophone and deployment depth (Zimmer 2011; Robinson et al. 2014). However, guideline documents are available which highlight the best practice for deploying hydrophones for a variety of purposes (Ainslie 2011; Zimmer 2011; Robinson et al. 2014). In addition, there are also ISO and ANSI standards for the measurement of underwater noise under different conditions and for the correct use of terminology (ISO 17208-1:2016;
ISO 17208-2:2019; ISO 18405:2017; ANSI/ASA S12.64). Despite this, there is still wide variation in the methodologies employed within the literature. Monitoring efforts in one location are rarely comparable to similar projects at different locations, and ship source spectra may vary due to measurement and analysis protocols (Sec. 2.2). This highlights the need for further standardised measurement protocols that would allow comparison across time and space for studies that measure shipping noise for a variety of aims. This will be particularly important as noise levels increasingly come under the control of legislation (see Sec. 2.6.2) and a measure of effectiveness will be required.

In relation to shipping noise, the majority of studies that have deployed hydrophones have undertaken measurements at long-term (years to decades) or short-term (weeks to months) fixed moorings (Arveson and Vendittis 2000; Andrew et al. 2011; McKenna et al. 2013; Schaar et al. 2014; Veirs et al. 2015), or for a few hours/days from a stationary boat (Würsig and Greene 2002; Bittencourt et al. 2014). Relatively fewer studies towed a hydrophone behind a moving vessel (Williams and Bain 2002; Buckstaff 2004). Vessel deployments are advantageous because the data can be easily monitored during collection, they are easily moved and there is a low risk of losing equipment (Robinson et al. 2014). However, it is impractical for long-term monitoring and can suffer from contamination by high levels of noise and confounding factors associated with the presence of the vessel conducting measurements (Robinson et al. 2014). In contrast moored systems are ideal for long-term deployments because they can be left unattended and they take measurements at a single repeatable location (Crawford et al. 2018; Vukadin et al. 2018). They are, however, at risk from data loss, disturbance and extraneous noise from moorings and tidal flow (Robinson et al. 2014; Crawford et al. 2018).

The depth of hydrophone deployments vary within the water column and include surface buoys, seabed moorings and suspended hydrophones 1-2 metres above the seabed (Robinson et al. 2014). Bottom mounted hydrophones are beneficial because they are isolated from artificial signals related to surface wave action.
and swell (Vukadin et al. 2018; Crawford et al. 2018). They are, however, influenced by reflections and currents at the seabed (Robinson et al. 2014). This is a particular problem in shallow water where sound waves interact with the seabed many times during propagation (Robinson et al. 2014). As a result, to avoid surface and bottom disturbance, it is recommended that deployment should be within the lower half or quarter of the water column (Robinson et al. 2014). Deviations from this may be necessary in deep water where deployment at such depths may be impractical or when there is motivation to study a particular layer within the water column (Robinson et al. 2014).

Many studies measure the overall contribution of shipping to ambient noise (Andrew et al. 2011; Chapman and Price 2011). Many others employ an opportunistic methodology where hydrophones record continuously, to capture source spectra from ships that happen to pass the hydrophone moorings (McKenna et al. 2013; Veirs et al. 2015). Relatively fewer measure a specific ship in an experimental setting where they have control over the movements and speed of the ship (Arveson and Vendittis 2000; Trevorrow et al. 2008; Brooker and Humphrey 2016). The Port of Vancouver is implementing a monitoring program which straddles both techniques. Each ship docking must pass over a fixed hydrophone under a standard protocol to develop a database of ship spectra (Vancouver Fraser Port Authority 2017). The measurement of ship source levels contends with a number of outstanding issues. Source level estimates are generated by calculating the propagation loss between the measurement location and the ship location. This is often achieved using simple geometric spreading laws (Sec. 2.4.2.3). These do not account for environmental variation in the water column potentially resulting in error (Farcas et al. 2016). As discussed in Section 2.2 measurements using these techniques have reported very different source levels because, for example, the measured aspect of the ship is very influential due to the directional nature of shipping noise. In addition, source levels can be different based on if they have been corrected for surface interactions (Lloyd Mirror Effect) (Gassmann et al. 2017).
When considering the impact of shipping noise on marine mammals, hydrophone deployments do not consider noise levels at the location of an individual animal. This would need to be estimated using a model of acoustic propagation, and makes it difficult to link changes in animal behaviour with noise exposure. In contrast, acoustic telemetry tags, which can be attached to an individual animal, record noise and behavioural parameters, and were initially developed to examine vocalisation characteristics and rates in target species (Johnson et al. 2009). However, they can also be useful for assessing the impact of shipping noise on marine mammals (Houghton et al. 2015; Blair et al. 2016). Crucially, they measure noise levels at the animal, and allow the synchronised recording of behaviour and acoustic signals, improving confidence in any cause and effect relationship between noise and behavioural change.

The most prevalent acoustic telemetry device is the Digital Acoustic Recording Tag (DTAG). It was developed with whales in mind and has been deployed on North Atlantic right whales (*Eubalaena glacialis*) (Nowacek 2004; Parks et al. 2011), sperm whales (*Physeter macrocephalus*) (Johnson and Tyack 2003), short-finned pilot whales (*Globicephala macrorhynchus*) (Jensen et al. 2009), Blainville’s beaked whales (*Mesoplodon densirostris*) (Ward et al. 2008), Cuvier’s beaked whales (*Ziphius cavirostris*) (Aguilar Soto et al. 2006), killer whales (*Orcinus orca*) (Houghton et al. 2015) and humpback whales (*Megaptera novaeangliae*) (Blair et al. 2016) since its development in 2003. It combines tri-axial accelerometers, magnetometers, pressure sensors and hydrophones to examine orientation, fluke stroke rate, depth, temperature and sound level simultaneously (Johnson and Tyack 2003; Johnson et al. 2009). Specifically, it has been used to assess behavioural responses of North Atlantic right whales to passing ships (Nowacek 2004), examine the relationship between observed vessel traffic and noise levels received by killer whales (Houghton et al. 2015), measure changes in the foraging dives of humpback whales in response to shipping noise (Blair et al. 2016) and investigate the response of a Cuvier’s beaked whale to noise from passing ships (Aguilar Soto et al. 2006). The recent
development of a DTAG on pinnipeds provides exciting opportunities to study the chronic exposure of pinnipeds to shipping noise and explore at-sea changes in behaviour, particularly diving behaviour, which is difficult to observe from the surface (Mikkelsen et al. 2019).

The DTAG is a very useful tool. However, it can be difficult to attach and can only be deployed for a short length of time, which limits the sample they record (Jensen et al. 2009). For example, DTAG deployments are generally between 0.75 and 22 hours (Johnson and Tyack 2003; Houghton et al. 2015; Blair et al. 2016). Usually, the whale is pursued by a boat for tag deployment and in some cases followed for tag retrieval, potentially influencing the behaviour of the subject. In contrast, pinnipeds are very good potential targets for tagging (Carter et al. 2016) and newer DTAG models (DTAG-3/DTAG-4) allowed for 21 days of recording (Mikkelsen et al. 2019). Pinnipeds haul-out on land at regular intervals creating a relatively easy environment for tag deployment and retrieval. Their fur is also a relatively easy surface to attach tags and they moult regularly simplifying tag removal. Tags are deployed for months rather than hours. In this time animals leave human contact and undertake what could be considered more undisturbed and biologically relevant behaviours. However, it is very difficult to retrieve these tags and hence any stored data. Perhaps the key limiting factor for acoustic tags is the volume of data that acoustic recording generates. Transmitting data via satellite systems or the Global System for Mobile Communication (GSM) can be difficult especially when considering the limited message size of these systems and the battery life of tags (Tomkiewicz et al. 2010).

The data are also subject to interference from extraneous sources such as flow noise past the hydrophone as the animal moves, noise when the animal breaks the surface to breathe and vocalisations (Jensen et al. 2009; Benda-Beckmann et al. 2016). Data must be carefully cleaned, which can be a difficult process. If amplitude thresholds are applied to detect signals within noise levels, recordings can be biased to louder signals that are only detectable above ambient noise.
levels (Merchant et al. 2014b). A major issue in relation to shipping noise is contamination of the recordings by flow noise because it occurs at low frequencies similar to noise emitted by ships (Merchant et al. 2014a). Flow past the hydrophone as the animal swims creates turbulent pressure variations detected by the hydrophone as low frequency noise. Merchant et al. (2014a) suggests that a tags ability to record low amplitude and low frequency noise will always be limited by the issue of flow noise. This echoed early assertions by Burgess et al. (1998) that found flow noise to be a fundamental limit on the potential of their acoustic tags especially when animals are swimming at high speed. However, Benda-Beckmann et al. (2016) recently proposed a data analysis technique that was able to separate the relative contributions of flow and ambient noise which may help with analysis of this type of data in the future, and shipping noise has been extracted successfully from tag recordings (Mikkelsen et al. 2019).

2.4.2 Modelling underwater noise from shipping

Modelling noise from shipping is a multi-stage process, which is represented by the classic equation shown in Equation 2.1, where received level (RL) equals the source level (SL) minus the propagation loss (PL). This equation is a simple and high level representation of a very complex process. SL describes the acoustic level of a sound source reported using sound pressure level in decibels. PL refers to the loss in acoustic intensity with increasing range from the source, and RL refers to the level of an acoustic quantity at a particular point in space. Section 2.4.2.2 describes the characterisation of SL, Section 2.4.2.3 discusses the different techniques that can be used to calculate PL but firstly, Section 2.4.2.1 will describe a variety of methods by which the location of ships at sea can be determined.

\[ RL = SL - PL \]  (2.1)
2.4.2.1 Identifying the location of ships at sea

In order to predict shipping noise on any scale, from local dose-response studies to global ship noise maps, it is necessary to obtain data on the location of ships at sea, their destination, route and how fast they are travelling. Collecting data using observations involves observers recording ship counts and ship type, or tracking ships using theodolite instruments (Williams et al. 2014; Culloch et al. 2016). This is an intensive process undertaken either from land or sea. It is severely limited in its spatial scope but does offer the opportunity to capture, at a fine scale, every vessel in an area, and lends itself well to studies looking at marine mammal responses using land observation where the data can be collected concurrently (Culloch et al. 2016). In light of its limited spatial range and time consuming nature, the use of data obtained from the Automatic Identification System (AIS) aboard large ships has gained real traction over the past few years, and is fast becoming the standard dataset from which to obtain information on ship movements and density (Svanberg et al. 2019). It is required by the Safety of Life at Sea (SOLAS) convention that all ships greater than 300 gross tonnes on international voyages, all cargo ships greater than 500 gross tonnes and all passenger ships report their location at sea (IMO 1974). This is mainly in the service of preventing collisions. However, these reports have also been recorded by commercial companies and are now widely available at a significant cost. The data identifies individual ships and reports their latitude and longitude co-ordinates, at temporal resolutions between 2 and 60 minutes. It will also record ship length and speed as well as current status (e.g. moored). Commercial companies have stored this information in databases, accruing an historical record of shipping across the world. The main AIS data providers generally have data commencing in 2009 and its use within the literature has increased rapidly with just 15 papers published in 2012 and over 50 in 2017 (Svanberg et al. 2019). This has provided a rich dataset from which to understand the movement and density of ships as well as predict shipping noise, most commonly in a particular region, but also on a global scale (Eriksen et al.
This is not to say, however, that AIS does not have its limitations. It has some key characteristics that must be considered when making conclusions based on the data. Primarily, that it only contains ships that are greater than 300 tonnes (and any voluntary smaller participants). Generally, smaller ships are, therefore, not considered but may still make a significant contribution to noise levels in an area (Hermannsen et al. 2016). This may be particularly prominent in coastal areas that are dominated by small recreational and fishing boats rather than commercial shipping or offshore industry (Hermannsen et al. 2016).

Furthermore, in the past AIS used land-based VHF receivers. Due to the curvature of the earth this limited the detection range to between 75 and 110 km (Eriksen et al. 2010). With the development of satellite AIS, coverage is improving (Eriksen et al. 2010). Finally, the information submitted by each ship is open to manual alteration and hence there may be several data errors or inconsistencies (Harati-Mokhtari et al. 2007). As a result, careful data cleaning is required before the data can be used.

These approaches provide information about the ships at a location but they say little about the amount of underwater noise they generate. As mentioned above, this requires a number of further steps. Firstly, characterisation of the ship as a noise source, secondly, propagation of that sound away from the source and finally, calculation of the received level of noise heard by the species of interest.

### 2.4.2.2 Ship noise source models

There are two primary methods to characterise the underwater radiated noise of a ship. The use of a ship source model or the use of data from sound level measurements at sea. When considering shipping noise, defining the source characteristics which are to be propagated is particularly difficult because of the immense range of different ships in the international fleet (McKenna et al. 2012). They differ widely in size, shape and propeller design, and could be travelling at a range of different speeds, all of which determine the source levels and
dominance of different frequencies (Arveson and Vendittis 2000; McKenna et al. 2012). The most enduring model of ship source spectra is that developed by Ross (1976) and subsequently adapted in the Research Ambient Noise Directionality (RANDI) model (Breeding et al. 1996). It is simple and parametrised by data that is readily available, ship length and ship speed. This model is given in Equation 2.2 where \( f \) is the frequency in hertz, \( v \) is the ship’s speed in knots and \( l_s \) is its length in feet. Breeding et al. (1996) added additional length dependent corrections \( df \) and \( dl \). \( L_{so} \) is considered a reference spectrum for a typical ship travelling at 12 knots and 300 feet long.

\[
L_s(f, v, l_s) = L_{so}(f) + 60\log\left(\frac{v}{12}\right) + 20\log\left(\frac{l_s}{300}\right) + df \times dl + 3.0 \quad (2.2)
\]

It is based on the relationship between source level, ship speed and length. However, Wales and Heitmeyer (2002) reported, using experimental recordings of 272 ships, that errors in their dataset compared to RANDI could be as high as 10 dB for 25% of ships. An ensemble statistical model of their dataset reported lower errors (Wales and Heitmeyer 2002). Validation of the model using a separate dataset found it overestimated ship source level by 1.7 dB but underestimated noise by 3.9 dB when an additional speed dependence term, similar to the RANDI model, was included (Brooker et al. 2015). In contrast, median error between the RANDI model and measured data of 57 merchant ships in the East China Sea was 0 (± 7.1) dB, demonstrating the high levels of uncertainty associated with ship source models.

A mechanistic but more complex approach was taken by Wittekind (2014) and Audoly et al. (2017), and considers three specific contributing factors to overall ship noise.

1. Low frequency noise from propeller cavitation \((F_1)\)

2. Medium to high frequency propeller cavitation \((F_2)\)

3. Medium frequency noise from four stroke diesel engines \((F_3)\)
The contribution of each factor is summed as given in Equation 2.3 to find the source level of the ship.

\[
SL = 10 \log\left(10^{\frac{F_1}{10}} + 10^{\frac{F_2}{10}} + 10^{\frac{F_3}{10}}\right)
\]  

(2.3)

\(F_1\) to \(F_3\) are contingent on a complex range of parameters. These are displacement, speed relative to cavitation inception speed, block coefficient, mass of diesel engines and if the diesel engine is resiliently mounted (Wittekind 2014). However, these parameters are not easily known for the majority of ships limiting but not entirely preventing its practical implementation (Audoly et al. 2017; Jalkanen et al. 2018).

The ship source spectrum is a fundamental requirement for modelling ship noise and yet, as discussed above, it remains one of great uncertainty. Therefore, it is important to critically evaluate a model's assumptions and the output of each ship noise model in relation to the overall purpose of the modelling task and report the uncertainties associated with the model so they can be properly considered by regulatory authorities. Ship noise modelling would be improved by a richer dataset of standardised experimental source spectra obtained from recordings in the field. This would not only widen understanding of ship noise spectra but also provide balanced validation datasets. The Ross, RANDI and ensemble models are based on measured data of samples of the commercial fleet (Ross 1976; Breeding et al. 1996; Wales and Heitmeyer 2002). The disagreement in validation could in part be attributed to differences between training and validation datasets of the model. Early data contains ships with fixed pitch propellers but controllable pitch propellers now have a 35% market share (Carlton 2019). Controllable pitch propellers cavitate more when off design pitch, which is usually at lower speeds (Carlton 2019). As a result, they can be noisier at lower speeds than fixed pitch propellers (Carlton 2019). The number of projects measuring underwater noise from ships has increased (Kipple 2002; Allen et al. 2012a; McKenna et al. 2012; Brooker and Humphrey 49
2016; Simard et al. 2016; Audoly et al. 2017; Jansen and Jong 2017; Gassmann et al. 2017; Peng et al. 2018). However, datasets are often recordings of opportunity resulting in high levels of variation in the environment, and methodological standards still need to be refined to address issues with propagation loss and surface interactions (Gassmann et al. 2017). These complicate the use of measurements as validation datasets but also as input values for propagation models which require monopole source values as input (McKenna et al. 2012; Brooker and Humphrey 2016; Simard et al. 2016; Gassmann et al. 2017; Peng et al. 2018). If models are to evolve away from simple statistical models, like those described by Wales and Heitmeyer (2002), to more mechanistic and potentially much more realistic approaches such as Wittekind (2014) data regarding ship design and operational characteristics would need to be more widely available.

### 2.4.2.3 Underwater acoustic propagation models

At the inception of acoustic propagation modelling, the field was primarily driven by the desire of defence organisations to use passive and active acoustic systems to detect and track enemy ships and submarines in the ocean (Lurton 2002). Today, the field draws proponents from many more industries, including ecological stakeholders that are concerned with assessing the impacts of noise on marine life. A detailed history of acoustic propagation modelling can be found in a series of texts by Etter (Etter 2001; Etter 2013; Etter 2018). The application of acoustic propagation modelling to ecological questions is well established but continues to evolve in light of increasing regulation and the need for quantitative environmental risk assessments (Farcas et al. 2016). The application of these models to environmental impact assessments has recently been reviewed by Farcas et al. (2016). They provided a useful introduction to the principles of acoustic modelling in this context. Simply, acoustic propagation models predict the loss in acoustic intensity of a sound wave as it travels through seawater (Etter 2013). The use of a propagation model allows the prediction of sound
levels at a specified distance from the sound source. As a result, in the context of marine mammals, it is useful to predict received noise levels at the location of the animal from many different sources such as ships. The loss in acoustic intensity is accumulated through geometric spreading and absorption via the viscosity of the seawater, relaxation of Magnesium Sulphate (MgSO$_4$) above 100 kHz and relaxation of Boric acid (B(OH)$_3$) above 1 kHz. (Lurton 2002).

Acoustic wave propagation is heavily determined by the physical properties of the ocean environment and the absorption or reflection of waves at boundaries between the water and the surface, the water and the sea floor and different water masses in the ocean (Etter 2013). The relationship between propagation, ocean boundaries and key properties such as temperature, salinity and depth are well resolved and are detailed by Urick (1983), Lurton (2002), and Etter (2013). In summary, temperature, salinity and depth are key determinants of sound speed in the ocean. As each property increases sound speed increases. Figure 2.3 shows an idealised sound speed profile. Initially, as temperature is constant, sound speed increases with depth (A). As temperature decreases, sound speed decreases despite increasing depth (B). When temperature reaches a constant value, sound speed increases with increasing depth as pressure increases (C). Sound speed also increases with increasing salinity but the impact is less pronounced than temperature and depth. The interplay between these factors results in a sound speed minimum (Fig. 2.3). This is typically near 1000 metres at mid-latitudes but can be much closer to the surface at the poles. These different sound speed boundaries, along with the sea surface and sea floor, have a strong influence on wave propagation through reflection, scattering and absorption (Lurton 2002).
Figure 2.3: An idealised sound speed profile in the ocean. Sound speed increases with increasing pressure at depth and decreases with decreasing temperature.

The sea surface reflects and scatters sound, and the sea surface roughness is a key determinant of the amount of loss experienced at the sea surface due to these factors (Etter 2013). When wind speed is low, wave height is low and so are resulting losses because a flat sea surface is a very good reflector (Etter 2013). The sea floor is similar to the surface in that it is a reflective and scattering boundary. However, sound is also transmitted further into the seabed and may be reflected from boundaries or attenuated in the sediment (Etter 2013). Therefore, it is important to understand the key properties of the different sedimentary layers. Specifically, the sediment density, sound speed and attenuation of the acoustic wave as it travels through the sediment. It is difficult to determine these properties and it is most commonly achieved using a geoacoustic model (Hamilton 1980; Etter 2013). However, there is great uncertainty associated with geoacoustic models because of a lack of empirical measurements of sediment acoustic properties. This is one of the key weaknesses in acoustic propagation models (Etter 2013; Farcas et al. 2016). In shallow water environments, bathymetry is also a key factor in sound propagation because of repeated reflections between the sea floor and sea surface, the direction of which is determined by the underlying topography of the sea floor (Etter 2013).
There are many different acoustic propagation models and each are suited to a particular application as shown in Table 2.1. The factors that determine which model should be used for a particular task are usually the frequency range required, whether the problem is in a deep water or shallow water setting and the computational power available (Etter 2013). The simplest representation of propagation loss is calculated using spreading laws. Propagation is assumed to radiate in all directions from a point source in a homogeneous and infinite medium (Lurton 2002). This is termed spherical spreading and is given by the equation $PL = 20 \log(r)$ where $r$ is range from source (Lurton 2002). Cylindrical spreading is similar but accounts for the defined height of the water column due to the sea surface and sea floor and is given by $PL = 10 \log(r)$ (Lurton 2002). These laws are very quick to calculate and are often used with an additional frequency dependent attenuation correction (Jones et al. 2017). These models produce a quick and reasonable estimation of propagation loss under certain conditions and hence have been popular for estimating propagation loss studies looking at marine mammals and shipping noise (Erbe et al. 2014; Jones et al. 2017). However, in shallow water environments, where there are spatial variations in the sound speed profile and multiple propagation paths a more complex representation of propagation is required (Lurton 2002).

Sophisticated numerical models are based on the wave equation derived from standard physical laws. The wave equation written in terms of pressure is given in Equation 2.4, where $c$ is sound speed, $P$ is acoustic pressure and $\nabla$ represents the Laplacian operator. The equation relates temporal variations in pressure (left side of Equation 2.4) with the spatial differences in the surrounding pressure field (right side of Equation 2.4) (Zimmer 2011). Each model starts to diverge significantly from another in the way it deals with solving this equation. As a result there are five broad classes of acoustic model (Tbl. 2.1). A detailed description of each family of models and the available implementations can be found in Etter (2013) (Chapter 4 p.103). For the calculation of shipping noise with respect to marine mammals, models that are suitable for the low
Table 2.1: The conditions under which each class of acoustic model is appropriate. A cross indicates that the model is not suitable, a tick indicates that the model is appropriate and computationally tractable, and a forward slash indicates that the model has limitations either in terms of accuracy or computational tractability. Low frequency sounds are those below 500 Hz and shallow water areas are generally considered to be less than 200 m. RI are range independent models and RD are range dependent models. RD is appropriate for areas of high variability in environmental parameters (e.g. temperature). Adapted from: Etter (2013)

<table>
<thead>
<tr>
<th>Model</th>
<th>Applications</th>
<th>Shallow Water</th>
<th>Deep Water</th>
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<td>Low Frequency</td>
<td>High Frequency</td>
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<tr>
<td>Ray Theory</td>
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<td>Normal Mode</td>
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<td>Fast Field</td>
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frequencies of shipping noise are the most relevant (Etter 2013). The choice between shallow and deep water models will depend on whether the species of interest occupies deep water or coastal zones, and in what zone a species comes into contact with the highest levels of shipping traffic, which is often the coastal shallow water zones (Fig. 2.1).

\[
\frac{d^2}{dt^2} P = c^2 \nabla^2 P
\] (2.4)

For marine mammal and shipping noise research, underwater acoustic propagation models are used to calculate ship source levels from hydrophone recordings (Simard et al. 2016; Peng et al. 2018), calculate the source level of marine mammals calls (Pirotta et al. 2012), create maps of shipping noise (Soares et al. 2012; Erbe et al. 2014; Maglio et al. 2015; Marine Management Organisation 2015; Sertlek et al. 2016; Sertlek et al. 2019) and calculate the exposure of marine mammals to shipping noise (Hatch et al. 2008; Chen et al. 2017; Jones et al. 2017). The most popular acoustic propagation models are the simple geometric spreading laws (McDonald et al. 2001; Hatch et al. 2008; Pirotta et al. 2012; Erbe et al. 2014; Marine Management Organisation 2015),
parabolic equation range dependent models such as RAM (Williams et al. 2014; Farcas et al. 2016; Audoly et al. 2017; Aulanier et al. 2017) or normal mode models such as KRAKEN (Simard et al. 2010; Soares et al. 2012; Porter and Henderson 2013; Maglio et al. 2015; Peng et al. 2018). Modelling of underwater noise can be complex but a number of systems generate a user interface and offer a selection of propagation models for specific problems in marine mammal research (Gisiner et al. 2006; Maggi and Duncan 2015; Ellison et al. 2016). Simple geometric spreading laws are very quick to execute, and therefore, are simple and efficient to use when there are many noise sources each with a different spatial relationship to a receiver (Marine Management Organisation 2015). However, they have been shown to underestimate noise close to the source and overestimate noise far away from a noise source (Farcas et al. 2016). This has implications for those studies that rely on this type of approach. The laws ignore the complex environmental parameters and interactions with boundaries within the water column, which are highly influential, when estimating noise (Shapiro et al. 2014). In contrast, range dependent models such as RAM, produce more accurate representations of underwater acoustic propagation but they are much more complex to execute and take much more time, expertise, computational power and input data with which to parameterise models (Marine Management Organisation 2015). It is necessary to address each of these issues to make the most accurate propagation models practical for implementation in research and policy settings. This will require further research, data collection and collaboration between specialists in biological and physical research. Furthermore, the production of three-dimensional sound fields require the calculation of multiple single frequency two-dimensional (range, depth) transects at different azimuths from a noise source. However, these do not account for horizontal refractions by features such as seamounts and canyons. There are models available to achieve this but they add another level of complexity to propagation calculations and generally only require consideration under certain scenarios such as propagation around complex
2.5 How does shipping noise impact marine mammals?

Marine mammals are particularly vulnerable to shipping noise because of the central role sound plays in foraging and communication, and the close spatial overlap between productive shelf waters and shipping traffic. As shown in Figure 2.4 the peak in the sound pressure levels of ship noise is well within the hearing range of low frequency cetaceans such as the humpback whale (*Megaptera novaeangliae*) and phocid seals such as the grey seal (*Halichoerus grypus*). The low frequency hearing of odontocetes is less sensitive but shipping noise extends to higher frequencies at which they hear well (Fig. 2.4) (Hermannsen et al. 2014; Veirs et al. 2015). This section will consider the research that has examined the impact of shipping noise on four taxonomic groups of marine mammal - pinnipeds, mysticetes, odontocetes and sirenians. Sea otters (*Enhydra lutris*) and polar bears (*Ursus maritimus*) are not considered here because there are very few studies looking at the relationship of these groups to shipping noise.

![Figure 2.4: Predicted 1/3 octave band ship source levels and the composite audiograms for the functional hearing range of high frequency cetaceans, low frequency cetaceans and phocid seals (National Marine Fisheries Service 2018).](image-url)
2.5.1 Pinnipeds

Pinnipeds are amphibious central place foragers that go to sea to feed before returning to haul-outs on land or ice to rest, breed, pup and moult (Thompson et al. 1991; McConnell et al. 1999). As a result, they have sensory systems optimised for sound production and reception in both air and water (Au and Hastings 2008). The functional hearing range of phocid pinnipeds is between 50 Hz and 86 kHz, and between 60 Hz and 39 kHz for otariid pinnipeds (National Marine Fisheries Service 2018). Pinniped calls have been recorded in-air at haul-outs as well as under the water, and are closely associated with mating (Van Parijs et al. 2001; Van Parijs and Kovacs 2002; Attard et al. 2010), sociality (Asselin et al. 1993), mother-offspring interactions (Charrier et al. 2010) and territoriality (Tripovich et al. 2005). Masking is the process by which the presence of noise can reduce the ability of an animal to detect a signal such as a call from a conspecific (Erbe et al. 2015). This can be measured by an increase in the auditory detection threshold (Box 2.4) or a decrease in the area over which an animal can communicate with conspecifics (Erbe et al. 2015). This area is termed the communication space (Clark et al. 2009). Bagočius (2014) examined the potential coincidence of noise from ships and grey seal vocalisations in the Lithuanian region of the Baltic Sea. The study showed that the calls of grey seals (Halichoerus grypus) were potentially masked by ships, and that the distance over which seals could communicate was potentially reduced. Furthermore, the loudness of harp seal (Pagophilus groenlandicus) calls were reduced after the presence of a boat. The seals could have reduced their vocalisations or have moved away from the area (Terhune et al. 1979). There is still a limited understanding of the impact of shipping noise on vocalisations and communication in the majority of pinniped species, otariid or phocid.

Literature on the interaction of vessel traffic and pinnipeds has focussed on the approach of ships to seals at haul-out locations rather than responses to underwater noise. These responses are easily observable and studies have arisen out of concern for disturbance by tourism rather than noise (Stafford-Bell
et al. 2012). Common reactions of pinnipeds to approaching vessels include increased alertness (Henry and Hammill 2001), flushing off haul-out sites into the sea (Jansen et al. 2010; Andersen et al. 2012; Blundell and Pendleton 2015) and head raising (Niemi et al. 2013). Such studies have examined a range of species including harbour seals (*Phoca vitulina*) (Blundell and Pendleton 2015), Saimaa ringed seals (*Phoca hispida saimensis*) (Niemi et al. 2013), Australian fur seals (*Arctocephalus pusillus doriferus*) (Stafford-Bell et al. 2012) and walrus (*Odobenus rosmarus*) (Øren et al. 2018). These responses are generally linked to the in-air sound, the sight and possibly the smell of the approaching ship. Tripovich et al. (2012) examined the behavioural responses of Australian fur seals pre-, during and post- stimulus responses to playback experiments of motor boat noise at a haul-out without the presence of the boat. During the stimulus phase there was a decrease in resting behaviour and an increase in alert or fighting behaviour. However, the perception of sound and the propagation of sound in-air is different than underwater (Dahl et al. 2007; Badino et al. 2012).

The literature on the impacts of underwater noise from shipping on pinnipeds is more limited. There are studies related to only three of the 33 recognised species of pinniped and these are all focussed in the seas of Europe and eastern Canada (Committee on Taxonomy 2018). An increase in shipping traffic was negatively correlated with the presence of grey seals in Broadhaven Bay, Ireland (Anderwald et al. 2013). At a broader spatial scale, the co-occurrence of seals and shipping traffic around the UK has been shown to be high, particularly within 50 km of the coastline (Jones et al. 2017). The study by Jones et al. (2017) suggested high overlap between harbour seals and shipping in the Moray Firth, and this was reflected in the predicted exposure of the seals. M-weighted 24-hr cumulative sound exposure levels were between 170.2 (95% CI 168.4 - 171.9) and 189.3 (95% CI 172.6 - 206.0) dB re $1\mu Pa^2s$. Only when considering the upper bounds of the 95% confidence intervals did the animals potentially experience temporary threshold shift (Jones et al. 2017). However, it is still not
known if there are any behavioural consequences of such exposure and what these consequences mean for the success of the population as a whole. Initial observations, that show changes in the ascent and descent phases of dives by harbour and grey seals, suggest that more comprehensive investigation is warranted (Mikkelsen et al. 2019). This is particularly relevant in variable oceanographic environments where diving animals will potentially experience step changes in broadband exposure levels of approximately 10 dB due to a strong thermocline and fronts (Chen et al. 2017).

**Box 2.4: Marine Mammal Audiograms and Auditory Damage**

Audiograms are a representation of how well an animal can hear. For each frequency it shows the minimum amplitude of the sound required for the animal to hear it. The lower the amplitude required the more sensitive the hearing of the animal at that frequency. Audiograms for marine mammals are usually obtained using either behavioural training tasks, where captive animals learn to display a behaviour if they can hear sounds of different amplitude played to them, or using electro-physiological techniques, where electrical pulses from neural activity are recorded after an acoustic stimuli (André and Nachtigall 2007). Auditory evoked potentials may be considered more robust because it does not rely on a conditioned behavioural response (André and Nachtigall 2007). However, it does not account for the perception of the animal which may be different from the theoretical biological ability of the animal to hear (Finneran and Branstetter 2013). Behavioural audiograms can account for this but are complicated by the need for scientists to interpret when an animal actually hears a sound (Finneran 2015). The nature of the techniques used to obtain audiograms means they are usually restricted to captive animals resulting in bias towards small animals and to species that can be appropriately be kept in captivity (Finneran 2015). It also results in very small sample sizes. Auditory damage as a result of noise can be assessed by looking at changes in the the threshold amplitudes required for an animal to hear at a certain frequency. If this threshold increases after noise exposure the sensitivity of the animal to that frequency has been reduced and there has been some damage to the structures of the ear (Kastak et al. 2005). This can happen for a short amount of time after which the hearing thresholds return to normal, this is termed temporary threshold shift (TTS) or this change can be irreversible, which is termed permanent threshold shift (PTS) (Southall et al. 2007).
2.5.2 Mysticetes

The primary driver of interest in baleen whales and shipping noise is the predicted overlap between the dominant ship noise frequencies and low frequency calls emitted by baleen whales (Southall et al. 2007). It is very difficult to determine the hearing of baleen whales because their size makes captivity and hence audiogram studies prohibitive (Southall et al. 2007). Therefore, the majority of knowledge is inferred from examination and modelling of the hearing physiology of stranded whales (Yamato et al. 2008; Cranford and Krysl 2015), behavioural responses of whales to sounds in the wild (Dahlheim and Ljungblad 1990; Deecke 2006), and the use of call frequency as a proxy for hearing (Au and Hastings 2008). While, there is some debate on the applicability of call frequency as a proxy for hearing ability (Au and Hastings 2008; Luther and Wiley 2009), the functional hearing range of baleen whales is thought to lie between 7 Hz and 35 kHz (Southall et al. 2007; National Marine Fisheries Service 2018). This is firmly within the peak spectral power of shipping noise (Fig. 2.4).

The research that deals with shipping noise and baleen whales has drawn variable and complex conclusions from studies of 6 out of the 14 known species mainly based in North America (Committee on Taxonomy 2018). Baleen whales are believed to make calls associated primarily with communication between individuals and groups, as well as social calls for mate choice and attraction (Au and Hastings 2008; Parks et al. 2011). However, the rising low frequency noise levels in the ocean may impede the ability of whales to communicate. Hatch et al. (2012) modelled the available communication space of the endangered North Atlantic right whale (*Eubalaena glacialis*) at Stellwagen Bank. Their calling rate and source level are relatively low when compared to other baleen whales (Hatch et al. 2012). In this study they measured a calling rate of 0.47 calls per minute at an average source level of 172 dB re $1 \mu P a$ (71-224 Hz). They estimated that right whales in this region had experienced a 63-67% loss in acoustic communication space when compared to historic ambient noise levels in the mid 20th century. Further exploration at the same location found that fin
(Balaenoptera physalus), humpback (Megaptera novaeangliae) and minke (Balaenoptera acutorostrata) whales experienced a loss in communication space of over 80% due to shipping noise (Cholewiak et al. 2018).

In addition, Simard et al. (2008) examined the detection of whale calls by passive acoustic hydrophones in the St. Lawrence-Saguenay marine park. They suggest that most baleen whale calls are masked by shipping noise beyond approximately 60 km in that area. Blue whales calls are classified based on structure. D-calls are a short downsweep call from 60 - 45 Hz (McDonald et al. 2001). Simard et al. (2008) suggest that these calls are at particular risk of being masked because they are naturally lower in source level and coincide with the peak spectral frequency of shipping noise. The probability of detection was close to zero for these calls at ranges greater than 5 km due to low frequency ambient noise. The aim of this study was to assess the ability of passive acoustic monitoring systems to detect whale calls under ambient noise conditions. Nevertheless, it highlights the challenges that individuals calling, and their conspecifics receiving calls, are likely to encounter under high levels of ship noise.

The consequences of communication masking in marine mammals are not well understood but possible impacts include an increase in energetic output to make louder calls or a decrease in an individuals ability to select a mate (Erbe et al. 2015). These could have implications for an individuals fitness and population success as a whole (Erbe et al. 2015). For baleen whales, a number of studies suggest that whales can change their calls to mitigate against the loss in communication space experienced under high noise levels. The conclusions of these studies are highly variable. Blue whales (Balaenoptera musculus), grey whales (Eschrichtius robustus) and North Atlantic right whales are thought to increase call amplitude in response to low frequency noise (Parks et al. 2011; Melcón et al. 2012; Dahlheim and Castellote 2016). Grey whales also increased calling rate (Dahlheim and Castellote 2016), and there was a significant negative correlation between the duration, bandwidth, fundamental frequency, centre
frequency and peak frequency of fin whale 20 Hz song notes with increasing ship noise levels (10 - 585 Hz) (Castellote et al. 2012). Parks et al. (2007) also report a change in the frequency of North Atlantic right whale calls. However, they report an increase in the fundamental frequency and a decrease in call rate.

Dunlop (2016) found no change in the amplitude of humpback whale calls in the presence of shipping noise. The humpback whale calls measured used a wider range of frequencies (15 Hz - 4 kHz) than the fin whale (20 Hz) and right whale calls (50 - 400 Hz) studied above. Dunlop (2016) also found no change in the frequency or duration of two humpback vocal sounds. Furthermore, whales appeared to stop singing rather than shifting frequency when a single passenger ship was present (Tsujii et al. 2018), and the probability of calling was 31 - 45 % lower when vessel noise contributed to the soundscape (Fournet et al. 2018). In contrast, humpback whales encountering noise from cruise and tour boats increased source level of calls 0.81 dB for every 10 dB increase in ambient sound levels, and a separate study in the same area found humpback whales increased the rate and repetitiveness of feeding calls (Doyle et al. 2008). The complexity of whale vocalisations and variability among individuals and species make it difficult to elucidate the impact shipping noise has on the calls of baleen whales. The evidence to date suggests that there is not an overarching strategy among whales for tackling shipping noise, and particularly, there may be different responses depending on the function of the calls. However, it appears individuals are exhibiting plasticity in their calls in response to communication masking.

Observed behavioural responses of baleen whales to vessels include avoidance (Stamation et al. 2009), investigative interactions (Dahlheim et al. 1981; Stamation et al. 2009) and an increase in dive times (Stamation et al. 2009). Specifically, humpback whale mothers increased linearity and mean speed of movement, and decreased blow intervals and time spent resting in response to vessels (Morete et al. 2007). Calves also spent less time rolling and resting (Morete et al. 2007) when ships were nearby. North Atlantic right whales tend to show no behavioural response to approaching ships either due to noise or to
indicators of its physical presence making them vulnerable to ship strike (Nowacek 2004). This may be due to habituation to shipping noise as a result of chronic exposure or the pattern of radiated noise from ships (Allen et al. 2012a). When active acoustic devices are used to alert whales to approaching ships, the most common response is to swim rapidly to the surface, placing them in more danger of ship strike (Nowacek 2004). In contrast, Blair et al. (2016) reported changes in humpback whale foraging behaviour. They found that when ship noise was high the descent rate of feeding dives decreased and there were fewer side-roll feeding events per dive. Behavioural responses such as these, like call masking, may have implications for individual fitness but work is ongoing to quantify this and what it means for the population as a whole (Harwood et al. 2016).

There are few studies examining the impact of shipping noise on the physiological state of marine mammals. A unique study by Rolland et al. (2012) assessed the incidence of chronic stress caused by shipping noise in North Atlantic right whales. The September 11th terrorist attacks in 2001 resulted in a decrease in noise levels in the Bay of Fundy due to the interruption of shipping in the area. This coincided with a decrease in baseline faecal glucocorticoid levels in the North Atlantic right whale. This may have important welfare consequences for the individual whale as well as population level consequences. Glucocorticoids are steroid hormones associated with energy balance (Blas et al. 2007). High glucocorticoid levels result from activation of the hypothalamic-pituitary-adrenal axis that regulates stress. It is important for strategies such as flight from predators but extended periods of activation can result in negative consequences for survival and reproduction (Sheriff et al. 2009; Rolland et al. 2012). The physiological consequences of noise remain understudied due to the methodological difficulties of experimentally manipulating long-term noise exposure and collecting physiological data from free-swimming large whales (Rolland et al. 2012). However, it remains a relevant concern across all taxonomic groups.
2.5.3 Odontocetes

The odontocetes can be broken down into two groups based on their hearing. Mid-frequency cetaceans such as bottlenose dolphins (*Tursiops truncatus*), killer whales and the pacific white-sided dolphin are thought to have a functional hearing range from 150 Hz to 160 kHz (Southall et al. 2007; National Marine Fisheries Service 2018) (Fig. 2.4). High frequency cetaceans such as the harbour porpoise (*Phocoena phocoena*) are thought to have a hearing range from 275 Hz to 160 kHz (Southall et al. 2007; National Marine Fisheries Service 2018). Historically, toothed whales were considered more robust than baleen whales to shipping noise because they have functional hearing ranges and vocalisations at frequencies above the most dominant low frequencies of shipping noise. The high frequency components of shipping noise attenuate more quickly than low frequencies in the ocean and so have received less attention. However, more recent explorations of shipping noise found raised noise levels (5 - 13 dB) at 10 - 96 kHz over ranges of 3 km from the source ship (Veirs et al. 2015). This was similar to the findings of Hermannsen et al. (2014). They found elevated noise levels from 0.025 - 160 kHz at lesser ranges of 60 - 1000 metres. These studies indicate that shipping noise does extend to frequency ranges used by odontocetes and the impact of this should be considered carefully. Despite being high frequency specialists, many charismatic odontocete species closely overlap with human activities and eco-tourism, hence there has been a number of studies related to the interaction of boats and odontocetes covering 14 of the 75 listed species (Committee on Taxonomy 2018).

In a manner similar to the baleen whales, odontocete vocal behaviour serves several important ecological functions and can be potentially disrupted by shipping noise. Odontocetes make two broad classifications of sounds. All species make pulsed sounds generally between 5 and 150 kHz that can be used for echolocation and communication (Perrin et al. 2009). A subset of these species also make narrowband tonal sounds such as whistles generally between...
5 and 80 kHz (Perrin et al. 2009). As discussed above call masking can be a problem for all marine mammals in the presence of noise. As with baleen whales but to a lesser extent, shipping noise has caused a 26% reduction in the communication space (2 - 12.5 kHz) of bottlenose dolphins and a 58% reduction in the communication space of short-finned pilot whales (Globicephala macrorhynchus) caused by a small vessel travelling 5 knots in shallow and deep water respectively (Jensen et al. 2009). Small vessels emit higher frequency noise than large vessels. However, larger ferries are also predicted to decrease the communication range of beluga whales (Delphinapterus leucas) to less than 30% of the expected range (Gervaise et al. 2012).

In response to loss in communication space individuals may attempt to alter the characteristics of their vocalisations (Buckstaff 2004). Evidence to support this hypothesis has been presented for a number of dolphin species, killer whales (Orcinus orca), sperm whales (Physeter macrocephalus) and beluga whales. Bottlenose dolphins whistled more at the onset of a vessel approach than when no vessels were present (Buckstaff 2004). Short beaked common dolphins (Delphinus delphis) had higher vocalisations in the English Channel than a separate population in the Celtic Sea with less vessel traffic (Ansmann et al. 2007). The whistle duration of Guiana dolphins (Sotalia guianensis) was negatively correlated with noise levels but whistling rate was positively correlated with noise levels (Bittencourt et al. 2017). Minimum, maximum, frequency modulations and peak frequency of bottlenose dolphin whistles had a significant positive relationship with ambient noise levels (Rako Gospić and Picciulin 2016; Ginkel et al. 2018). Similarly, Heiler et al. (2016) found an upward shift in whistle frequency parameters, which was strongly influenced by surface behaviour and group composition. Killer whales increased call amplitude by 1 dB for every dB increase in background noise levels, and the presence of whale watching boats resulted in longer call durations (Foote et al. 2004; Holt et al. 2009; Holt et al. 2011). Sperm whales decreased the number of clicks as ships became closer (Azzara et al. 2013) and beluga whales increased the amplitude of calls as noise
increased (Scheifele et al. 2005). The frequency of beluga whale calls also increased from 3.6 kHz to between 5.2 and 8.8 kHz during exposure to ferry boat and motorboat noise (Lesage and Barrette 1999). It was hypothesised that such a shift in frequency results from a desire to improve signal clarity by avoiding those frequencies that are occupied by ship noise (Lesage and Barrette 1999). Interestingly, the motorboat and ferry have different noise spectra but the belugas moved the frequency of calls independent of ship type and hence avoided shipping noise with respect to the ferry but moved to a frequency with greater levels of motorboat noise (Lesage and Barrette 1999).

The above findings, while highly variable between species, are particularly concerning in light of evidence to suggest performing vocal communication causes an increase of 1.2 to 1.5 times resting metabolic rate in Atlantic bottlenose dolphins and therefore, increases in vocal effort could have a metabolic cost (Holt et al. 2015). If there is a significant metabolic cost to changing vocalisations this will have important consequences for individual fitness.

The behavioural response of odontocetes is also variable but a number of studies across different species suggest that the response seen is often related to behavioural state and other contextual factors. Williams et al. (2014) modelled received noise level for a retrospective analysis of 6 years of killer whale theodolite monitoring data. They found there was an increasing probability of a response (based on a qualitative scale of locomotion speed, direction and respiration rate) from whales as the noise levels increased. Generalised linear models demonstrated that killer whale response was not only explained by noise but also by the number of ships present, year, month and the whales age (Williams et al. 2014). Furthermore, southern resident killer whales spent less time foraging and the probability of switching between behavioural states were significantly impacted by vessel traffic (Lusseau and Bain 2009). Chilean dolphins (Cephalorhynchus eutropia) showed no significant change in swimming speed but changed direction more when foraging after a boat encounter (Ribeiro
et al. 2005). Indo-Pacific bottlenose dolphins (Tursiops aduncus) spent 66.5% less time feeding, 44.2% less time socialising and maintained more cohesive groups when boats were present (Steckenreuter et al. 2012). However, in other studies such behaviour was contingent on group composition. When tour boats were audible mother-calf pair groups were less cohesive and increased the rate of whistling whereas groups without calves whistled less often (Guerra et al. 2014).

Foraging behaviour is central to the survival of individuals and stable populations. Consequently, it is concerning that the literature presents examples of disrupted foraging behaviour across a number of species. A DTAG was used to record the dives of a Cuvier’s beaked whale (Ziphius cavirostris). This provided anecdotal evidence from a single dive with high exposure to ship noise compared to 7 quiet dives. Noise from a passing ship reduced the total dive time and reduced the vocal phase of the dive, which is associated with foraging, to 41% of the total dive time from 60% (Aguilar Soto et al. 2006). Furthermore, beluga whales displayed avoidance behaviour of whale watching boats and terminated foraging behaviour (Blane and Jaakson 1994). Harbour porpoise have exhibited decreased surface feeding behaviour (Akkaya Bas et al. 2017), and undertook fewer prey capture attempts at received noise levels above 96 dB re $1\mu$Pa (Wisniewska et al. 2018). Missed foraging opportunities may have severe consequences for fitness. However, it is not known to what degree these behavioural responses result in biologically relevant changes in foraging (Pirotta et al. 2018).

It is intuitive to expect that animals may avoid shipping traffic if they perceive it as a threat. It is reported that Irrawady dolphins (Orcaella brevirostris) surfaced less in the presence of boats suggesting they were trying to avoid them (Kreb and Rahadi 2004). Indo-Pacific bottlenose dolphins increased average movement speeds in high vessel densities. They spent more time travelling and less time resting or socialising suggesting avoidance of the area (Marley et al. 2017). In addition, vessel noise has impacted the longer-term behaviour of Indo-Pacific
bottlenose dolphins in East Africa. Dolphins were more likely to undertake temporary emigration from the Kisite-Mpunguti marine protected area when boat numbers were high (451 boats); 78% of the population left the study area (Pérez-Jorge et al. 2016). Furthermore, there was a decrease in detections of finless porpoise (*Neophocaena phocaenoides*) with increasing ship numbers (Akamatsu et al. 2008; Dong et al. 2012). Pirotta et al. (2012) detected a change in the behaviour of Blainville’s beaked whales up to 5.2 km from a vessel through the use of hydrophones. This change in hydrophone detections could have been caused by a restriction in group movement, directed travel or decrease in clicking behaviour in the group. The avoidance behaviours reported by these studies may reduce the possibility of ship strike but could be problematic if it results in exclusion from important habitats or interruption of important behaviours.

Dyndo et al. (2015) studied the responses of four harbour porpoises kept in a semi-natural net pen to 133 vessel passages. They report low levels of high frequency noise from passing ships resulted in porpoising behaviour. However, it is difficult to determine if the same stereotyped behaviour would be the response of free-ranging animals. Enclosure in a net pen is likely to reduce the behavioural repertoire available to the animals. It has been shown that bottlenose dolphins become less and less likely to stay in an area as boat noise increases (La Manna et al. 2013). This response was not available to the harbour porpoises because they were constrained within a pen. These studies highlight common methodological problems with the study of shipping noise and marine mammals. Captive studies benefit from carefully controlled exposures and it is simpler to measure received noise levels at the animal. However, they do not allow the display of natural behaviour or exposure to natural patterns of sound.

### 2.5.4 Sirenians

Sirenians are herbivorous marine mammals that primarily inhabit coastal estuaries and wetlands (Gannon et al. 2007; Hieb et al. 2017). Large and slow moving animals, populations are vulnerable to a number of anthropogenic
threats. Ship strike has been a major concern in most regions. It is estimated that 25 - 31% of West Indian manatee (*Trichechus manatus*) deaths are a result of vessel strike (Miksis-Olds et al. 2007). This has catalysed research related to vessel traffic but it has generally been limited to the West Indian manatee. Video recordings suggest that manatees react to vessels by turning towards and retreat to deeper water and increasing swim speed (Nowacek et al. 2004). This is exacerbated by the close approach of boats in shallow water (Nowacek et al. 2004; Miksis-Olds et al. 2007). Elevated noise resulted in a decrease in call rates during feeding and social behaviours (Miksis-Olds and Tyack 2009). Similarly, dugongs (*Dugong dugon*) were less likely to continue feeding after close approaches by boats and increased the number of harmonics in their calls after boat passes (Hodgson and Marsh 2007; Ando-Mizobata et al. 2014). However, there is no information on the Amazonian or African manatees (*Trichechus inunguis, Trichechus senegalensis*) and given the vulnerability of sirenians further research would assist in appropriate conservation.

In summary, each taxonomic group of marine mammals discussed above was sensitive to certain frequencies of the ship noise spectrum. The studies reviewed demonstrate that exposure to shipping noise can impact the auditory sensitivity, communication and behaviour of individuals. There were much fewer studies relating to the physiological impacts on each group of species, and research has focussed on only a few species within each taxonomic group. The studies reveal large variation in responses to noise within and between individuals, and as such internationally agreed targets for shipping noise still depend on a better understanding of exposure, and individual to population level impacts (Van der Graaf et al. 2012; DEFRA 2019).
2.6 The future: potential solutions and required research

2.6.1 How can shipping noise be reduced?

At present, there are solutions available that can be implemented to address shipping noise (IMO 2014; Ebrahimi et al. 2019), and if noise producing sources are removed from the ocean, the noise pollution is also removed without leaving an enduring acoustic footprint. In the Haro Strait (British Columbia, Canada) it is estimated radiated noise levels could be reduced by a half by reducing noise from just the noisiest 15% of the fleet (Veirs et al. 2018). The solutions do, however, require political and industrial will, supported by effective policy, to implement them effectively. Possible solutions include reducing the speed of vessels, retrofitting vessels with quieting technology, changing the design of new ships, relocating shipping lanes, ship maintenance, no-go areas and ships travelling in convoy (Williams et al. 2019).

A reduction in the travel speed of ships can result in the reduction of noise levels because cavitation only begins above a certain cavitation inception speed (Arveson and Vendittis 2000). Williams et al. (2019) estimated a 3 dB reduction in noise levels can be achieved by limiting ships to 11.8 kts. This is a simple solution that can be easily implemented by all ships. However, the issue of speed limit compliance, and the influence of longer transit times on noise exposure must also be considered (Joy et al. 2019). Furthermore, if ships use controllable pitch propellers they may cavitate more at slower speeds and the industry would have to account for slower delivery times (Carlton 2019; Gonyo et al. 2019).

The relocation of shipping lanes and ships travelling in a convoy are also proposed solutions. However, these solutions do not reduce noise levels but redistribute it in time and space (Williams et al. 2019). The relocation of shipping lanes could be beneficial in reducing ship strike but they may need to be moved dramatically to achieve noise reduction targets. In the Haro Strait the reverberations off bordering land limited the noise reductions (Williams et al. 2019). A convoy increases noise levels when all the ships are travelling but then
reduces noise at other times of the day. It is unclear how this pattern would influence marine mammal exposure and behaviour.

The noisiest component of shipping noise is propeller cavitation (Urick 1983). This can be addressed directly by replacing or modifying propellers and ensuring new ships have the quietest possible designs. This would be of benefit to all stakeholders because cavitation can cause damage to propellers and loss in efficiency (IMO 2014). Solutions include regular cleaning and maintenance of propellers, coating propellers, ducted propellers, operating controllable pitch propellers at design settings, increasing number of propeller blades, and changing the shape and design of propellers (IMO 2014; Ebrahimi et al. 2019). Laboratory tests suggest these solutions can result in noise reduction of between 2 and 30 dB (Ebrahimi et al. 2019). These solutions can be retrofitted to older ships but it is a costly solution and the newest innovations are generally brought to new ships (IMO 2014). The average age of the fleet in 2016 was 20.31 years, which demonstrates that it will take many years for the quietest technology to be the norm (UNCTAD 2016). Nevertheless, it remains unclear if the noise reductions achieved through such methods will result in a net increase or net reduction in noise levels when combined with the additional noise associated with a greater number of larger ships. It is also of consideration that when adopting ship noise quieting measures they may influence other ship noise characteristics and conservation issues. For example, it is not only the total broadband noise which is relevant to marine mammals but also the spectral components of that noise (Ellison et al. 2012), and radiated noise patterns from ships can play a role in the detection of ships by whales influencing ship strikes (Allen et al. 2012b). The interaction of ship noise quieting measures and wider conservation issues should be explored further to ensure the development of comprehensive policy (Laist et al. 2014).
2.6.2 Conclusions and future research

This review has summarised the key characteristics of shipping noise, examined its current trends and discussed how shipping noise is measured and predicted. The final sections of this review examined the documented impacts of shipping noise on marine mammals and how shipping noise could be reduced. The review highlighted increasing levels of low frequency noise in the world's oceans and an increasing number of studies that suggest this can have an impact on marine mammals. However, it also highlights that there are many questions still to be answered. The studies presented are limited in their geographic range, species studied and often have limited sample sizes. It is difficult to distinguish between normal and abnormal behavioural responses because the ethogram of many species is understudied, and research often fails to consider contextual factors that influence an individuals response. Future research should consider this when seeking to understand current noise levels in ecologically sensitive areas, how this is set to change in the future, and what this will mean for marine mammals, not just in relation to the individual, but also with respect to how short-term responses such as startling and foraging disturbance will influence population level success. Particularly, this could be more successful if more studies measured or predicted the received noise levels at the animal as well as noise levels in the wider area, and utilised the most accurate propagation models when predicting noise levels.

Future research will take place against a background of increasing regulation and targets set by policy. This will no doubt help to shape the agenda of ship noise research as regulators look to the research community to provide evidence and certainty on the acceptable levels of noise and the best way to mitigate against shipping noise. The majority of legislation that might apply to noise specifies requirements for the protection of species and their habitat rather than setting specific noise regulations (Lucke et al. 2013). For example, in the US the Marine Mammal Protection Act (1972) protects marine mammals from harassment, which could be defined as a noise source. However, in the EU
shipping noise is explicitly under the regulatory control of the Marine Strategy Framework Directive (MSFD). This is the EU regulatory framework to ensure the ‘Good Environmental Status’ (GES) of EU waters by 2020. One descriptor of GES is that underwater noise should not be at levels that adversely affect the marine environment (European Commission 2008). Despite the volume of existing research, member states still find it difficult to recommend target levels of ambient noise that represent GES (Van der Graaf et al. 2012; DEFRA 2019). This stems from a lack of data on current levels of shipping noise and how elevated levels of noise impact not only marine mammals but the ecosystem as a whole. This review has focussed on marine mammals but the MSFD is interested in an ecosystem level approach and as a result, it will be necessary in the future to be able to quantify the impacts of ship noise on many more components of the ecosystem such as fish and invertebrates, and integrate marine mammals into such an assessment (Van der Graaf et al. 2012).

To address the lack of data pertaining to ship noise levels, there are a number of projects that are already in progress to monitor shipping noise and instigate a long-term record of noise levels (Van der Graaf et al. 2012; Merchant et al. 2018; Crawford et al. 2018). However, there needs to be an improved understanding of how noise exposure levels result in problematic consequences. It is not possible to consider mitigation measures until such questions are resolved because the implementation of inappropriate measures can be either costly and damaging for the species it is meant to protect or overly harsh on industrial activity, which may impact economic growth (Merchant 2019). However, there is an emphasis on the use of a precautionary approach until more information is known. The International Maritime Organisation (IMO) has been proactive in the face of criticism of the levels of noise that shipping produces. In 2014 they released non-binding guidelines for the reduction of underwater noise from shipping (IMO 2014). The document briefly outlines the design considerations for new ships and operational and maintenance activities that may help reduce noise from shipping. The IMO is a key stakeholder in the regulation of shipping and it was
an important step for the organisation to recognise the problems associated with shipping noise by issuing the guidelines. However, there is still very little consideration of noise in the IMO’s international conventions, including the International Convention for the Prevention of Pollution from Ships or the new International Code for Ships Operating in Polar Waters. In order to drive the consideration of shipping noise as a pollutant at this level of international co-operation, which is important to address such a global pollutant, there is need for more conclusive evidence on harmful noise levels (Van der Graaf et al. 2012; DEFRA 2019).

The current evidence focuses on the impact of noise on individuals for a limited number of species. However, there is less information available about when behavioural disturbance or auditory damage are biologically significant and may impact vital life functions such as survival, breeding and migration. The Population Consequences of Acoustic Disturbance (PCAD) Model was proposed by the National Research Council (NRC) in the USA. Figure 2.5 shows the PCAD framework, which highlights the pathways through which acoustic disturbance can be traced to population level consequences. It demonstrates, as has this review, that sound can be measured relatively well but less is known regarding behavioural change and the link between this, life functions and vital rates. It is possible to suggest that research should focus on resolving these areas but these are not necessarily easily knowable entities. In response, the interim Population Consequences of Disturbance (iPCoD) model was developed, which uses expert opinion to estimate how behavioural or physiological changes will influence vital rates (Harwood et al. 2016). This may not be as accurate as parametrising the model with real data but it is a good way to start estimating the possible consequences of shipping noise. New et al. (2013) used a similar approach to estimate the consequences of an increase in vessel numbers from 40 per year to 470 per year a year in the Moray Firth on a bottlenose dolphin population in the area. They estimated that there would be no impact on the population as a result of behavioural changes because there was no change in
the health of the individuals. This illustrates how behavioural change or auditory threshold change may not always result in a biologically significant consequence and that this needs to be understood in more detail (Pirotta et al. 2018). This information would help policy-makers set thresholds for ship noise levels.

Furthermore, it is necessary to look at shipping noise in the context of cumulative impacts of noise and other threats. For example, the right whale is listed as an endangered species and is at risk from fishery entanglement and ship strike (Kraus and Rolland 2007) as well as low productive rates from chemical toxins (Reeves et al. 2001). As a result, shipping noise is an additional stressor for this already stressed population potentially magnifying the negative impacts.

**Figure 2.5:** The Population Consequence of Acoustic Disturbance Framework proposed by NRC (2005). The framework suggests how noise that causes behavioural disturbance can be linked to population level change. The ⋆ sign shows how much is known about these factors and how easily they can be measured. ⋆ ⋆ ⋆ is well known and easily observed.

The majority of research looking at the impacts of shipping noise on marine mammals is predicated on the premise that it is known at which frequencies they are able to hear sound. For example, audiograms are used to determine if a species is able to hear shipping noise, set thresholds for what constitutes auditory damage (Southall et al. 2007) and calculate possible masking and behavioural scenarios (Erbe et al. 2015). The reality is that, for many species,
hearing data is not very robust (very small sample sizes or limited frequencies) or has not yet been measured. The current thresholds for auditory damage are grouped into five broad functional hearing groups. It is difficult to calculate behavioural or electrophysiological audiograms because marine mammals are often too large to work with in captivity and many are protected species (Southall et al. 2007). It is desirable to have more data about the hearing of marine mammals but this is not necessarily simple to collect and the limitations that have restricted its development to date are not easily overcome (Southall et al. 2007). The current data that is available does not preclude the assessment of shipping noise and the grouping of species into functional groups can be useful for policy-makers looking to set broad precautionary thresholds (Southall et al. 2007; National Marine Fisheries Service 2018). A more detailed knowledge of the audiograms of a greater range of species is most pertinent for calculating the loss in communication space and the degree of communication masking (Southall et al. 2007). However, it will require the innovation of methodological techniques for measuring auditory hearing thresholds, which may be developed for this specific use or may come from the wider auditory field in mammals or birds (Southall et al. 2007). This type of knowledge will also pave the way for more research on the impacts of noise on hearing including TTS and PTS onset and masking (Southall et al. 2007).

Section 2.3 of this review examined the temporal trends and spatial patterns of shipping noise in the oceans. It suggests that general trends in shipping noise will be determined by the number of ships, the size of these ships and the design of the ship, and that the forces controlling economic development and trade will dictate demand for shipping (Frisk 2012; UNCTAD 2018). However, it is interesting to note that climate change might influence low frequency sound levels in the ocean through ocean acidification. The key mechanism of sound absorption in the ocean is the relaxation of boric acid and magnesium sulphate ions. These are influenced by the pH of the ocean which is becoming increasingly acidic under climate change. It is suggested that under projected
pH changes sound absorption of frequencies between 100 Hz and 10 kHz, which overlap with ship noise, could be significantly reduced (Ilyina et al. 2010). The impacts of this are not well understood nor is the magnitude by which changes in sound absorption will result in changes in the levels of shipping noise, especially because climate changes will be accompanied by changes in water temperature, another key determinant of sound propagation.

This review has presented evidence relating to the impacts of shipping noise. However, it has also highlighted several important possible mitigation measures. Unlike many pollutants which have long residence times in the ocean the cessation of noise generating activities, removes the pollutant from the environment completely. Therefore, if there is real legislative desire to address this pollutant the prognosis could be very good. However, as with most pollutants, shipping is closely linked to economic growth and development, which more often than not is the key driver and winner in environmental legislation debates. The mitigation of shipping noise can be achieved through the development of quieting technologies and ship design, appropriate ship maintenance, the implementation of speed limits in ecologically important areas and the relocation of shipping lanes (IMO 2014; Williams et al. 2019). The implementation of these measures would require the support of industry and careful enforcement. The Port of Vancouver is already assessing many of these options and introduced incentives for industry to implement ship quieting technologies by offering reduced port fees to ships that meet certain noise footprint criteria (Vancouver Fraser Port Authority 2017). In addition, they have set up a noise monitoring network in the port and are, therefore, well placed to determine if such measures help to reduce shipping noise (Vancouver Fraser Port Authority 2017; Joy et al. 2019). This type of work will help inform mitigation in other ports around the world.

In conclusion, shipping noise is a low frequency, pervasive pollutant in the world’s oceans. It has been shown that trends in ship numbers and the gross tonnage of ships have been on a long-term upward trajectory, and that as a
result, the levels of ship noise, and hence the potential harm associated with it, is likely to follow the same trend. There is a lot of available knowledge on the frequencies and sources of ship noise and how shipping noise can be measured in the field or predicted using models. To date, it has been possible to use such approaches to understand, to some extent, the detrimental impact ship noise can have on the communication, hearing and behaviour of marine mammals. However, for many species of marine mammal it is still necessary to determine, in detail, the impact ship noise may have on important life functions to inform an increasing level of understanding and regulation among the policy making community. Reductions in shipping noise will require regionally and internationally agreed limits on noise emissions from ships through the development of effective policy drivers. The development of these policy instruments depends on a better understanding of the impacts of noise, and on more efficient and effective predictions of shipping noise. The output from the remaining chapters of this thesis sit firmly within this current gap in our understanding.
Exposure of diving grey seals to shipping noise in the Celtic Sea and English Channel
3.1 Introduction

Chapter 2 highlighted the negative impacts of shipping noise on marine mammals. In response to such evidence, a number of international regulatory bodies are taking steps to mitigate the associated risks. Strategies and guidelines to address anthropogenic underwater noise, including shipping noise, have been generated by the International Maritime Organisation (IMO 2014), National Oceanic and Atmospheric Administration, Fisheries and Oceans Canada, and the United Nations. Notably, the EU Marine Strategy Framework Directive (MSFD) requires member states to monitor shipping noise and take mitigating steps to ensure it does not compromise the ‘Good Environmental Status’ of EU waters (European Commission 2008; European Commission 2010; European Commission 2017), and work to achieve this is co-ordinated by regional agreements such as OSPAR (OSPAR Commission 1992). However, effective management is still constrained by a lack of data pertaining to the exposure of marine life to shipping noise. It is difficult for policy to set targets for acceptable noise levels without data on historic and current noise levels against which to track trends and measure the effectiveness of policy measures to mitigate noise (Merchant et al. 2016). Data on noise exposure and at-sea spatial usage of different marine species are also central to developing an understanding of the zones of highest risk with respect to shipping noise (Erbe et al. 2012a; Merchant et al. 2017). It is necessary to understand the exposure of an individual, and consequently populations, in order to explore the impact of this exposure on marine animals (Van der Graaf et al. 2012; Merchant 2019).

As highlighted in Chapter 1, approximately 38% of the world’s grey seal population breed around the UK coastline and the species is protected under Annex II of the EU Habitats Directive (SCOS 2018). The size of the population has a positive trend but growth has been slowing in recent years as the population faces many different pressures such as by-catch (Cosgrove et al. 2016; Osinga et al. 2012), climate change (SCOS 2018), renewable energy
installations (Russell et al. 2016), disease (Yon et al. 2019) and chemical pollutants. Furthermore, as central-place foragers that return to haul-out sites to rest, breed and moult, seals heavily utilise the coastal zones which are also home to busy shipping lanes. Jones et al. (2017) highlighted the daily rate of co-occurrence for harbour seals, grey seals and shipping (mean number of seals × mean daily number of vessel transits in a 5 km grid square) was ≥ 100 within 50 km of the coast near to haul-out sites. At haul-out sites, evidence suggests seals can be flushed into the water by approaching ships and exhibit alert and orienting behaviour in response to the sound of boat playbacks (Jansen et al. 2015; Tripovich et al. 2012). However, there is still very little information about the at-sea exposure of seals to shipping noise and their spatial relationship with shipping given their three-dimensional use of the underwater environment.

Phocid seals are quoted to have a functional hearing range from 50 Hz – 80 kHz (National Marine Fisheries Service 2018), which overlaps with the dominant frequencies of noise emitted by ships (10 – 1000 Hz). Seals utilise sound production and reception during mating, mother-offspring interactions and while maintaining territory (Van Parijs et al. 2001; Hayes et al. 2004). Asselin et al. (1993) recorded the ‘rups’ and ‘growls’ of grey seals (Halichoerus grypus) at frequencies between 100 and 500 Hz. These are frequencies at which shipping noise could mask such calls and reduce the communication space between seals (Bagočius 2014). Grey seal pups, particularly, warrant consideration with respect to disturbance by shipping noise due to their naivety. Following a short lactation period, pups are left on the colony by their mother and undergo a post-weaning fast during which they can lose up to a quarter of their body weight (Noren et al. 2008). As a result, when they go to sea for the first time they are under great pressure to feed successfully and they must do so without parental guidance. Consequently, they may be vulnerable to disturbance at this high-risk time. They are also not likely to be habituated to shipping noise, as may be expected of more experienced adults, and they may have more sensitive hearing placing them at risk of disturbance (Gotz and Janik 2010).
Exposure to anthropogenic underwater noise has the potential to induce auditory damage. Auditory damage is exhibited by an increase in the threshold level at which an animal can hear at a given frequency. This can be either temporary or permanent (Southall et al. 2007). Hastie et al. (2015) calculated that 50% of harbour seals in their study would experience permanent threshold shift (PTS) at an onset threshold of 186 dB re $1\mu Pa^2s$ during pile driving for an offshore windfarm. However, Southall et al. (2007) proposed, for non-impulsive sounds such as shipping noise, pinnipeds will experience temporary threshold shift (TTS) when m-weighted cumulative sound exposure levels over 24 hours exceeds 183 dB re $1\mu Pa^2s$ and PTS when cumulative sound exposure levels over 24 hours exceeds 203 dB re $1\mu Pa^2s$. These were recently revised down by the (National Marine Fisheries Service 2018) to 181 dB re $1\mu Pa^2s$ and 201 dB re $1\mu Pa^2s$ for TTS and PTS respectively. Jones et al. (2017) predicted the exposure of harbour seals in the Moray Firth, Scotland, UK, and reported that only when considering upper confidence intervals did some estimates exceeded the 183 dB re $1\mu Pa^2s$ threshold for the onset of TTS. However, these predictions suggest there is still great uncertainty associated with noise predictions and they were based on the two-dimensional movements of seals at-sea. They also relied on basic spreading models of underwater noise propagation with limited consideration of environmental properties. Mean sound pressure levels for these seals was greater than 140 dB re $1\mu Pa$ near port areas. In contrast, acoustic tags, which record seal location and noise levels, have recorded ship noise exposure for 2 harbour and 2 grey seals (Mikkelsen et al. 2019). The tags recorded for approximately 14 days and 2.2% to 20.5% of the at-sea time during these days contained audible ship noise. They reported an example ship encounter event where ship noise reached a maximum broadband level of 113 dB re $1\mu Pa$ RMS (0.1 - 50 kHz, 1 s average).

To assess noise from shipping, predictions primarily take the form of two-dimensional maps (Erbe et al. 2014). These allow managers to examine how noise varies in time and space. However, marine life utilises a
three-dimensional environment. Underwater noise from shipping can vary significantly in horizontal space and vertically through the water column (Chen et al. 2017). This is particularly relevant for shallow shelf seas, which are regions of intersection between dynamic environmental properties, high density shipping and ecologically productive regions with high species diversity (Simpson and Sharples 2012). It is well understood that environmental properties such as salinity, temperature, bathymetry and sediment type have a profound impact on sound propagation (Etter 2013). Studies have shown that the presence of a thermocline or oceanic front can result in horizontal and vertical changes in noise levels (Shapiro et al. 2014). Vertical changes in noise levels are particularly relevant for grey seals. As air breathing, benthic foragers, they repeatedly dive throughout the water column, utilising the three-dimensional environment. Evidence suggests that they can potentially experience differential noise exposure of up to 10 dB as they move vertically throughout this environment (Chen et al. 2017). This suggests depth may be a factor in heightened or reduced exposure of seals to shipping noise compared to two-dimensional map predictions.

This chapter aims to predict the exposure of individual seals to shipping noise using a sophisticated underwater acoustic propagation model and the three-dimensional location and dive track of seals generated from Fastloc® GPS/GSM tags. Specifically, the study aims to investigate the at-sea exposure of grey seals at two different life stages; pups and adults. The seal tracking data will link noise exposure directly to at-sea vertical and horizontal spatial use by seals, improving the applicability of the results to risk calculations and marine spatial planning. The study also aims to investigate the influence of ship source level, the number of ships and the proximity of ships to seals on predicted noise exposure levels.
3.2 Methodology

This study undertook an historic reconstruction of ship noise exposure levels for seal pups in the Celtic Sea and adult seals primarily located in the English Channel. Seals were tagged with Fastloc® GPS/GSM tags, which provided location and dive data for each seal. The seals were tagged as part of separate studies on animal movement and habitat use from 2009 to 2013 (Thompson 2012; Huon et al. 2015). Ship noise exposure levels in a 24 hour period were predicted along each seal's three-dimensional track using historic records of ship movements, a ship source level model and a range dependent acoustic propagation model.

3.2.1 Study area

Noise reconstruction was undertaken in the seas around south-west UK and northern France as shown in Figure 3.1. This region hosts shipping lanes through the Celtic Sea and western English Channel (Fig. 3.2). The area has a high volume of shipping traffic and acts as a gateway to the large ports of northern Europe. These include Rotterdam and Antwerp; Europe’s largest ports (Eurostat 2018) and the UK’s 5th largest port of Milford Haven (DfT 2017). However, grey seals also utilise breeding and haul-out sites along the coastlines of the region resulting in significant overlap between grey seals and shipping (Jones et al. 2017; SCOS 2018). The region is a dynamically active, shallow, shelf sea (Pingree 1980). The water column structure is characterised by mesoscale eddies and fronts, as well as the development of a strong thermocline in the summer (April to November) and its slow breakdown over winter (Pingree 1980). These features have a significant impact on acoustic wave propagation and the prediction of underwater noise levels (Shapiro et al. 2014).

3.2.2 Seal location and movement data

The at-sea movement of seals was obtained from dive and location data collected using Global Positioning System (GPS) and Global System for Mobile
Communications (GSM) Fastloc® tags (Sea Mammal Research Unit, UK). The devices carry the Fastloc® GPS system (Wildtrack Telemetry Systems, UK), pressure sensor, wet/dry sensor and temperature sensor. The Fastloc® GPS system provides accurate location data for marine animals, which are only visible to satellites when surfacing for a small amount of time, by capturing satellite information in <1 second and determining position estimates in post-processing (Dujon et al. 2014). The details of 18 seals included in the study are given in Table 3.1. Celtic Sea animals were tagged in 2009 or 2010 at sites in Anglesey or Ramsey Island, Wales, UK (Table 3.1, Fig. 3.1) under Home Office Licence No. 60/4009. English Channel animals were tagged in the Iroise Marine Park under licences No. 10/102/DEROG and 13/422/DEROG provided by the French Ministry of the Environment (Fig. 3.1). Seals were caught, anaesthetised using Zoletil® (Vibrac, France) and tags glued to clean, dry fur at the base of the neck using epoxy resin or cyano-acrylate contact adhesive. Seals caught in 2009 were not anaesthetised. The tagging methodology
followed McConnell et al. (1999) and is explained further by Thompson (2012, p. 6) and Huon et al. (2015, p. 1093).

**Table 3.1:** Details of seal tag data used in the study. A total of 18 seals were included; 9 adults and 9 pups. Noise was calculated for a total of 86 days. The table shows the percentage of the total time the seal spent at sea used in the study. ISMP - Iroise Sea Marine Park

<table>
<thead>
<tr>
<th>ID</th>
<th>Location tagged</th>
<th>Mass (kg)</th>
<th>Sex</th>
<th>% track used</th>
<th>Days</th>
<th>Age Class</th>
<th>Dates</th>
</tr>
</thead>
<tbody>
<tr>
<td>B23</td>
<td>ISMP</td>
<td>129</td>
<td>M</td>
<td>3.4</td>
<td>4</td>
<td>Adult</td>
<td>Oct/Nov 2011</td>
</tr>
<tr>
<td>B24</td>
<td>ISMP</td>
<td>124</td>
<td>M</td>
<td>4.8</td>
<td>6</td>
<td>Adult</td>
<td>Sep/Nov/Dec 2011</td>
</tr>
<tr>
<td>B26</td>
<td>ISMP</td>
<td>68</td>
<td>F</td>
<td>0.6</td>
<td>1</td>
<td>Adult</td>
<td>Jan 2012</td>
</tr>
<tr>
<td>B27</td>
<td>ISMP</td>
<td>152</td>
<td>M</td>
<td>2.4</td>
<td>4</td>
<td>Adult</td>
<td>Sep/Oct 2011</td>
</tr>
<tr>
<td>B31</td>
<td>ISMP</td>
<td>206</td>
<td>M</td>
<td>4.0</td>
<td>4</td>
<td>Adult</td>
<td>Jul/Sept 2013</td>
</tr>
<tr>
<td>B32</td>
<td>ISMP</td>
<td>114</td>
<td>F</td>
<td>3.4</td>
<td>4</td>
<td>Adult</td>
<td>Nov 2013</td>
</tr>
<tr>
<td>B33</td>
<td>ISMP</td>
<td>210</td>
<td>M</td>
<td>7.3</td>
<td>11</td>
<td>Adult</td>
<td>Jul/Oct/Nov 2013</td>
</tr>
<tr>
<td>B35</td>
<td>ISMP</td>
<td>148</td>
<td>M</td>
<td>3.5</td>
<td>4</td>
<td>Adult</td>
<td>Oct/Dec 2013</td>
</tr>
<tr>
<td>B37</td>
<td>ISMP</td>
<td>70</td>
<td>M</td>
<td>3.8</td>
<td>4</td>
<td>Adult</td>
<td>Aug/Oct 2013</td>
</tr>
<tr>
<td>hg27-01-09</td>
<td>Anglesey</td>
<td>37</td>
<td>M</td>
<td>2.1</td>
<td>3</td>
<td>Pup</td>
<td>Feb/Mar 2010</td>
</tr>
<tr>
<td>hg27-04-09</td>
<td>Anglesey</td>
<td>38</td>
<td>M</td>
<td>3.3</td>
<td>5</td>
<td>Pup</td>
<td>Jan/Mar/May 2010</td>
</tr>
<tr>
<td>hg29-11-10</td>
<td>Anglesey</td>
<td>35</td>
<td>M</td>
<td>2.0</td>
<td>5</td>
<td>Pup</td>
<td>Jun/Jul 2011</td>
</tr>
<tr>
<td>hg29-15-10</td>
<td>Ramsey</td>
<td>39</td>
<td>F</td>
<td>0.5</td>
<td>1</td>
<td>Pup</td>
<td>Dec 2010</td>
</tr>
<tr>
<td>hg29-16-10</td>
<td>Anglesey</td>
<td>40</td>
<td>F</td>
<td>4.4</td>
<td>5</td>
<td>Pup</td>
<td>Dec 2010/Jan 2011</td>
</tr>
<tr>
<td>hg29-18-10</td>
<td>Ramsey</td>
<td>32</td>
<td>M</td>
<td>10.6</td>
<td>9</td>
<td>Pup</td>
<td>Nov/Dec 2010</td>
</tr>
<tr>
<td>hg29-21-10</td>
<td>Ramsey</td>
<td>37</td>
<td>M</td>
<td>5.5</td>
<td>7</td>
<td>Pup</td>
<td>Oct/Dec 2010</td>
</tr>
<tr>
<td>hg29-23-10</td>
<td>Ramsey</td>
<td>29</td>
<td>M</td>
<td>5.3</td>
<td>1</td>
<td>Pup</td>
<td>Nov 2010</td>
</tr>
<tr>
<td>hg29-24-10</td>
<td>Ramsey</td>
<td>32</td>
<td>F</td>
<td>25.8</td>
<td>8</td>
<td>Pup</td>
<td>Oct 2010</td>
</tr>
</tbody>
</table>

Erroneous GPS locations were identified as those obtained using fewer than 5 satellites and/or having high residual error values from the Fastloc® position algorithm (Russell and McConnell 2014). This cleaning procedure can result in 95% of locations with distance error < 50 m (Russell and McConnell 2014). An animal was given the status ‘diving’ when the tag registered a depth of 1.5 m or deeper for greater than 8 seconds. A dive ended when depth was shallower than 1.5 m. The tags simplify dive profiles to nine dive inflection points to compress data for transmission (Fedak et al. 2001). These points reflect where the dive path changes most rapidly (Fedak et al. 2001) but are not georeferenced. In order to produce a three-dimensional track for each seal, the timestamps of location and dive inflection points were used to interpolate each dive in space using hermite curve interpolation (Tremblay et al. 2006; Kuhn et al. 2010). In addition, the tags do not record regular location fixes because they rely on the
seal surfacing to capture satellite data. As a result, the time between location points can vary, and there can be bias in the number of GPS points to locations where the seal is not diving. To address this, the three-dimensional seal track was re-sampled using hermite curve interpolation at a rate of 1 second to produce a track with regularly spaced location points (Tremblay et al. 2006; Kuhn et al. 2010). Hermite curve interpolation can more closely represent the curvilinear paths of animals moving through a fluid environment than linear interpolation (Tremblay et al. 2006).

The accuracy of the interpolated dive locations was determined by the amount of time to the nearest measured GPS fix. In order to retain as much continuous track as possible to allow noise exposure predictions, but remove inaccurate dive locations, tracks were segmented. A segment ended where a dive was greater than 180 minutes from a GPS fix. A new track segment started at the next GPS location fix. A value of 180 minutes retained enough track for 24-hr noise exposure calculations but ensured 95% of dives included within the study were still within 30 mins or less of a GPS fix limiting potential error.

In order to calculate at-sea 24-hr cumulative sound exposure levels, periods of haul-out were excluded and track segments longer than 24 hours in duration extracted. Haul-outs were determined by the wet/dry sensors aboard the tag and periods of haul-out were transmitted as part of the tag data message. In addition, track segments had to be located entirely within the study area to ensure AIS data coverage. Seals spent a large proportion of time in very shallow shelf areas. Specifically, the English Channel seals spent a substantial share of their time on the island systems at the north-west tip of France (Fig. 3.1) (Huon et al. 2015). The shallow water and islands complicate the calculation of noise propagation. The acoustic propagation model used in this study does not represent the horizontal reflection and refraction of noise within and around a group of islands (Wang et al. 2014). As a result, the study did not include tracks where the seal was in water less than 10 m deep. This was not considered significant because low frequencies are absorbed by soft sediments at very
shallow depths (Urick 1983), and the island system is not dominated by the large commercial ships, which are the focus of this study, and are included in AIS data (Fig. 3.2). It should be recognised, however, there may be noise exposure at these locations from small recreational boats that do not carry AIS transmitters and emit higher frequency noise (1 - 125 kHz) (Li et al. 2015). As a result, reported noise level estimates are minimum exposure values from shipping. The 24-hr track segments along which noise was estimated are shown in Figure 3.1 and the number of days processed for each seal is shown in Table 3.1.

### 3.2.3 Ship location data

The International Convention for the Safety of Life at Sea (SOLAS) 1974 regulation V/19 requires that all ships of 300 gross tonnage or upward engaged on an international voyage, cargo ships greater than 500 gross tonnage not on an international voyage and all passenger ships must be fitted with an automatic identification system (AIS). AIS systems report the location of a vessel at sea aiding the safety of navigation and monitoring of vessel traffic. Due to the safety implications, a large number of smaller boats and fishing vessels also take part in the scheme voluntarily and are hence included in the study. AIS transceivers transmit the location of a ship and a number of descriptive meta-data variables to land based receivers or more recently, satellite.

This study utilised historic terrestrial AIS data to determine the location of ships at sea in relation to the grey seal tracks. AIS data was obtained from shipais.com and Marine Traffic for the time periods shown in Table 3.2. Each dataset provided coverage for a subsection of the total study area (Fig. 3.1), but overall this resulted in coverage of the whole area for the seal data (Fig. 3.1 and Fig. 3.2). Terrestrial AIS has a limited range from the coast, therefore, it is difficult to distinguish between areas of no coverage and areas of no shipping. However, the presence of AIS-B receivers, which are less powerful and usually fitted voluntarily to recreational boats and fishing vessels, was used as an indication of good coverage within an area. The data from all sources were
combined in a SQLite database and matched on the unique field ‘MMSI number’. This approach increased the number of records for each ship, allowed more accurate interpolation of a ship's transect, and increased the spatial coverage of the study area.

**Table 3.2:** Details of raw AIS data used for the study. Data was uploaded to a SQLite database. Data sources include; MTCS – Marine Traffic complete coverage of Celtic Sea region, MTEC – Marine Traffic partial coverage of English Channel area, and SA – data from shipais.com for the complete region.

<table>
<thead>
<tr>
<th>Data Source</th>
<th>Start Date</th>
<th>End Date</th>
<th>Records</th>
</tr>
</thead>
<tbody>
<tr>
<td>MTCS</td>
<td>01-01-2010</td>
<td>31-12-2010</td>
<td>1607605</td>
</tr>
<tr>
<td>MTEC</td>
<td>01-09-2011</td>
<td>30-11-2011</td>
<td>920460</td>
</tr>
<tr>
<td>MTEC</td>
<td>01-09-2013</td>
<td>30-11-2013</td>
<td>1252352</td>
</tr>
<tr>
<td>SA</td>
<td>01-01-2010</td>
<td>31-12-2010</td>
<td>5153999</td>
</tr>
<tr>
<td>SA</td>
<td>16-06-2011</td>
<td>28-01-2012</td>
<td>2260067</td>
</tr>
<tr>
<td>SA</td>
<td>09-06-2012</td>
<td>04-01-2013</td>
<td>2948130</td>
</tr>
<tr>
<td>SA</td>
<td>09-06-2013</td>
<td>04-02-2014</td>
<td>3509790</td>
</tr>
</tbody>
</table>

The data were processed as summarised in Table 3.3. The combined data were sorted by ship MMSI number, and timestamp. The data were then split into
transects. A transect was defined as containing more than one actual AIS location point, and the ship was moving at a speed over ground between 1.5 and 60 knots. Ships slower than this were likely to be stationary or drifting at anchor (Marine Management Organisation 2014; Marine Management Organisation 2015), and those higher than this had MMSI corresponding to search and rescue aircraft. A transect ended and a new transect started when there was greater than 180 minutes between location points. The next point was the start of a new transect. This 180 minute time interval was used because it was short enough to resolve ships rounding Land's End, UK and heading north into the Celtic Sea, as well as those leaving and returning to the study area, while retaining the presence of as many ships as possible. The location of a ship along the transect at a particular time was estimated using linear interpolation. Initially, a total of 23,373 ships were included in the AIS database. A subset of 930 MMSI numbers were removed from the analysis because no data on vessel length was available or length was recorded as zero resulting in 22,443 ships in the final AIS database. A summary of ships included in the database is given in Table 3.4. The cleansed data resulted in 365,998 individual transects. It must be noted however, that while AIS data provides a good estimate of locations for large ships there were still a number of incomplete transects (i.e. start/end within the study area not at a port or location outside the study area) and missing smaller ships from the data. As a result, the reported values are minimum ship noise estimates.

3.2.4 Ship source model

The source level of each ship was calculated using the Research Ambient Noise Directionality (RANDI) model, which has been widely implemented in ship noise modelling (Breeding et al. 1996; Chen et al. 2017; Erbe et al. 2014; Jones et al. 2017). The model is based on work by Ross (1976) and is assumed to be a monopole source level (Wales and Heitmeyer 2002; Ainslie et al. 2009). The model is based on the relationship between noise level, speed and vessel length.
Table 3.3: Steps undertaken to cleanse and process the raw AIS data into ship passage transects.

<table>
<thead>
<tr>
<th>Stage</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Data from each source was combined into a database. It was standardised on input to give time in seconds since the unix epoch (January 1st 1970 00:00:00) and speed in knots. Fields common to all input sources were retained (i.e. status was dropped because it was not available for data from shipais.com). Fields: MMSI, LAT, LON, SECS, TIMESTAMP, SPEED, LENGTH, DATA_SOURCE.</td>
</tr>
<tr>
<td>2</td>
<td>Data without a valid 9 digit MMSI number was removed and all data sorted by MMSI and timestamp.</td>
</tr>
<tr>
<td>3</td>
<td>Data was split into transects for each ship. Transects ended when the time between position points was greater than 180 minutes. Locations with a gap of less than 180 minutes were linearly interpolated to give a transect. Only transects with two position points or more were retained.</td>
</tr>
<tr>
<td>4</td>
<td>Transects for MMSI numbers known to be search and rescue aircraft, had a length less than or equal to zero, were known base stations or where no reference details were available were removed from the study.</td>
</tr>
<tr>
<td>5</td>
<td>A ship was excluded from the analysis if it was travelling less than 1.5 knots, or greater than 60 knots, in order to ensure aircraft, stationary ships and ships drifting at anchor were excluded.</td>
</tr>
</tbody>
</table>

Table 3.4: Details of ships included in the cleaned AIS database summarised in length classes.

<table>
<thead>
<tr>
<th>Length Class (ft)</th>
<th>Number of Ships</th>
<th>Median 1/3 octave band source level 10-1000 Hz (db re 1(\mu)Pa)</th>
<th>Median speed (kts)</th>
</tr>
</thead>
<tbody>
<tr>
<td>&lt; 30</td>
<td>360</td>
<td>132</td>
<td>4.3</td>
</tr>
<tr>
<td>30 – 180</td>
<td>5247</td>
<td>139</td>
<td>4.6</td>
</tr>
<tr>
<td>180 – 330</td>
<td>2871</td>
<td>172</td>
<td>9.1</td>
</tr>
<tr>
<td>330 – 480</td>
<td>3314</td>
<td>183</td>
<td>11.9</td>
</tr>
<tr>
<td>480 – 630</td>
<td>4897</td>
<td>189</td>
<td>13.1</td>
</tr>
<tr>
<td>&gt; 630</td>
<td>5762</td>
<td>196</td>
<td>14.4</td>
</tr>
</tbody>
</table>
as shown in Equation 3.1, where $f$ is the frequency in hertz; $v$ is the speed in knots and $l_s$ is length in feet. Breeding et al. (1996) included additional correction parameters ($df$ and $dl$). $L_{so}$ is a reference spectrum for a ship of 300 feet travelling at 12 knots. The length and speed of the ship is derived from the AIS message. Source levels were calculated at every 1 Hz between 10 and 1000 Hz and integrated to give 1/3 octave band source levels.

$$L_s(f, v, l_s) = L_{so}(f) + 60\log\left(\frac{v}{12}\right) + 20\log\left(\frac{l_s}{300}\right) + df \times dl + 3.0$$

0.00 \leq f \leq 28.40 : df = 8.1

28.4 < f \leq 191.60 : df = 22.3 - 9.7 \times \log(f)

$$dl = \frac{l_s^{1.15}}{3643}$$

$f < 500 Hz : L_{so}(f) = -10\log(10^{-1.06/\log(f-14.34)} + 10^{3.32/\log(f-21.425)})$

$f > 500 Hz : L_{so}(f) = 173.2 - 18.0\log(f)$

(3.1)

Figure 3.3 shows the average 1/3 octave band ship source levels between 10 and 1000 Hz for all ships in the database grouped by vessel length. This demonstrates the range of source levels included in the study but when calculating noise, source level was obtained for each ship individually in a 15 minute period using the ships length and speed over ground at that point along its transect. Ship source levels were not grouped into classes.

### 3.2.5 Acoustic propagation model

The parabolic equation model RAMSurf\(^1\) (Collins 1993) was used to calculate propagation loss between each sound source and the location of each seal. This model is suitable for range dependent, low frequency, shallow water scenarios (Etter 2013). In this study, the horizontal and vertical step parameters for the

\(^1\)available at: http://oalib.hlsresearch.com/PE/ramsurf/
Figure 3.3: Median predicted 1/3 octave band source levels for all ships in the AIS database grouped by length class. When calculating noise, the source level for each ship was calculated separately using the speed the ship was travelling at that time. For this diagram, ship source level was calculated using the average speed for each ship and then the median source level for each ship length class is reported above.

acoustic model were fixed at 50 m and 0.5 m respectively for all simulations. These ensured a convergent solution across all frequencies tested. The maximum depth of computational space was 500 m but results were only output to 200 m. The sea surface was assumed to be flat. The source depth was related to vessel length as suggested by Erbe et al. (2012b) with ships greater than 164 ft (∼50 m) assigned a source depth of 6 m (Scrimger and Heitmeyer 1991) and smaller vessels a depth of 3 m (Erbe et al. 2012b). Additional fixed model parameters are shown in Supplementary Material A.1. The model considers detailed three-dimensional environmental changes. The environmental conditions were described along each transect by submitting the bathymetric depth, a sound speed profile for the water column and geoacoustic parameters for the sediment, as described in the following sections, every 2 km to the maximum range of each transect.

Simulations were conducted at the centre frequencies of one-third octave bands between 10 and 1000 Hz. This frequency range encompasses the maximum energy output for ships and covers both of the frequencies (63 and 125 Hz)
recommended by the Marine Strategy Framework Directive as important for monitoring shipping noise (European Commission 2008; European Commission 2010; European Commission 2017). However, it is noted that ship source levels do extend beyond this (Veirs et al. 2015). It would be significantly more complex to include higher frequencies within the noise modelling framework because the RAMSurf model does not operate efficiently at higher frequencies (Etter 2018). The propagation loss output was smoothed to remove variation associated with the coherent nature of the model. This was completed using a moving average (Harrison and Harrison 1995). A comparison of the moving average smoothing method and a Butterworth filter, which can also be used for this task, is shown in Section 3.3.3.

### 3.2.5.1 Sediment

The sediment type at each 2 km point in range along the transect was determined using data modified from the EMODnet Geology project seabed substrate map (1:1000000) with five seabed substrate classes generated from the modified Folk triangle (European Commission 2016; Long 2006). If there were any missing data points in the study area they were classified as the same substrate type as the nearest point with a known sediment classification. The geoacoustic parameters for the 5 classified sediment types are given in Table 5. These were extracted from the literature based on the percentage of mud, sand and gravel given in the sediment classification.

**Table 3.5:** Geoacoustic parameters for RAMSurf model. Values were selected from published geoacoustic data. Cp: P-wave sound speed, $\alpha$: P-wave attenuation.

<table>
<thead>
<tr>
<th>Sediment Type</th>
<th>Density (g cm$^{-3}$)</th>
<th>Cp (ms$^{-1}$)</th>
<th>$\alpha$ (dB$^{-1}$)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mud/Muddy Sand</td>
<td>1.740</td>
<td>1615</td>
<td>1.00</td>
<td>Hamilton (1980) and NURC (2008)</td>
</tr>
<tr>
<td>Sand</td>
<td>1.941</td>
<td>1749</td>
<td>0.80</td>
<td>Hamilton (1980) and NURC (2008)</td>
</tr>
<tr>
<td>Coarse Sediment</td>
<td>2.000</td>
<td>1800</td>
<td>0.60</td>
<td>NURC (2008)</td>
</tr>
<tr>
<td>Mixed Sediment</td>
<td>2.034</td>
<td>1836</td>
<td>0.90</td>
<td>Hamilton (1980) and Lurton (2002)</td>
</tr>
<tr>
<td>Rock</td>
<td>2.200</td>
<td>2400</td>
<td>0.20</td>
<td>NURC (2008)</td>
</tr>
</tbody>
</table>
3.2.5.2 Bathymetry

The bathymetry of UK and Irish waters were determined using the EMODnet Digital Bathymetry (DTM 2016) at 1/8 * 1/8 arc minute resolution (EMODnet Bathymetry Consortium 2016). This data is given in metres with reference to lowest astronomical tide. EMODnet data for these waters was converted from lowest astronomical tide to mean sea level using the Vertical Offshore Reference Frame surface developed by the UK Hydrographic Office (Adams et al. 2006). Bathymetric data for French waters were taken from the MNT Bathymétrique de façade Atlantique (Projet Homonim) which is provided in metres with reference to mean sea level (Shom 2015).

Bathymetry data is traditionally referenced to lowest astronomical tide. However, the seal tag data provides depth with reference to the water surface. This varies in height with respect to the sea floor throughout the tidal cycle. The seals are diving throughout the tidal cycle and therefore, can dive deeper than the bathymetry layer at certain points. To minimise this phenomenon, bathymetric data was given relative to mean sea level. This is not a perfect solution and some dives still penetrate the sea floor due to location error in the seal tags and the coarse resolution of the bathymetric data. The consequences for noise exposure were addressed using a correction as described in Section 3.2.7. Further steps were taken to assess the impact of this with respect to noise levels as described in Sections 3.2.7 and 3.2.9.

3.2.5.3 Sound speed

The sound speed profile was calculated every 2 km using the 9 term equation proposed by Mackenzie (1981). The equation calculates sound speed as a function of temperature, salinity and depth as shown in Equation 3.2 where T is temperature in degrees Celsius, S is salinity in parts per thousand and D is equal to depth in metres. The temperature, salinity and depth for each point were extracted from the Iberian Biscay Irish Ocean Reanalysis system available through the E.U. Copernicus Marine Environment Monitoring Service (CMEMS;
product identifier: IBI_REANALYSIS_PHYS_005_002). The model provides three-dimensional daily ocean fields for temperature and salinity. The IBI model numerical core is based on the NEMO v3.6 ocean general circulation model run at 1/12° horizontal resolution. Altimeter data, in situ temperature and salinity vertical profiles and satellite sea surface temperature are assimilated. CMEMS (n.d.) provide a detailed description of model development. The model has a resolution of 0.083 degrees × 0.083 degrees with 50 depth levels.

\[
c(D, S, T) = 1448.96 + 4.591T - 5.304 \times 10^{-2}T^2 + 2.374 \times 10^{-4}T^3 \\
+ 1.340(S - 35) + 1.630 \times 10^{-2}D + 1.675 \times 10^{-7}D^2 - 1.025 \\
\times 10^{-2}T(S - 35) - 7.139 \times 10^{-13}TD^3
\]

(3.2)

### 3.2.6 Construction of three-dimensional received noise levels

At each 15 minute time step a three-dimensional noise field of m-weighted broadband (10 - 1000 Hz) received levels (RL) was generated for the area enclosing the dive and location track of the seal (Fig. 3.4). RLs for each ship were calculated by subtracting smoothed propagation loss (PL) values, calculated using the RAMSurf model, from the ship source levels (SL), calculated using the RANDI ship source model (Equation 3.3). The RAMSurf model output is two-dimensional (range and depth). Three-dimensional coverage of the area enclosing the seal track was generated by calculating PL along multiple transects at an azimuth of 2.5°. This produced a noise field composing depth and range at multiple azimuths (Figure 3.4). PL was calculated at the centre frequencies of 1/3 octave bands between 10 and 1000 Hz and subtracted from the 1/3 octave band source levels generated by the RANDI ship source model (Sec. 3.2.4). RL were weighted using the frequency m-weighting function for pinnipeds proposed by Southall et al. (2007) (Fig. 3.5). The frequency weighting function reduces noise levels at frequencies that pinnipeds cannot perceive sound. The weighting function is derived from available
pinniped audiograms (Southall et al. 2007). M-weighted broadband noise levels (10-1000 Hz) were calculated by integrating across all frequencies (approximated by summation). Total m-weighted broadband RL (10-1000 Hz) from all ship noise sources at each point along the seal track was calculated by summing the intensities of each ship as shown in Equation 3.4 where $l_i$ is the $i^{th}$ ship and $n$ is the number of ships in 15 minutes.

$$RL = SL - PL$$  \hspace{1cm} (3.3)

$$totalRL = 10 \log_{10} \sum_{i=1}^{n} 10^{l_i/10}$$  \hspace{1cm} (3.4)

The ship locations were determined for the mid-point of each 15 minute time period. Noise was calculated every 15 minutes at 0.5 m in depth but processed at a depth of 1 m to aid in computational efficiency. The ships remained stationary during each 15 minute period and the seal moved throughout the noise field. It is recognised that in reality the ships and seals would move relative
to each other in a 15 minute period. However, the computational time required to recalculate the sound field using the RAMSurf model is a key factor in determining the possible temporal resolution for noise calculations. This parameter was included in the sensitivity analysis (Section 3.2.9 and 3.3.3). The median absolute deviation between using a 5 and 15 minute temporal resolution was less than 1 dB suggesting a temporal resolution of 15 minutes is both efficient and sufficiently accurate.

All ships within 120 km of the seals location in a 15 minute period were included in noise calculation estimates. At this distance propagation loss is estimated to be between 100 and 120 dB. Average maximum source level for the 1/3 octave band centred at 31.5 Hz, for the longest group of ships, is approximately 187 dB re 1 µPa. As a result, ships at a distance greater than 120 km are likely to be making negligible contributions (~60 dB) to the sound field at the location of the seal. This cut-off procedure improved the efficiency of the noise modelling calculations. It is a precautionary threshold. Ships closer to the seal will be contributing much greater noise levels (> 100 dB) to the total RL than ships at 120 km. Ships 120 km away could still contribute to noise levels despite low individual noise levels, especially if there are many ships in that location.

Figure 3.5: M-weighting frequency function for pinnipeds proposed by Southall et al. (2007). It is based on an estimated functional hearing range for pinnipeds between 75 Hz and 75 kHz.
because the intensity of each source is summed. Values included in the study are m-weighted and therefore, are considered perceivable by the seal but would not necessarily result in any negative effect or may not be discernible from other low frequency ambient noise sources.

Due to the high costs associated with AIS data, it was only possible to obtain the data for a fixed spatial area. The seals and ships utilised a region much greater than this so it was necessary to consider the impact the edge of the study area would have on noise estimates. Seals located close the boundary would be exposed to fewer ships due to the lack of AIS data outside the boundary. To combat this issue a 15 km buffer zone was implemented. The majority of days processed were located greater than 70 km from the AIS boundary with only 5 days located at the edge of the 15 km buffer zone.

### 3.2.7 Cumulative noise exposure levels and prediction of auditory damage

The noise exposure of the seal was linearly interpolated from the noise field for each 24-hr period to give sound exposure levels with a temporal resolution of 1 second. The temporal cumulative exposure period of 24 hours is known to be arbitrary (National Marine Fisheries Service 2018). Mean at-sea time between haul-outs for the seals in this study was greater than 24 hours for the Celtic Sea pups at 33 hours and less than this for English Channel adults at 21 hours, although maximum time between haul-outs was considerably greater than this. However, the standard cumulative period advocated for and utilised in current legislation for assessing auditory threshold shift is 24 hours, and it is again utilised here to ensure outputs are relevant to regulatory bodies and comparable to previous studies (National Marine Fisheries Service 2018; Southall et al. 2007). This value did not influence the number of days included in the study for each seal. The limiting factor on the number of 24 hour days was overlap between seal and AIS data as well as execution time of the modelling workflow.

As discussed in Section 3.2.5.2, the maximum dive depth of a seal can be greater than the bathymetric depth of the model. To address this problem, the
points below the bathymetric surface were corrected to the sound pressure level at 5 m above the bathymetric seabed. For time periods when the seal was ‘cruising’ (i.e. the device is wet but above 1.5 m). The seal was assumed to be receiving a level of noise equivalent to that at a depth of 1.5 m. There is very little variation in noise in this portion of the water column and noise levels were always low compared to the complete dive profile due to surface scattering effects.

Noise exposure has the potential to have a negative impact on auditory systems through permanent threshold shift or temporary threshold shift, as well as instigate maladaptive behavioural or physiological responses. Consequently, this study reports two cumulative sound exposure values, $cSEL_{24}$ and $cSEL_{eq}$, total m-weighted 24-hr cumulative sound exposure levels and m-weighted 24-hr cumulative sound exposure levels above effective quiet respectively. Cumulative levels were both calculated by integrating received sound exposure levels over a 24 hour period.

$cSEL_{24}$ represents the total contribution of shipping noise perceivable by seals to the soundscape (given the limitation in AIS data) and includes sound pressure levels (SPL) emitted by ships that, while may not be at an intensity to cause auditory damage, may be pertinent in assessing behavioural responses to noise levels or when assessing the contribution of shipping to the wider soundscape. $cSEL_{eq}$ were calculated by removing SPL values below the estimated level of effective quiet for grey seals, 124 dB re 1 $\mu$Pa (Finneran 2015). Effective quiet can be defined as the exposure levels which do not result in TTS nor retard the recovery of TTS from a previous exposure (Ward et al. 1976). It recognises that some noise exposures are at a level that no matter how long the exposure lasts it will never result in TTS (Ward et al. 1976). It is important to consider the effective quiet threshold when calculating cumulative sound exposure levels because accumulating sound over longer and longer durations may result in inflated noise levels (Finneran and Branstetter 2013). However, there is very little data on appropriate levels of effective quiet in marine mammals (Finneran 2015).
Hence the value used here is an estimated value generated from considering the lowest value known to cause TTS in pinnipeds (Finneran 2015). $cSEL_{eq}$ were compared to a best estimate value of 183 dB re $1\mu Pa^2s$ for the onset of TTS in pinnipeds (Southall et al. 2007).

3.2.8 Statistical analysis of shipping traffic

The relative influence of ship source levels, distance and the number of ships on the calculated ship noise exposure levels was analysed using a Generalised Additive Mixed Model (GAMM). The GAMM model allows for non-linear relationships between the response variable and the explanatory variables. In additive modelling this is achieved by using a smoothing function (Zuur 2009). The mean of the response is dependant on the explanatory variables through the sum of the smooth terms, and random effects are used to model correlation between observations (Lin and Zhang 1999; Wood 2006). The response variable, $cSEL$ in each 15 minutes, was modelled using the explanatory variables, closest point of approach of a ship (CPA), the maximum source level of any ship in the 15 minutes ($SL_{max}$), the number of ships within 120 km of the seal for those 15 minutes (NUM) and the location of the seal (English Channel or Celtic Sea). CPA, NUM and $SL_{max}$ were included in the model as individual smooths as well a multivariate smoothed term using tensor product smooths of cubic regression splines (Wood 2006). This was appropriate because each covariate was not isotropic (i.e. they did not have the same scale) and allowed the inclusion of each variable as a main effect and the interaction between the three variables, CPA, NUM and $SL_{max}$ (Wood 2006). The GAMM models were implemented in R version 3.5.3 (R Core Team 2019) using the mgcv package version 1.8-28 (Wood 2003; Wood 2004; Wood 2006). The models were implemented using a Gaussian error structure with an identity link function. The response variable was log transformed ($log(y)$) to improve the normality of the residuals. If there were no ships in a 15 minute section this was removed from the analysis because there would be no predicted noise and it was not possible
to generate a valid value for the explanatory variables, closest point of approach or maximum ship source level.

The random variable seal was included to account for different exposure levels as a result of individual seal behaviour. Each 15 minute sample was highly autocorrelated because it was likely to contain the same ships as those before and after it. As a result, the data was subsampled and every 10th 15 minute section was included in the model. This resulted in predicted values every 150 minutes. Examination of AIC values revealed that a spherical correlation structure \( \text{cor Spher(form} = \sim 1|\text{seal}) \) dealt with remaining autocorrelation between the residuals appropriately. Model selection was completed using Akaike’s Information Criterion (AIC) and followed the methodology laid out by Zuur (2009) by first creating a model with all variables, determining the random structure that gave the lowest AIC and then determining the optimum fixed effects structure by removing variables and comparing AIC values. AIC was given by \(-2\log \text{likelihood} + 2k\) where k is the number of parameters. The lowest AIC values are considered superior (Zuur 2009). The criterion assesses goodness of fit but penalises models for increasing complexity by including a penalty based on the number of parameters. Model validation was completed by visual inspection of the residuals.

### 3.2.9 Sensitivity analysis

There are a number of sources of uncertainty associated with modelling shipping noise. Uncertainty can be assessed by repeating the execution of the model and drawing input values from a distribution that characterises the error associated with a model input. However, there is very little information on appropriate error distributions and uncertainty for model inputs utilised here. As a result, a deterministic one-way sensitivity analysis was conducted for model inputs and the structural composition of some coding decisions (e.g. the size of area around the seal track noise). This analysis demonstrates how deviation in these values would influence the final exposure values and is presented in the
absence of appropriate validation data. The sensitivity analysis was conducted using 12, 15 minute sections to ensure good representation of different noise levels. Six of these were from the English Channel area and six were from the Celtic Sea area. A selection of two 15 minute sections representative of high SPL, low SPL and median SPL were selected.

The input variations are shown in Table 3.6. The amount by which to vary each parameter was chosen from the literature (Table 3.6). Variations were made individually and compared to baseline values. Where there is no baseline or new value listed in the table, this parameter can take on many possible values (e.g. temperature profiles to calculate sound speed) and all possible values were changed by the deviation value listed in the table. An assessment of uncertainty in estimated $cSEL_{24}$ based on the sensitivity analysis is presented in Supplementary Material A.3.
Table 3.6: Details of parameter variations undertaken for sensitivity analysis. Where no baseline values are included, the parameter can take on many different values, and therefore, each value was varied by the deviation value. The rationales for the magnitude of each deviation are shown in footnotes at the end of the table.

<table>
<thead>
<tr>
<th>Name</th>
<th>Deviation in Parameter</th>
<th>Baseline</th>
<th>New value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number of transects (Azimuth)</td>
<td>-2</td>
<td>2.5</td>
<td>0.5</td>
</tr>
<tr>
<td>Number of transects (Azimuth)</td>
<td>-1.5</td>
<td>2.5</td>
<td>1.0</td>
</tr>
<tr>
<td>Number of transects (Azimuth)</td>
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<td>2.5</td>
<td>3.0</td>
</tr>
<tr>
<td>Width of Area (Total Azimuth)</td>
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<td>2.5</td>
<td>5.0</td>
</tr>
<tr>
<td>Width of Area (Total Azimuth)</td>
<td>-1.5</td>
<td>2.5</td>
<td>1.0</td>
</tr>
<tr>
<td>Width of Area / Number of transects</td>
<td>2.5/-2.0</td>
<td>2.5/2.5</td>
<td>5.0/0.5</td>
</tr>
<tr>
<td>Width of Area / Number of transects</td>
<td>2.5/-1.5</td>
<td>2.5/2.5</td>
<td>5.0/1.0</td>
</tr>
<tr>
<td>Width of Area / Number of transects</td>
<td>2.5/0.5</td>
<td>2.5/2.5</td>
<td>5.0/3.0</td>
</tr>
<tr>
<td>Width of Area / Number of transects</td>
<td>-1.5/-2.0</td>
<td>2.5/2.5</td>
<td>1.0/0.5</td>
</tr>
<tr>
<td>Width of Area / Number of transects</td>
<td>-1.5/-1.5</td>
<td>2.5/2.5</td>
<td>1.0/1.0</td>
</tr>
<tr>
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<td></td>
</tr>
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<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Salinity (psu)</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Salinity (psu)</td>
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<td></td>
<td></td>
</tr>
<tr>
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<td></td>
<td></td>
</tr>
<tr>
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<td></td>
</tr>
<tr>
<td>Sediment Velocity (ms(^{-1}))</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Sediment Velocity (ms(^{-1}))</td>
<td>-15</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sediment Attenuation (dB(^{-\lambda}))</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Sediment Attenuation (dB(^{-\lambda}))</td>
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<td></td>
<td></td>
</tr>
<tr>
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<td></td>
<td></td>
</tr>
<tr>
<td>Source Level (dB)</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Bathymetry (m)</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Bathymetry (m)</td>
<td>-5</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Temporal Resolution (min)</td>
<td>-10</td>
<td>15</td>
<td>5</td>
</tr>
</tbody>
</table>

1 Errors in temperature and salinity data are taken from errors in model as reported by CMEMS Quality Information documentation here: http://marine.copernicus.eu/documents/QUID/CMEMS-IBI-QUID-005-002.pdf. In addition a 10°C increase in temperature and 2 psu increase in salinity were calculated to show upper bounds in the case of extreme errors in the data.

2 As shown by Hamilton (1980)

3 Approximate mean differences in reported values by McKenna et al. (2012) and the RANDI source model (Supp. Mat. A.2).

4 Approximate absolute mean difference in the maximum depth of seal dives and the depth given by uncorrected bathymetry data calculated using the bathymetry and seal tag data in this study.
3.3 Results

3.3.1 Shipping traffic and seals

The sound exposure levels of adult grey seals in the English Channel and grey seal pups in the Celtic Sea varied as they moved throughout their environment, particularly, lower received noise levels resulted from scattering and absorption at the boundaries with the surface and bottom of the ocean (Fig. 3.6). Spatial variation in received noise levels was mainly driven by the number of ships, the source level of the ships and the distance between the seal and the ship. The mean number of ships processed in a 15 minute period within 120 km was 16.5 (SD = 20.6). This was higher for the English Channel group at 26.9 (SD = 24.5) ships and lower for the Celtic sea group at 6.5 (SD = 7.2) ships. However, there was very little difference between the mean number of ships within 5 km of the seal during a 15 minute period (Fig. 3.7a). This was only 1.1 for the Celtic Sea and 1.3 for the English Channel. Over the first 25 km from the seal there is very little difference between the number of ships contributing to the received noise levels of either group of seals (Fig. 3.7a). However, the mean number of ships between 25 and 100 km was higher for the English Channel compared to the Celtic Sea, highlighting the overall busier nature of the English Channel area. It is of note that, ships were included in the calculations if they were located within a circle of radius 120 km centred at the location of the seal. The area of this circle is given by $A = \pi r^2$. Each 5 km circle segment is given by $A = (\pi r^2) - (\pi (r - 5)^2)$. Due to the nature of this relationship, the area of each increasing circle segment is larger. Therefore, there may inherently be more ships further away from the seal because there is a bigger area over which to count ships.
Figure 3.6: Broadband sound exposure levels for (a) Seal B23 in the English Channel and (b) Seal hg29-11-10 in the Celtic Sea along their three-dimensional track for 24 hours. Note different noise scales for each seal. The Celtic Sea seal experiences lower noise levels than the English Channel seal, and noise levels are lower at the surface and bottom of dives in some places.
The closest point of approach (CPA) for any ship was 161 m for the English Channel seals and 535 m for the Celtic Sea seals. The majority of 15 minute sections had a CPA below 35 km (Fig. 3.7b). The English Channel seals had a greater number of close approaches, whereas ships in the Celtic Sea were generally not as close to the seals. A proportion of ships (24%) have a CPA above 50 km, and therefore, made smaller contributions to the overall received noise levels.

![Diagram](image_url)

**Figure 3.7:** (a) The mean number of ships in a 15 minute period at each 5 m radius from the location of the seal and (b) the closest point of approach between the seal and ships in a 15 minute period.

The source levels of ships included in the predictions were greater in the English Channel (Median = 176 dB re $1\mu Pa$, Inter-Quartile Range (IQR) = 46 dB re $1\mu Pa$) than the Celtic Sea (Median = 170 dB re $1\mu Pa$, IQR = 34 dB re $1\mu Pa$). This difference is even more stark when only considering those ships that were within 5 km of the seal. The median source level (SL) in the English Channel was 177 dB re $1\mu Pa$ (IQR = 30 dB re $1\mu Pa$) but this was only 154 dB re $1\mu Pa$ (IQR = 20 dB re $1\mu Pa$) in the Celtic Sea. This is most likely mediated by seal habitat use. Seals included in the study in the Celtic Sea, generally utilise areas located further from the major shipping lanes (Fig. 3.1 and 3.2).

The relationship between 15 minute cumulative sound exposure levels ($cSEL_{15}$),
the closest point of approach of a ship (CPA), maximum ship source level ($SL_{max}$) and the number of ships within 120 km of the seal (NUM) in that 15 minutes was modelled using a GAMM. The model, following stepwise model selection using AIC, included the multivariate smooth of CPA, NUM and $SL_{max}$, as well as, the main effect smooths of $SL_{max}$ and CPA as significant explanatory variables (Table. 3.7). It did not include location or the number ships as an individual smooth (Table. 3.7). The $cSEL_{15}$ decreased as the closest point of approach increased, and $cSEL_{15}$ increased as the maximum ship source level increased (Fig. 3.8). As the closest point of approach increased, noise remained constant if the maximum source level increased and or the number of ships increased (Fig. 3.9). This relationship did not differ between the Celtic Sea or English Channel. However, in the Celtic Sea there are fewer 15 minute sections with high numbers of ships, a close approach and high $SL_{max}$ than the English Channel (Fig. 3.9). Model validation plots are included in Supplementary Material A.4 and show the residuals and autocorrelation was appropriately modelled.

Table 3.7: The structure of the maximal model with all explanatory variables and each model tested during model selection for the response variable 15 minute cumulative sound exposure level.

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>$R^2$ (adj)</th>
<th>AIC</th>
<th>$\Delta$ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>A: Full $^1$</td>
<td>15</td>
<td>0.66</td>
<td>-1242</td>
<td></td>
</tr>
<tr>
<td>B: Full - Location $^2$</td>
<td>14</td>
<td>0.64</td>
<td>-1248</td>
<td>-6</td>
</tr>
<tr>
<td>C: B - NUM $^3$</td>
<td>12</td>
<td>0.631</td>
<td>-1250</td>
<td>-2</td>
</tr>
<tr>
<td>D: C - CPA $^4$</td>
<td>10</td>
<td>0.606</td>
<td>-1188</td>
<td>62</td>
</tr>
</tbody>
</table>

\[ ^1 \log(cSEL_{15}) \sim t(i(SL)) + t(i(num)) + t(i(CPA)) + t(i(CPA, NUM, SL)) + location + (1 | seal) + corSpher(1 | seal) \]
\[ ^2 \log(cSEL_{15}) \sim t(i(SL)) + t(i(num)) + t(i(CPA)) + t(i(CPA, NUM, SL)) + (1 | seal) + corSpher(1 | seal) \]
\[ ^3 \log(cSEL_{15}) \sim t(i(SL)) + t(i(CPA)) + t(i(CPA, NUM, SL)) + (1 | seal) + corSpher(1 | seal) \]
\[ ^4 \log(cSEL_{15}) \sim t(i(SL)) + t(i(CPA, NUM, SL)) + (1 | seal) + corSpher(1 | seal) \]

The relationship between CPA, NUM and $SL_{max}$ can be examined more closely in Figure 3.10 and Figure 3.11, which also show the distance between a seal and the ships that were included in the soundscape calculations. Figure 3.10 shows three peaks in sound pressure levels greater than 120 dB re $1 \mu Pa$ just before 12:06, at 14:53, and between 23:13 and 02:00. The high noise levels at the seal are mediated by the source level of the ship, how close the ship came to
the seal and the number of ships. Just before 12:06 at Peak 1 a loud ship (> 190 dB re 1\(\mu\)Pa) is close to the seal. At Peak 2 just after 14:53, the ships are further away from the seal than during Peak 1 but there is a second loud ship and the presence of a quieter ship (< 170 dB re 1\(\mu\)Pa) in the area, which results in similar overall noise levels at Peak 1 and Peak 2. The peak in noise between 23:13 and 02:00 has a high number of different ships, which result in sustained noise levels across the time despite variation in traffic. At 20:26 a loud ship results in higher noise levels, just before this, a ship follows an almost identical path to the ship at 20:26 but the lower source level of the ship results in lower noise levels.

**Figure 3.8:** Generalised additive mixed model smoothing curves representing the modelled non-linear effect of (a) closest point of approach and (b) source level on cumulative sound exposure level in 15 minutes. Rug plot and points indicate data points and their distribution. Blue dotted lines show 95% confidence intervals. There are few data points below 160 dB re 1\(\mu\)Pa leading to wide confidence intervals and suggesting that maximum source levels in each 15 minutes were generally above 160 dB re 1\(\mu\)Pa.

This pattern is also reflected in an example seal from the Celtic Sea (Fig. 3.11). However, overall noise levels are much lower than the English Channel seal because there are fewer ships and these are further away from the seal. Maximum sound pressure levels are approximately 110 dB re 1\(\mu\)Pa and minimum approach distances between the seal and ships are all above 25 km. Despite the seal and ships being further apart as the day goes on noise levels increase. For example, at 18:13 a loud ship is present resulting in higher noise levels than before 12:40 despite the ship being further away.
Figure 3.9: The cumulative sound exposure level in 15 minutes ($cSEL_{15}$) given the number of ships, the closest point of approach for a single ship and the maximum ship source level in that 15 minute period in (a) the English Channel and (b) the Celtic Sea. Note different colour scales. The Celtic sea has fewer of the high noise scenario data points with high source levels, a close approach and high ship numbers.
Figure 3.10: Distance between the seal and each ship (a) and the sound exposure levels for Seal B23 in 24 hours (b). The source level of each ship is classified to show the loudest ships. The noise levels are a reflection of the number of ships, distance between seal and ships, and the source level of each ship.

The source level of each ship is classified to show the loudest ships. The noise levels are a reflection of the number of ships, distance between seal and ships, and the source level of each ship.

Time of day starts at 29th October 2011 05:05:00. Black line indicates bathymetry. Dives below the bathymetry arise from bathymetry referenced to mean seal level, the seal diving throughout the tidal cycle and location error.
Figure 3.11: Distance between the seal and each ship (a) and sound exposure levels for Seal hg29-11-10 in 24 hours in the Celtic Sea (b). The source level of each ship is classified to show the loudest ships. The noise levels are a reflection of the number of ships, distance between seal and ships, and the source level of each ship. Time of day starts at 21st June 2011 at 11:40:00. Black line indicates bathymetry. Dives below the bathymetry arise from bathymetry referenced to mean seal level, the seal diving throughout the tidal cycle and location error.
3.3.2 Cumulative sound exposure levels

The cSEL$_{24}$ ranged from 121 to 170 dB re 1 µPa$^2$s for all seals for a total of 86 days (Fig. 3.12). Median cSEL$_{24}$ for all seals was 149 dB re 1 µPa$^2$s. Median cSEL for the Celtic Sea pups was 143 (121 - 156) dB re 1 µPa$^2$s and 159 (134 - 170) dB re 1 µPa$^2$s for the English Channel adults. These values represent the total exposure of seals to shipping noise during these 24 hour periods. However, SPL values throughout the 24 hours ranged from 0 to 140 dB re 1 µPa with a median maximum SPL value for a day of 115 dB re 1 µPa.

In order to assess if temporary threshold shift (TTS) could occur in the seals, cSEL$_{eq}$ was also calculated using only exposures to SPL greater than or equal to the value of effective quiet (124 dB re 1 µPa) in a 24 hour period (Fig. 3.13). As would be expected, the number of days with 24-hr cSEL above zero decreases dramatically from 86 to 18. Mean exposure duration above effective quiet was 38.57 (SD = 47.86) minutes (Tbl. 3.8). This is considerably less than the standard 24-hr accumulation period used in policy documentation (National Marine Fisheries Service 2018). All but one of the days with an exposure level above effective quiet were for seals in the English Channel. cSEL$_{eq}$ ranged from 141 to 169 dB re 1 µPa$^2$s with a median value of 154 dB re 1 µPa$^2$s although for the majority of days, 68 of 86, the cSEL$_{eq}$ was zero (Fig. 3.13; Tbl. 3.8). The values did not exceed the threshold of 183 dB re 1 µPa$^2$s for the onset of TTS (Southall et al. 2007).

3.3.3 Sensitivity analysis

The factors that resulted in the greatest deviation from baseline cSEL values were the source level, smoothing method, sediment type and total azimuth of transects for three-dimensional noise calculations (Fig. 3.14). Median absolute deviation due to a reduction in the total azimuth (i.e. area over which noise was calculated) was less than 1 dB (Fig. 3.14a). However, for some conditions, deviation was as large as 20 dB in the 15 minute cSEL because of a sharp decrease in the number of transects that could be calculated within the total
azimuth. For example, following the methodology shown in Figure 3.4, if the total area is 10° then 4 transects can be calculated in this area separated by 2.5°. If this is reduced to 7° only 2 transects separated by 2.5° can be calculated to characterise the noise field. However, increasing the total azimuth (i.e. area over which noise is calculated) resulted in < 2.5 dB changes (median = 0) in 15 minute cSEL indicating that the area over which noise was calculated in this study was sufficiently fine scale to capture the sound field.

A 10 dB increase or decrease in the ship source levels resulted in a corresponding increase/decrease in the overall 15 minute cSEL (Fig. 3.14b). As a result, source level errors are likely to be a large contributor to any uncertainty in the noise calculations. To assess the impact of sediment type on the output cSEL values, sediment type was randomly selected for 5 trials and the cSEL values compared to baseline values for which sediment type was determined by the EMODnet Geology project seabed substrate map. Median deviation from

Figure 3.12: The m-weighted 24-hr cumulative sound exposure levels (cSEL_{24}) for adult seals in the English Channel and pups in the Celtic Sea. A total of 86 days were processed for 9 adult seals and 9 seal pups.
Figure 3.13: Histogram of 24-hr cumulative sound exposure levels (cSEL) without considering effective quiet and given a value of effective quiet equal to 124 dB re 1µPa. All SPL indicates all SPL values are included in the calculation of cSEL over 24 hours. SPL > 124 includes all SPL over 124 dB in the calculation of cSEL. For the All SPL condition all days have cSEL above 0. However, for the SPL > 124 condition 68 days have cSEL equal to 0 (bar not shown on graph).

Baseline cSEL was 2.9 (0.1 - 6.8) dB (Fig. 3.14c). The incorrect classification of sediment was likely to arise due to location uncertainty when there was a long time between dives and GPS location fixes in the seal telemetry data, and because the resolution of sediment data can be patchy with very little information in some areas and fine-scale data in other regions (See Fig. 5.3. However, the errors may not be as extreme as given by random sediment selection because there is probably not an equal chance that the sediment would be classified as any one of the five categories but more likely that it would be classified as a sediment close to its location. However, even if sediment type is selected correctly uncertainty in the density, attenuation and velocity parameters must also be considered.

The RAMSurf model is run at individual frequencies and coherent interference effects result in fluctuations in the amplitude of the output propagation loss.
Table 3.8: The m-weighted 24-hr cumulative sound exposure levels (cSEL) for a value of effective quiet (EQ) at 124 dB re 1 µPa including the number of minutes sound pressure levels were above the given value of effective quiet.

<table>
<thead>
<tr>
<th>Seal</th>
<th>Maximum SPL (dB re 1 µPa)</th>
<th>cSEL above EQ (dB re 1 µPa² s)</th>
<th>Seconds above EQ</th>
<th>Approx. number of minutes above EQ</th>
</tr>
</thead>
<tbody>
<tr>
<td>B31</td>
<td>133</td>
<td>162</td>
<td>2059</td>
<td>34</td>
</tr>
<tr>
<td>B31</td>
<td>134</td>
<td>168</td>
<td>10609</td>
<td>176</td>
</tr>
<tr>
<td>B32</td>
<td>126</td>
<td>152</td>
<td>536</td>
<td>9</td>
</tr>
<tr>
<td>B32</td>
<td>124</td>
<td>142</td>
<td>58</td>
<td>1</td>
</tr>
<tr>
<td>B35</td>
<td>130</td>
<td>159</td>
<td>1439</td>
<td>24</td>
</tr>
<tr>
<td>B37</td>
<td>126</td>
<td>158</td>
<td>2251</td>
<td>38</td>
</tr>
<tr>
<td>B32</td>
<td>126</td>
<td>153</td>
<td>584</td>
<td>10</td>
</tr>
<tr>
<td>B27</td>
<td>131</td>
<td>162</td>
<td>3146</td>
<td>52</td>
</tr>
<tr>
<td>B23</td>
<td>126</td>
<td>154</td>
<td>716</td>
<td>12</td>
</tr>
<tr>
<td>B24</td>
<td>130</td>
<td>164</td>
<td>6423</td>
<td>107</td>
</tr>
<tr>
<td>B24</td>
<td>125</td>
<td>141</td>
<td>49</td>
<td>0.8</td>
</tr>
<tr>
<td>B31</td>
<td>126</td>
<td>153</td>
<td>708</td>
<td>12</td>
</tr>
<tr>
<td>B33</td>
<td>140</td>
<td>169</td>
<td>5383</td>
<td>90</td>
</tr>
<tr>
<td>B33</td>
<td>128</td>
<td>156</td>
<td>758</td>
<td>13</td>
</tr>
<tr>
<td>B33</td>
<td>125</td>
<td>147</td>
<td>191</td>
<td>3</td>
</tr>
<tr>
<td>B33</td>
<td>138</td>
<td>168</td>
<td>5420</td>
<td>90</td>
</tr>
<tr>
<td>B37</td>
<td>126</td>
<td>153</td>
<td>714</td>
<td>12</td>
</tr>
<tr>
<td>hg29-24-10</td>
<td>126</td>
<td>154</td>
<td>610</td>
<td>10</td>
</tr>
</tbody>
</table>

(Wang et al. 2014). This can be addressed by averaging over frequency for broadband signals but this can be time consuming. The sensitivity analysis compared two alternative methods for smoothing the output at individual frequencies; the use of a Butterworth filter (Gervaise et al. 2012) and averaging over range (Harrison and Harrison 1995). The median absolute difference between the two methods was 1.9 (0.1 - 16.2) dB (Fig. 3.14c). A moving average method was used as part of this study because the Butterworth filter over-smoothed some areas resulting in poor representation of the bathymetric floor. It was difficult to select parameters for the Butterworth filter that suited all different frequencies. It highlights that when modelling noise, the method of smoothing applied should be appropriate.

The deviations from baseline of other parameters tested were less than 2 dB. This includes changes as a result of calculating the sound field at a finer temporal resolution (Fig. 3.14c). If noise is calculated every 5 minutes rather than every 15 minutes the resulting median deviation from the baseline value
was only 0.9 dB. The results for all other parameters are shown in Supplementary Material A.5. The sensitivity analysis has been conducted using variations in parameter values of a magnitude given the literature. However, there is very little information on the true error profiles expected for many of the parameters. Therefore, changes in parameters of a magnitude greater or less than considered here could result in higher or lower deviations in the results than expected. Estimated uncertainty in the cumulative sound exposure levels given the results of the sensitivity analysis are shown in Supplementary Material A.3. The interquartile-range of predicted $cSEL_{24}$ given estimated uncertainty was between 3 and 7 dB for all seals.

**Figure 3.14:** Median absolute deviation from baseline cSEL values by (a) increases and decreases in the total width of area (total azimuth) over which noise was calculated (b) changes in source level and (c) given 5 trials of random sediment type selection, given a Butterworth filter compared to a moving average smoothing method and the difference in a temporal resolution of 15 or 5 minutes.
3.4 Discussion

This study presented predictions of the cumulative sound exposure levels for 18 seals over 86 days given the three-dimensional at-sea behaviour of individual seals. For all seals, maximum $cSEL_{24}$ was 170 dB re $1\mu Pa^2s$ and median $cSEL_{24}$ was 149 dB re $1\mu Pa^2s$. For pups primarily located in the Celtic Sea, median $cSEL_{24}$ was 143 dB re $1\mu Pa^2s$ and for adults primarily located in the English Channel median $cSEL_{24}$ was 159 dB re $1\mu Pa^2s$. It is not possible to give direct comparisons between the two areas or between the adults and pups because data were only available for pups in the Celtic Sea region and adults in the English Channel region confounding any possible comparative analysis. However, given the results presented here, it is reasonable to assume that differences in shipping activity (CPA, NUM, $SL_{max}$) is a driver of differential noise exposure in the two groups. Merchant et al. (2016) highlighted that 125 Hz octave band noise the south-eastern Celtic Sea was quieter than Falmouth Bay in the English Channel, and noted it as one of the quietest regions compared to locations in the North Sea. The mean $cSEL_{24}$ recorded using a hydrophone in Falmouth Bay and weighted using the same m-weighting curve for pinnipeds was $156 \pm 19.1$ dB re $1\mu Pa^2s$; a remarkably similar match to average exposure for seal in the English Channel (Merchant et al. 2012). The seals occupy water south-west of Falmouth Bay in busier and therefore, noisier waters but their occupation of these waters is temporary because they are transiting through the area unlike the stationary hydrophone in Falmouth Bay. The results are also between 20 and 36 dB lower than $cSEL_{24}$ values reported for harbour seals in the Moray Firth (Jones et al. 2017). The disparities between the results presented here and those by Jones et al. (2017) could arise from differences in the propagation model used by Jones et al. (2017), the two-dimensional modelling approach, and the wider frequency range (12.5 Hz to 20 kHz) studied by Jones et al. (2017). In addition, Jones et al. (2017) studied harbour seals, which do not travel as far from haul-out sites (Thompson et al. 1996), and therefore, may be more resident in areas of high shipping traffic. However, the
results highlight spatial variation in noise patterns and shipping traffic in different regions. It provides evidence that confirms regional variations must be considered carefully in underwater noise management plans.

SPL values ranged from 0 to 140 dB re $1\mu Pa$ and median maximum SPL in a day was 115 dB re $1\mu Pa$. Ambient noise levels in the region were not available as part of this study but measurements by Merchant et al. (2016) at one location in the Celtic Sea suggested median noise levels to be 83.3 dB re $1\mu Pa$ at 125 Hz. In the English Channel recordings from Falmouth Harbour measured SPLs between 86.1 and 148.6 dB re $1\mu Pa$ and the mean threshold level (minimum recorded level in a period – representative of background noise levels) was 96.2 dB re $1\mu Pa$ (Merchant et al. 2012). These values suggest that the seals are exposed to shipping noise above that which could be considered ambient levels in both the Celtic Sea and English Channel. However, the estimated level of effective quiet for grey seals is 124 dB re $1\mu Pa$ and the median SPL value remained below this for many of the seals.

The cSEL in 15 minutes was closely related to the number of ships, the closest point of approach of any ship and the source level of the loudest ships in that 15 minutes. For example, loud ships over 50 km from the seal resulted in received noise levels greater than 100 dB re $1\mu Pa$ for a seal in the Celtic Sea (Fig. 3.11). These exposures may be indistinguishable from ambient noise for seals but they will raise the overall ambient noise levels and may be of concern for issues such as call masking and chronic stress related to sustained exposure (Rolland et al. 2012). Ship noise exposure detectable above ambient noise levels will be most relevant for determining auditory damage and possible behavioural responses to noise, and these generally arose from ships closer to the seal and with higher source levels. However, the results demonstrated the ability of high numbers of loud ships far away from the seal to generate high noise exposure levels at the seal’s location. This suggests that when assessing the impacts of shipping noise, the area over which ships are included in calculations of noise levels should be sufficiently wide to capture such exposure and not just focus on the
first few kilometres from the seal (Mikkelsen et al. 2019).

In addition to shipping traffic alone, difference in behaviour between English Channel adults and Celtic Sea pups as a result of age or location specific factors such as bathymetry may also be mediating noise exposure in the two groups. Figure 3.1 shows that the seals in the Celtic Sea used in this study were mainly located to the north of the region where shipping density is lower. English Channel seals cross an area of very high intensity shipping. However, compared to their whole track they tend to make this crossing only once or twice, and visual inspection of the track suggests they are undertaking directed travel through the area. The majority of their time was spent around the islands within the Iroise Marine Park. The noise levels in this area are unknown but is likely to be different as a result of lower numbers of large ships. Huon et al. (2015) studied 19 seals, 9 of which are included here, and found that individuals spent 67% of their time within the Marine Park. Harbour seals in the Moray Firth, which experience much higher cumulative noise levels, also tend to remain close to the coast. However, they are resident within the zones of higher intensity shipping (Jones et al. 2017). This could account for their higher exposure.

Southall et al. (2007) proposed pinnipeds will experience TTS for non-impulsive sounds such as shipping noise when exposures over 24 hours exceeds 183 dB re 1\(\mu\)Pa\(^2\)s and the National Marine Fisheries Service (2018) revised this value to 181 dB re 1\(\mu\)Pa\(^2\)s based on expanded audiogram data. The exposure of seals above effective quiet in the study did not exceed these threshold values. Only 9 seals for a total of 18 days experienced SPL greater than the values of effective quiet. The \(cSEL_{eq}\) values range from 141 to 169 dB re 1\(\mu\)Pa\(^2\)s and as such are between 12 and 40 dB below the lowest threshold level for auditory damage. These thresholds are based on audiograms for pinnipeds, or in the case of National Marine Fisheries Service (2018) values, considers audiograms for phocid seals alone. These include a limited number of harbour seals and northern elephant seals aged between 4 and 14, and are based on exposure to a limited range of frequencies (Finneran 2015; Erbe et al. 2015). However, there
are no audiograms specifically for grey seals or grey seal pups (Finneran 2015; Erbe et al. 2015). Pups may be more sensitive to noise and future work is necessary to explore the sensitivity of animals in this vulnerable juvenile stage, as well as test exposures to a wider range of frequencies.

Temporary threshold shift is determined by exposure frequency, duration, SPL, temporal pattern of noise and available recovery time. Kastak and Schusterman (1999) found average threshold shift of 4.8 dB given exposure for 20 minutes at 100 Hz to SPLs ranging from 133 – 156 dB re 1\textmu Pa. These conditions were met three times in this study. Many studies of TTS growth and recovery in phocid seals examined frequencies higher (2.5 - 4 kHz) than the peak shipping noise used in this study (10-1000 Hz) and higher SPL values than seals were exposed to in these calculations. Kastelein et al. (2012) tested the hearing of two harbour seals using octave band noise at a centre frequency of 4 kHz. They showed maximum TTS of 10 dB 1 – 4 minutes after a 120 minute exposure to 148 dB re 1\textmu Pa. TTS began to occur at SPLs of 136 dB for 60 minutes. This suggests any one of the properties (exposure frequency, duration etc.) determining TTS should be closely monitored for changes that may result in exposures great enough to induce TTS. If noise levels increase as projected (Frisk 2012; Kaplan and Solomon 2016), it will be necessary to take measures to ensure that the rising noise is curbed before it can lead to auditory damage but this should be considered in combination with exposure duration and frequency.

As such, cumulative sound exposure levels are often considered for regulatory assessments because the metric considers the duration of exposure as well as SPL and frequency (Finneran and Branstetter 2013). The standard duration of exposure for non-impulsive sounds such as shipping noise has been 24 hours (National Marine Fisheries Service 2018; Southall et al. 2007). However, it is recognised that this is an arbitrary period over which to assess noise exposure for many marine mammals. If a species shows high site fidelity at a high exposure zone they may be exposed for much longer than 24 hours. Alternatively, individuals may move in and out of high exposure zones.
Particularly, for sources such as ships that are highly mobile, peaks in noise may be quite short and an individual may have periods where shipping noise could be zero. The development of a more ecologically relevant value is key for future policy and management of noise (National Marine Fisheries Service 2018). Seals spend time at-sea between periods of haul-out, therefore, the duration over which seals are potentially exposed to underwater noise varies and supports the assertion that the accumulation period appropriate for a specific species or noise source will vary. In addition, mean length of exposures above effective quiet in 24 hours was 38.47 minutes. The cSEL metric assumes the ‘equal energy’ hypothesis, where by exposures of equal energy are assumed to result in the same amounts of threshold shift regardless of how the exposure is distributed in time (Finneran and Branstetter 2013). It is well known that the equal-energy approach overestimates intermittent exposures because it does not consider the recovery that can occur from TTS between the noise exposures within the total accumulation period. Hence, for seals, a continuous accumulation period of 24 hours, as used in this study, may result in higher levels of TTS than if periods of recovery are included.

In addition to possible auditory damage, behavioural responses and physiological responses have been recorded for a number of marine species to shipping noise (Blair et al. 2016; Celi et al. 2015; Rolland et al. 2012; Williams and Bain 2002). Seals have shown behavioural reactions such as entering the water, decrease in resting behaviour and increase in alert behaviour at the sight of approaching boats and boat noise playbacks when hauled out (Jansen et al., 2015; Tripovich et al., 2012). There is only limited anecdotal evidence of changes in the at-sea behaviour of seals in response to shipping noise (Mikkelsen et al. 2019). As such, acceptable exposure levels with respect to behavioural changes are unknown, and crucially, if there is a behavioural response, what level of behavioural response is harmful for individual survival and population stability (McHuron et al., 2017). The results show that seals are exposed to shipping noise and this is likely to be above ambient noise levels.
Therefore, further assessment of the behavioural responses of seals to this noise is warranted, especially, given the naivety of seal pups to underwater anthropogenic noise. This will be explored further in Chapter 4.

Exposure levels and at-sea spatial usage are key parameters in understanding the spatial risk of marine animals to shipping noise, and are required to set effective management targets (Erbe et al. 2014). The results can contribute to noise budgets and assessments of soundscapes, and close the gap to noise budgets that allow the establishment of quantitative targets that regulators can enforce. As described by Merchant et al. (2017) population density and noise exposure can be combined to provide risk maps. This is a similar approach as implemented by Erbe et al. (2012a). However, the majority of the distribution and noise based information is related to two-dimensional maps. The results presented here are the first assessment of noise exposure for seals using their three-dimensional dive track and adds the new dimension of depth to risk based assessment of noise levels for management goals. Utilising a three-dimensional model could also result in lower predicted cumulative values due to surface and bottom losses (Fig. 3.6), highlighting the importance of considering the three dimensional space use by marine animals, especially those that utilise the complete water column (Chen et al. 2017).

The results show a close relationship between noise levels, the closest point of approach and ship source levels. This highlights that to reduce noise levels, it is necessary to, not only reduce ship numbers, but also address the distance between ships and marine life, and ship source levels. Ship source levels can be altered by decreasing the speed of ships, proper maintenance, retrofitting quieting technologies as well as designing new ships to be quieter (IMO 2014). Source level had the greatest impact on noise levels and reducing the speed of ships is the simplest solution to achieve a reduction in source levels compared to engineering solutions. The technologies capable of exacting the most radical reductions in noise are most easily incorporated into new ship designs rather than retro-fitted to old ships (IMO 2014). These include careful hull design to
ensure uniform flow over the propellers to reduce cavitation, the resilient mounting of on-board machinery and the careful design of the propeller blades (IMO 2014). However, the average age of the current fleet is over 20 years old (UNCTAD 2018) suggesting it may be some years before new technologies make any impact on current underwater noise levels. A reduction in speed is a much more timely solution but noise levels may be influenced by longer transit times (Joy et al. 2019). The evidence suggests that current noise levels do not result in TTS for the seals in the English Channel and Celtic Sea but noise reduction technologies will also be important when considering possible behavioural changes, masking and chronic stress. Shipping noise is also increasing and if seals spent more time close to ships, ship numbers keep increasing and ship source levels are not improved, noise could potentially reach harmful levels.

The predictions presented in this study are subject to a number of limitations and uncertainties that were highlighted in Section 3.2. To examine the impact that uncertainties in the model may have on the resulting exposure value, a sensitivity analysis was conducted. It revealed that source-level is likely to be one of the greatest contributors to uncertainty in the modelling framework. The RANDI model is particularly useful because length and speed data are readily available. However, it was developed using a dataset of older ships. McKenna et al. (2013) demonstrated that for container ships speed does explain most variability in noise levels. However, there is evidence to suggest that this relationship is not as relevant to the modern fleet, primarily due to changes in the drive machinery of ships (McKenna et al. 2012; Wales and Heitmeyer 2002). For example, McKenna et al. (2012) found that vehicle carriers had lower source levels than tankers but were larger and travelled faster. This is most likely due to design factors such as a shallow draught and propeller depth. This variation is not captured by the RANDI model. As a result, ship source level models are a significant source of uncertainty when modelling shipping noise. The resulting error in output will be of the same magnitude as errors in the input source level
(i.e. 10 dB error in source level prediction would result in 10 dB output error).
The greatest level of error is associated with the newest, largest ships. A brief comparison of measured ship noise levels published by McKenna et al. (2012) and RANDI predicted source levels suggests mean error of 9.48 dB with maximum errors of 23.8 dB for the longest fastest ships (Supp. Mat. A.2). However, more complex models can be difficult to parameterise given the available data (Wittekind 2014) or represent average source levels (Wales and Heitmeyer 2002). Therefore, until a more accurate model, which can be easily parameterised, or more comprehensive experimental measurements from modern commercial ships are available, the RANDI model provided the best available predictions.

Furthermore, the error will only be enhanced by ships that are missing from the AIS data and incomplete transects in the AIS data. This error is not well characterised. However, a comparison of recorded data and AIS data in Falmouth Bay, UK was able to match 64% of peaks to AIS data resulting in 36% unidentified noise peaks (Merchant et al. 2012). This is similar to the Moray Firth where a comparison or recordings, AIS data and time lapse footage reported 38% of peaks in noise levels were from ships not present in the AIS data (Merchant et al. 2014a), and Mikkelsen et al. (2019) found one-third of ship noise events recorded by DTAGs attached to seals could not be linked to an AIS message. These results demonstrate that any noise modelling estimates based on AIS are conservative as a result of uncertainties from the input data.

Experimental data was not available to validate the complete workflow utilised here. However, this study used a sophisticated acoustic propagation model that has been benchmarked and compared to experimental data (Hanna and Rost 1981; Davis et al. 1982). In comparison to the basic model used by Jones et al. (2017) RAMSurf considers detailed representations of environmental properties that are particularly important in shallow water propagation scenarios. It has been highlighted that simple propagation models underestimate noise close to the source and overestimate noise far from the source when modelling pile
driving noise (Farcas et al. 2016). The uncertainties associated with the simple spreading model could account for some of the differences seen in noise exposure between the Moray Firth and the region of south-west UK considered here. Validation of the Jones et al. (2017) model alone suggests that median absolute error in the model was 9.75 (2.11 - 24.51) dB.

In summary, at-sea three dimensional exposure of grey seals ranged from 121 to 170 dB re $1\mu Pa^2s$. However, only 9 seals were exposed to SPLs greater than the estimated value of effective quiet for phocid seals, and these values ranged from 141 to 169 dB re $1\mu Pa^2s$. The exposure of seals to shipping noise did not exceed best evidence thresholds for TTS, despite a number of uncertainties associated with the calculations. The CPA of a ship to a seal was 161 m and on average 1.1 ships were within 5 km of the seal in the Celtic Sea and 1.3 in the English Channel. The exposure of the seals was mediated by the number of ships, CPA of these ships, maximum ship source level and the at-sea behaviour of the seals. The chapter presents vital data on the exposure of grey seals and the influence of shipping traffic behaviour on this exposure. This is central to our understanding of the risks posed by shipping noise and can inform marine spatial planning in the future. A major obstacle to concrete policy commitments on shipping noise is a lack of understanding of marine noise budgets, which characterise the contribution of different noise sources to the overall underwater soundscape (Merchant et al. 2017). Exposure values reported here contribute to such noise budgets by representing the total contribution of shipping to the seals soundscape.
Impact of shipping noise on the diving behaviour of grey seals in the Celtic Sea and English Channel
4.1 Introduction

The exposure of marine mammals to shipping noise, and the potential for this noise to cause auditory damage has been discussed in detail in the preceding chapters. However, evidence suggests that the possible impacts of underwater noise on marine mammals extends beyond this to include changes in vocalisations (Hatch et al. 2012), increases in the stress response (Rolland et al. 2012) and in extreme cases, mortality (Jepson et al. 2003). Furthermore, behavioural change can be a significant, non-lethal and ephemeral consequence of underwater noise. To date, the recorded behavioural responses of marine animals to underwater noise include, for example, startle responses (Tripovich et al. 2012), moving away from a sound source (Harris et al. 2001; Hastie et al. 2018) and displacement from critical habitats (Russell et al. 2016). Specifically, underwater noise from shipping has resulted in a decrease in the number of side-roll feeding events in humpback whales (*Megaptera novaeangliae*) (Blair et al. 2016), stereotyped porpoising behaviour from harbour porpoises (*Phocoena phocoena*) (Dyndo et al. 2015) and slower reaction times to predators in European eels (*Anguilla anguilla*) (Simpson et al. 2015). While some minor and transient responses may be of little consequence, the cessation or interruption of feeding, resting, travelling and socialising can have a direct impact on vital rates (growth rate, survival and reproductive success) which in turn influences population dynamics (Costa et al. 2016; Pirotta et al. 2018). It is, therefore, critical to understand if, and how, underwater noise from shipping drives behaviour change in order to safeguard individuals and at-risk populations. Especially, given the ubiquitous and chronic nature of shipping noise in the environment compared to other noise sources.

Behavioural responses can range from mild orienting responses that last for a short period of time, to intense panic and fleeing responses that can even result in death (Southall et al. 2007). The type and severity of a behavioural response is mediated by a range of factors relating to the context in which the exposure
occurs. These include the relevance of the noise to the animal (e.g. related to a predator) (Deecke et al. 2002; Curé et al. 2015), its behavioural state (Guerra et al. 2014), previous experiences (Gotz and Janik 2010) and age (Houser et al. 2013). In pinnipeds, motivation and experience are shown to be key factors in the response of individuals to underwater sounds (Gotz and Janik 2010). For example, juvenile California sea lions (*Zalophus californianus*) showed stronger reactions to lower received sound pressure levels than older seal lions in sonar playback scenarios (Houser et al. 2013). This may be particularly relevant to grey seals. After nursing, pups are abandoned on the breeding colony and undergo a terrestrial post-weaning fast (Noren et al. 2008). When they do go to sea they are under metabolic pressure to forage successfully but have no parental guidance in the development of an appropriate foraging strategy (Bennett et al. 2010; Carter et al. 2017). Consequently, they are highly motivated for food but naive to underwater noise exposure and could be vulnerable to disturbance. However, there is no assessment of the behavioural response of grey seals to at-sea shipping noise that considers the differential life experience of adults and pups.

Current understanding of the at-sea behavioural responses of pinnipeds to underwater noise is largely focused on the efficacy of acoustic deterrent devices. This literature has shown that seals exhibit behavioural responses to aversive noise exposures but that the responses can be diverse and based on the context of exposure (Götz and Janik 2013). Furthermore, seals have been shown to exhibit alert behaviour and flee from haul-out sites when disturbed by approaching boats or people (Andersen et al. 2012; Tripovich et al. 2012; Jansen et al. 2015), and land based observations of ringed seals at-sea have observed a stop and look reaction to construction sounds (Blackwell et al. 2004). A number of studies suggest that seals avoid noise sources. They are reported to swim away from seismic survey vessels (Harris et al. 2001), increase surfacing distance from an operating wind farm (Koschinski et al. 2003), and decrease habitat usage in areas during pile driving and tidal turbine operations.
As air breathing benthic foragers, diving is key to the life strategy of phocid seals (Boyd 1997). They dive repeatedly throughout the water column when travelling, foraging and resting (Baechler et al. 2002). Consequently, dive behaviour may be sensitive to disturbance from shipping noise and any changes in the behaviour may have important implications. Ultimately, the diving behaviour of seals is determined by their physiological abilities (Costa et al. 2004). They are constrained by two opposing forces; the need to forage at depth and the need to replenish oxygen reserves at the surface (Boyd 1997). Optimal diving behaviour minimises the physiological limitations through metabolic and behavioural strategies such as a reduction in heart rate and regulation of swim speed (Thompson and Fedak 1993; Boyd 1997). As a result, in addition to lost foraging opportunities, disturbance in the diving behaviour of seals may be associated with a metabolic cost as a result of changes in the timing of air-exchange (Thompson and Fedak 1993). It is suggested that pinnipeds can use plasticity in diving behaviour to account for environmental stressors but alterations in diving behaviour can have implications for foraging efficiency and energetics (Cornick et al. 2006; Atkinson et al. 2015). For example, during El Niño years, female northern elephant seals increased foraging trip duration to compensate for decreases in foraging success (Crocker et al. 2006). As a result, in moderate El Niño years there was negligible change in female mass gain. However, this could not compensate for severe El Niño years which saw a sharp decline in female mass gain, as well as a decrease in dive shapes that were related to foraging and an increase in dive shapes related to transit (Crocker et al. 2006).

The development of telemetry devices has provided detailed information on seal diving behaviour (Thompson et al. 1991; McConnell et al. 1999; Kuhn et al. 2010; Russell and McConnell 2014; Jessopp et al. 2013; Carter et al. 2016; Carter et al. 2017). Specifically, changes in diving behaviour have been detected in response to a number of anthropogenic underwater sound sources. The acoustic thermometry of the ocean climate system (ATOC) elicited subtle
changes in the diving behaviour of juvenile northern elephant seals (Costa et al. 2003). The ATOC sound source was a low frequency, high intensity (195 dB, 75 and 37.5 Hz) signal introduced into the deep sound channel off the coasts of California and Hawaii, USA. There was a significant positive correlation between ATOC sound pressure levels and descent rate (Costa et al. 2003). A change in descent rate has also been detected in grey seals exposed to pile driving events in the North Sea (Aarts et al. 2017). A statistically significant decrease in descent rate was reported in 39 of 58 exposures to pile driving within 36 km of the pile driving location (Aarts et al. 2017). However, there is very little information about the at-sea behavioural response of seals to non-impulsive noise sources such as shipping.

In the UK there is significant overlap between seals and shipping traffic within 50 km of the coast (Jones et al. 2017). Furthermore, seals commonly enter the water and display alert behaviour when disturbed by boats and cruise ships approaching haul-out sites to between 100 and 830 m (Andersen et al. 2012; Tripovich et al. 2012; Jansen et al. 2015). Evidence suggests that while the exposure of seals to shipping noise is often below the levels required to induce permanent or temporary threshold shift, they do experience high levels of exposure that could potentially induce a behavioural response (See Chp. 3) (Jones et al. 2017; Mikkelsen et al. 2019). Acoustic telemetry tags recording the acoustic exposure and diving behaviour of 5 seals in the North Sea recorded behavioural changes that coincided with vessel encounters (Mikkelsen et al. 2019). They provide an example of one seal that was ascending from a resting dive, when it descended briefly before returning to the surface due to noise (Mikkelsen et al. 2019). Chen et al. (2017) observed shallower diving as a ship approached its closest point to a grey seal pup and a resumption of bottom diving when the ship moved away. These case studies suggest that dive behaviour may be influenced by shipping noise. However, there is no quantitative analysis of at-sea diving behaviour of pinnipeds to shipping noise. To address this gap, this chapter aims to investigate changes in the diving
behaviour of adult grey seals and pups with exposure to shipping noise in the English Channel and Celtic Sea.

4.2 Methodology

To examine the impact of shipping noise on the diving behaviour of grey seals, this study utilised dive metrics derived from GPS/GSM Fastloc® tags on grey seals in the English Channel and Celtic Sea (Sec. 3.2.2). Shipping noise was predicted at each point along the seal’s track using a combination of AIS ship location data (Sec. 3.2.3), the RANDI ship source model (Sec. 3.2.4) and the range dependent acoustic propagation model RAMSurf (Sec. 3.2.5). The response of dive metrics to shipping noise was modelled using Generalised Additive Mixed Models (GAMM).

4.2.1 Noise exposure of seals

Dive and location data from Fastloc® GPS/GSM tags on 8 adult seals from the English Channel and 8 seal pups from the Celtic Sea were analysed (Tbl. 4.1). Chapter 3 calculated the exposure of these seals to shipping noise along 24-hr segments of their reconstructed three-dimensional track. The three-dimensional shipping noise soundscape was constructed every 15 minutes. Noise was extracted from each soundscape reconstruction along the seal track at a resolution of 1 second. Ship locations were determined using data from the Automatic Identification System of ships. The source level of each ship was calculated using the RANDI ship source model (Breeding et al. 1996), and the loss in acoustic intensity as noise propagated between the ship and the seal was modelled using the range dependent acoustic propagation model RAMSurf (Collins 1993). Chapter 3 provides a detailed description of each of these components of the methodology which were used to predict the noise exposure of seals. This analysis utilised these predictions to investigate changes in the diving behaviour of grey seals with shipping noise.

The reconstruction of the seal track using GPS locations and dive profiles are
Table 4.1: Details of seal tag data used in the chapter. A total of 16 seals were included; 8 adults and 8 pups. The table reports the number of dives and the number of high noise events included for each seal. ISMP - Iroise Sea Marine Park.

<table>
<thead>
<tr>
<th>ID</th>
<th>Location tagged</th>
<th>Mass (kg)</th>
<th>Sex</th>
<th>No. of dives</th>
<th>No. of events</th>
<th>Age Class</th>
</tr>
</thead>
<tbody>
<tr>
<td>B23</td>
<td>ISMP</td>
<td>129</td>
<td>M</td>
<td>67</td>
<td>5</td>
<td>Adult</td>
</tr>
<tr>
<td>B24</td>
<td>ISMP</td>
<td>124</td>
<td>M</td>
<td>97</td>
<td>4</td>
<td>Adult</td>
</tr>
<tr>
<td>B27</td>
<td>ISMP</td>
<td>152</td>
<td>M</td>
<td>119</td>
<td>9</td>
<td>Adult</td>
</tr>
<tr>
<td>B31</td>
<td>ISMP</td>
<td>206</td>
<td>M</td>
<td>61</td>
<td>4</td>
<td>Adult</td>
</tr>
<tr>
<td>B32</td>
<td>ISMP</td>
<td>114</td>
<td>F</td>
<td>130</td>
<td>5</td>
<td>Adult</td>
</tr>
<tr>
<td>B33</td>
<td>ISMP</td>
<td>210</td>
<td>M</td>
<td>270</td>
<td>13</td>
<td>Adult</td>
</tr>
<tr>
<td>B35</td>
<td>ISMP</td>
<td>148</td>
<td>M</td>
<td>48</td>
<td>3</td>
<td>Adult</td>
</tr>
<tr>
<td>B37</td>
<td>ISMP</td>
<td>70</td>
<td>M</td>
<td>136</td>
<td>6</td>
<td>Adult</td>
</tr>
<tr>
<td>hg27-01-09</td>
<td>Anglesey</td>
<td>37</td>
<td>M</td>
<td>34</td>
<td>3</td>
<td>Pup</td>
</tr>
<tr>
<td>hg27-04-09</td>
<td>Anglesey</td>
<td>38</td>
<td>M</td>
<td>33</td>
<td>3</td>
<td>Pup</td>
</tr>
<tr>
<td>hg29-11-10</td>
<td>Anglesey</td>
<td>35</td>
<td>M</td>
<td>21</td>
<td>1</td>
<td>Pup</td>
</tr>
<tr>
<td>hg29-16-10</td>
<td>Anglesey</td>
<td>40</td>
<td>F</td>
<td>181</td>
<td>7</td>
<td>Pup</td>
</tr>
<tr>
<td>hg29-18-10</td>
<td>Ramsey</td>
<td>32</td>
<td>M</td>
<td>81</td>
<td>3</td>
<td>Pup</td>
</tr>
<tr>
<td>hg29-21-10</td>
<td>Ramsey</td>
<td>37</td>
<td>M</td>
<td>356</td>
<td>9</td>
<td>Pup</td>
</tr>
<tr>
<td>hg29-23-10</td>
<td>Ramsey</td>
<td>29</td>
<td>M</td>
<td>24</td>
<td>4</td>
<td>Pup</td>
</tr>
<tr>
<td>hg29-24-10</td>
<td>Ramsey</td>
<td>32</td>
<td>F</td>
<td>275</td>
<td>11</td>
<td>Pup</td>
</tr>
</tbody>
</table>

detailed in Chapter 3 Section 3.2.2. For each seal, a dive began when the tag registered a depth of 1.5 m or more for 8 seconds and a dive was terminated when the tag registered a depth shallower than 1.5 m. Each dive was recorded as nine dive inflection points without georeferencing, but were reconstructed as in Section 3.2.2. A dive was associated with shipping noise by calculating the maximum received sound pressure level ($SPL_{\text{max}}$) (10-1000 Hz) experienced by the seal during that dive. $SPL_{\text{max}}$ was chosen as the most appropriate metric because it is not inherently related to the response variables dive duration and maximum depth. Total cumulative noise exposure would increase with the dive duration, and dive duration increases with maximum depth. The predictions of ship noise exposure suggest that seals are exposed to ambient level noise for the majority of their time in the 24-hr periods examined. Median noise levels would be more representative of this ambient noise exposure but seals could be habituated to this level of noise (Gotz and Janik 2010). However, intense high
noise exposure may be more likely to elicit a behavioural response (Gotz and Janik 2010; Götz and Janik 2011). Hence, maximum sound pressure level was considered a more appropriate noise metric to characterise noise levels during each dive. Sound pressure level alone is not always related directly to the severity of behavioural responses (Ellison et al. 2012). The inclusion of contextual variables such as signal-to-noise ratio, weather, behavioural state and sex can provide additional information (Ellison et al. 2012). However, the only available variables in this study were age and hearing threshold. The predicted noise exposure levels were weighted using the m-weighting function for pinnipeds (Southall et al. 2007), and the results in Chapter 3 support the assertion that noise levels from shipping were within the hearing range of the seals in the study.

4.2.2 Dive extraction and dive metrics

Chapter 3 calculated noise levels for 11,008 dives in the English Channel and 14,323 dives in the Celtic Sea. The $SPL_{\text{max}}$ was calculated for each dive as described above. Dives were described using the variables dive duration, maximum depth, ascent rate, descent rate, bottom time and inter-dive interval. Dive duration and maximum depth were transmitted directly by the telemetry tag. Ascent rate, descent rate, bottom time and inter-dive interval were derived from the transmitted values of dive duration, maximum depth and the nine depth inflection points describing the dive profile. Bottom time was defined as the length of time in seconds that was greater than or equal to 80% of the total dive depth (Lesage et al. 1999; Aarts et al. 2017). Descent rate was calculated as the rate of travel between the start of the dive and the beginning of bottom time at 80% of the maximum dive depth (Lesage et al. 1999; Costa et al. 2003). Ascent rate was calculated as the rate of travel between the end of bottom time and the end of the dive (Lesage et al. 1999; Costa et al. 2003). Dives less than 2 m in depth were removed because 80% of the total dive depth would be above the diving threshold of 1.5 m. Inter-dive interval was the length of time in seconds
between the start time of a dive and the end time of the preceding dive.

Preliminary exploration of the data suggested that $SPL_{\text{max}}$ had no clear relationship with each of the dive metrics. The diving behaviour of grey seals is complex and is driven by many other factors such as sex (Beck et al. 2003), developmental life stage (Carter et al. 2017), environmental properties (bathymetry, sediment type and time of day) (Photopoulou et al. 2014; Jessopp et al. 2013) and behavioural state (Thompson et al. 1991). It is also possible to assume that the effect size of noise may be small and the error associated with the location of the seal and noise predictions may be relatively large. As a result, it was preferable to focus on dives associated with high noise levels. As shown in Chapter 3, high noise events depend on the source level of the ships, number of ships and closest point of approach a ship. Therefore, high noise events arose under a number of scenarios: (i) a ship was close to the seal, (ii) a ship was not as close to the seal but had a louder source level (as a result of ship size or speed), or (iii) several ships further away from the seal cumulatively increased noise levels. Dives were extracted and classified into a noise category; before, during and after high noise events. High noise events were determined by calculating the range in $SPL_{\text{max}}$ for all seals in the English Channel and Celtic Sea separately. High noise events were bouts of one or more dives with noise equal to or above the 95th percentile. The before and after periods were defined as bouts of dives equal in number of dives to the high noise event. For example, for a high noise event with 3 dives, the before and after period also had 3 dives each. Each event was visually examined to ensure that the before, during and after categories did not overlap with that of any other. Any events which overlapped with another event (i.e. dives in an event were classified as two different categories in two events) both events were removed from the analysis. Extreme outliers in calculated dive metrics generated due to erroneous transmitted values were also removed from the analysis.

Table 4.1 shows the number of dives and events included for each seal. In the English Channel a total of 928 dives across 49 high noise events were included.
and 1005 dives in 41 events were included for the Celtic Sea (Fig. 4.1). The median length of high noise events (during category) was 32 (3.7 - 150) minutes with 5 (1-29) dives for the English Channel adults, and 18 (1.6 - 177.1) minutes with 5 (1-57) dives for Celtic Sea pups. The location of each event was diverse but each category within an event (before, during and after) occurred in a similar habitat. This reduces the influence of environmental variables in driving changes in diving behaviour within an event. However, individual responses may vary between seal but also within seal for each event.

Figure 4.1: Map of noise events showing location of each dive in each noise category and the bathymetric depth. Map a shows events for the English Channel adults and Map b shows events for the Celtic Sea pups.

4.2.3 Statistical analysis

The influence of noise on the diving behaviour of seals was analysed using Generalised Additive Mixed Models (GAMM). These models allow for a non-linear relationship between the response and explanatory variables (Lin and Zhang 1999; Wood 2006). As a result of previous studies, and the logarithmic nature of decibel noise levels, it was suspected that the dive metrics and each of the explanatory variables may not be linearly related (Photopoulou et al. 2014).
Therefore, an additive model was considered appropriate. As described in Chapter 3, the mean of the response is dependant on the explanatory variables through the sum of a number of smoothing functions and fixed effects, and random effects are used to model correlation between observations (Lin and Zhang 1999; Wood 2006). GAMMs were implemented in R version 3.5.3 (R Core Team 2019) using the mgcv package version 1.8-28 (Wood 2003; Wood 2004). The models were implemented using a Gaussian error distribution with an identity link function. Due to the strictly positive nature of the response variables a Gamma error distribution with an inverse link function was also considered. However, the Gaussian structure was appropriate because it produced superior AIC values, and did not predict negative values over the range of the response variables (Zuur 2009). Each model followed the general structure shown in Equations 4.1 to 4.4 where \( D \) is the diving parameter of interest, \( \beta \) are fixed parameter coefficients of the explanatory variables (Noise, Dive and Bathy), and \( a_i \) and \( b_{ij} \) are nested random intercepts for seal and event respectively. The distribution, mean and variance of the dive variable are given in Equation 4.1 and 4.2. The link function and predictor functions are given in Equations 4.3 and 4.4 respectively. Additional variance and correlation structures were also included where appropriate.

\[
D_{ijk} \sim N(\mu_{ijk}, \sigma^2) \quad i = \text{seal}, \ j = \text{event}, \ k = \text{dive} \\
E(D_{ijk}) = \mu_{ijk} \text{ and } \var(D_{ijk}) = \sigma^2 \\
\mu_{ijk} = \eta_{ijk} \\
\eta_{ijk} = \beta_1 \text{Noise}_{ijk} + \beta_2 \text{Dive}_{ijk} + f(\text{Bathy}_{ijk}) + a_i + b_{ij}
\]

Models were fit using maximum likelihood, which estimates the model parameters by maximising the likelihood function so that the measured dive data is the most probable, or using restricted maximum likelihood (REML), which corrects for biased variance estimation during estimation of fixed effects (Zuur
Model selection followed Zuur (2009) by first generating a model including all fixed components and interactions using REML. This was used to find the random structure that produced the lowest value of Akaike’s Information Criterion (AIC). The fixed effects structure was then determined using maximum likelihood and AIC values to remove variables that did not provide significant explanatory power to the model. AIC was given by \(-2\log\text{likelihood} + 2k\). The criterion assesses goodness of fit but penalises models for increasing complexity by including a penalty based on the number of parameters (k) (Zuur 2009). The lowest AIC values are considered superior (Zuur 2009). Each model was tested for possible violations of the basic assumptions by visual examination of the normalised residuals (Zuur et al. 2014).

The response variables dive duration, maximum depth, ascent rate, descent rate, bottom time and inter-dive interval were each modelled separately. The English Channel adults and Celtic Sea pups were analysed separately to avoid unnecessary complexity due to the confounding factors of location and age between the two groups. Furthermore, the diving behaviour of pups changes as their age increases (Carter et al. 2017). As a result, different variance structures could be expected between the two groups. Each response variable was modelled as a function of the explanatory variables noise category, bathymetry, and dive class (described below). Bathymetry is included because it constrains maximum dive depths and is known to partially explain diving behaviour (Photopoulou et al. 2014), and dive classes have been linked to habitat mediated dive behaviour in grey seals (Jessopp et al. 2013). Sex was not included as an explanatory variable due to the limited number of samples from female individuals in the study.

Noise category (before, during and after) and dive class (benthic, pelagic and shallow) were included as categorical variables with three levels. The interaction between noise category and dive class was also included. The assignment of each dive to a noise category was described in Section 4.2.2. The percentage of the total bathymetric depth for each dive was calculated following
\[ \text{bat}_{\text{percent}} = \left( \frac{\text{maximum depth}}{\text{bathymetric depth}} \right) \times 100 \] and was used to assign dives to a dive class. Benthic dives utilised \( \geq 80\% \) of water column, shallow dives used \( \leq 20\% \) of the water column and pelagic dives used between 20 and 80 \% of the water column (Jessopp et al. 2013). The inclusion of this variable improved the explanatory power of the models. Due to the relationship of this variable with bathymetry, variance inflation factors (VIFs) were calculated for the explanatory variables to test for collinearity (Zuur 2009). A VIF is an index that quantifies how much the variance of a variable is increased due to collinearity between it and other variables (Zuur 2009). VIFs less than three are within the acceptable range (Zuur 2009; Huon et al. 2015). To enable post-hoc comparison between each of the noise categories and dive classes, estimated marginal mean values of the model factors were calculated using the package emmeans and compared with adjusted p-values using Tukey’s correction for multiple comparisons (Lenth 2018). These values allow the examination of the mean response for individual factors given the covariates of the model (Lenth 2018). Bathymetric depth was extracted for the mid point in time of each dive using the EMODnet Digital Bathymetry at 1/8*1/8 arc minute resolution (EMODnet Bathymetry Consortium 2016) for UK waters. This was converted from lowest astronomical tide to mean sea level using the Vertical Offshore Reference Frame surface. The surface was developed by the UK Hydrographic Office and provides the relationship between different chart datums in UK waters (Sec. 3.2.5.2) (Adams et al. 2006). The bathymetry of French waters was extracted from the MNT Bathymétrique de Façade Atlantique (Projet Homonim) in metres with reference to mean sea level (Shom 2015). Bathymetry was included in the GAMMs as a smoothed term using a thin-plate regression spline (Wood 2003).

Individual seal reference number was included in the GAMMs as a random effect with event number as a nested random effect within seal. The random effect accounts for variation in the individual diving behaviour of seals and the variation in the response of each seal between events that occur in different habitats. The
nature of telemetry data (sequential measurements close together in time and space) indicate strong autocorrelation between dives. Autocorrelation in each model was examined using the autocorrelation-function in R (R Core Team 2019) on the normalised residuals of each model. The inclusion of random effects induces a correlation structure within each seal/event (Zuur 2009). However, if any remaining autocorrelation was present in the model an additional spherical correlation structure \( \text{corSpher}(form = \sim 1 | \text{seal}) \) was included in the model. The spherical structure is appropriate for dealing with irregularly spaced telemetry data (Zuur 2009). If the spread of residuals showed heterogeneity, an additional variance structure \( \text{varIdent}(form = \sim 1 | \text{dive}_c) \) was included to address this issue. The response variable was \( \log(y + 1) \) transformed to improve heterogeneity where other methods (e.g. alternate error distributions and different variance structures) did not improve the model residuals mainly as a result of convergence failure.

4.3 Results

4.3.1 Noise exposure levels

The median maximum received sound pressure level \( SPL_{\text{max}} \) for dives in the English Channel during high noise exposure was 122 dB re 1\( \mu \)Pa (Tbl. 4.2). The difference between this high noise period and the median \( SPL_{\text{max}} \) of dives in the before and after category was 7 dB. In the Celtic Sea, median \( SPL_{\text{max}} \) for the during noise category was lower than the English Channel at 111 dB re 1\( \mu \)Pa (Tbl. 4.2). However, the difference between median \( SPL_{\text{max}} \) during high noise exposure and before and after was much greater at 13 dB and 15 dB respectively.

4.3.2 Noise categories and dive class

There were very few shallow dives included in the study for the English Channel in comparison to benthic and pelagic dives. For the Celtic Sea pups the majority of dives included in the study were pelagic. In the English Channel the greatest
Table 4.2: Median maximum sound pressure level (SPL) of dives in each noise category for English Channel adults and Celtic Sea pups.

<table>
<thead>
<tr>
<th>Noise Category</th>
<th>English Channel</th>
<th>Celtic Sea</th>
</tr>
</thead>
<tbody>
<tr>
<td>Before</td>
<td>115</td>
<td>98</td>
</tr>
<tr>
<td>During</td>
<td>122</td>
<td>111</td>
</tr>
<tr>
<td>After</td>
<td>115</td>
<td>96</td>
</tr>
</tbody>
</table>

number of benthic dives occurred before high noise exposure (Fig. 4.2a). As noise levels increased during noise exposure the number of benthic dives decreased. This trend continued after noise exposure with a further decrease in the number of benthic dives. The greatest number of pelagic dives occurred during noise exposure with a lower number of pelagic dives after and before noise exposure. Shallow dives occurred least often during noise exposure with a very similar number of shallow dives occurring before and after noise exposure. The same patterns were also seen in the Celtic Sea for the pelagic and shallow dives (Fig. 4.2b). However, in contrast the greatest number of benthic dives occurred during the high noise exposure with a lower number of benthic dives occurring before and after high noise exposure.

4.3.3 English channel adults

Following stepwise model selection using AIC, the final selected models for dive duration, maximum dive depth, descent rate, bottom time and inter-dive interval did not include noise category or the interaction with dive class as a significant explanatory variable (Supp. Mat. B.1). In contrast, noise category and the interaction of noise category with dive class were significant explanatory factors in the minimum adequate model for the response variable ascent rate (Tbl. 4.3). This model also included dive class as a main variable but the bathymetric smooth was not significant and, therefore, was removed.

The mean ascent rate for the English Channel adults was 0.71 (0.05 - 2.56) ms$^{-1}$. For benthic dives, there was a significant increase between ascent rate after exposure to high levels of shipping noise compared to before (Fig. 4.3; Tbl.
However, the difference in estimated mean ascent rate was only 0.09 $ms^{-1}$. There was no significant difference between noise categories for pelagic dives (Fig. 4.3; Tbl. 4.4). For shallow dives, there was a significant increase (0.31 $ms^{-1}$) in ascent rate during noise exposure compared to before noise exposure, and decrease (0.30 $ms^{-1}$) in ascent rate after compared to during exposure to high levels of shipping noise (Fig. 4.3; Tbl. 4.4). The random effect coefficients for each individual seal and event ranged from $\pm 0.128$ $ms^{-1}$ revealing variation in ascent rate across seals and events (Supp. Mat. B.2). Variation in response was also present between individual seals responses to different events. Seal B27 showed both the maximum increase and decrease in the random intercept coefficients (Supp. Mat. B.2).

### 4.3.4 Celtic sea pups

The minimum adequate models for dive duration, maximum depth, ascent rate, bottom time and inter-dive interval did not include noise category or an interaction between noise category and dive class as significant explanatory
Table 4.3: The structure of the maximal model and the final minimum adequate model for the response variable ascent rate ($A_r$) in the English Channel.

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>$R^2$ (Adj)</th>
<th>AIC</th>
<th>△ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Full</td>
<td>15</td>
<td>0.244</td>
<td>-69</td>
<td></td>
</tr>
<tr>
<td>Final</td>
<td>13</td>
<td>0.245</td>
<td>-73</td>
<td>-4</td>
</tr>
<tr>
<td>Final - Noise</td>
<td>7</td>
<td>0.245</td>
<td>-62</td>
<td>11</td>
</tr>
</tbody>
</table>

1 $A_r \sim s(bathy) + noise_cat + dive_c + noise_cat : dive_c + (1|seal/event) + corSpher(1|seal)$
2 $A_r \sim noise_cat + dive_c + noise_cat : dive_c + (1|seal/event) + corSpher(1|seal)$
3 $A_r \sim dive_c + (1|seal/event) + corSpher(1|seal)$

Figure 4.3: The estimated marginal means (point) and confidence intervals (blue box) for GAMM of ascent rate for diving seals in the English Channel exposed to shipping noise.

Table 4.4: Contrasts of the estimated marginal means from model of English Channel adults ascent rate by noise category and dive class. B - Before; D - During and A - After.

<table>
<thead>
<tr>
<th>Dive Class</th>
<th>Contrast</th>
<th>Difference in Means</th>
<th>SE</th>
<th>t ratio</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Benthic</td>
<td>B - D</td>
<td>-0.040</td>
<td>0.029</td>
<td>-1.405</td>
<td>0.339</td>
</tr>
<tr>
<td></td>
<td>B - A</td>
<td>-0.087</td>
<td>0.030</td>
<td>-2.952</td>
<td>0.009 *</td>
</tr>
<tr>
<td></td>
<td>D - A</td>
<td>-0.047</td>
<td>0.029</td>
<td>-1.607</td>
<td>0.243</td>
</tr>
<tr>
<td>Pelagic</td>
<td>B - D</td>
<td>-0.063</td>
<td>0.029</td>
<td>-2.179</td>
<td>0.075</td>
</tr>
<tr>
<td></td>
<td>B - A</td>
<td>-0.001</td>
<td>0.030</td>
<td>-0.046</td>
<td>0.999</td>
</tr>
<tr>
<td></td>
<td>D - A</td>
<td>0.062</td>
<td>0.028</td>
<td>2.233</td>
<td>0.066</td>
</tr>
<tr>
<td>Shallow</td>
<td>B - D</td>
<td>-0.310</td>
<td>0.118</td>
<td>-2.634</td>
<td>0.023 *</td>
</tr>
<tr>
<td></td>
<td>B - A</td>
<td>-0.008</td>
<td>0.084</td>
<td>-0.093</td>
<td>0.995</td>
</tr>
<tr>
<td></td>
<td>D - A</td>
<td>0.302</td>
<td>0.117</td>
<td>2.573</td>
<td>0.028 *</td>
</tr>
</tbody>
</table>
variables (Supp. Mat. B.3). The interaction between noise category and dive class was a significant explanatory variable for the descent rate of dives undertaken by the pups (Tbl. 4.5). The mean descent rate of pups was 0.69 (0.05 - 2.08) $ms^{-1}$ but the spread of data over dive class and noise category was large.

Table 4.5: The structure of the maximal model and the final minimum adequate model for the response variable descent rate ($D_r$) in the Celtic Sea.

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>$R^2$ (Adj)</th>
<th>AIC</th>
<th>$\Delta$ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Full 1</td>
<td>17</td>
<td>0.438</td>
<td>-936</td>
<td></td>
</tr>
<tr>
<td>Final 2</td>
<td>15</td>
<td>0.452</td>
<td>-939</td>
<td>-3</td>
</tr>
<tr>
<td>Final - Noise 3</td>
<td>9</td>
<td>0.449</td>
<td>-932</td>
<td>7</td>
</tr>
</tbody>
</table>

$1 \ log(D_r+1) \sim s(bathy)+noise_{cat}+dive_{c}+noise_{cat}\cdot dive_{c}+(1|seal/event)+corSpher(1|seal)+varIdent(1|dive_{c})$

$2 \ log(D_r+1) \sim noise_{cat}+dive_{c}+noise_{cat}\cdot dive_{c}+(1|seal/event)+corSpher(1|seal)+varIdent(1|dive_{c})$

$3 \ log(D_r+1) \sim dive_{c}+(1|seal/event)+corSpher(1|seal)+varIdent(1|dive_{c})$

For benthic and shallow dives there were no significant differences in the descent rate of dives. However, for pelagic dives there was a significant decrease in descent rate after exposure to high levels of shipping noise when compared to descent rate before exposure (Fig. 4.4; Tbl. 4.6). Furthermore, there was a significant decrease in the ascent rate of dives undertaken by seals after exposure compared to during exposure to high levels of shipping noise (Tbl. 4.6; Fig. 4.4). The random effect coefficients for the individual seals ranged from -0.11 to 0.23 $ms^{-1}$. This is slightly wider than the ascent rate of English Channel adults. There is more variation in the random effect coefficients for each seal than the ascent rate for adults suggesting individuals displayed more variation in descent rate across events (Supp. Mat. B.2)

Model validation plots for both the model of English Channel ascent rate and the model of Celtic Sea descent rate are shown in Supplementary Material B.4. They show that each model met the basic assumptions of the GAMM methodology. All VIF values were less than 2 demonstrating collinearity between variables was within acceptable levels.
Figure 4.4: The estimated marginal means (points) and confidence intervals (blue boxes) for GAMM of descent rate for diving seals in the Celtic Sea exposed to shipping noise. Values are given on the transformed log(y+1) scale.

Table 4.6: Contrasts of the estimated marginal means from model of Celtic Sea pups descent rate by noise category and dive class. B - Before; D - During and A - After. Values are given in the transformed scale log(y+1).

<table>
<thead>
<tr>
<th>Dive Class</th>
<th>Contrast</th>
<th>Difference in Means</th>
<th>SE</th>
<th>t ratio</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Benthic</td>
<td>B - D</td>
<td>0.013</td>
<td>0.023</td>
<td>0.536</td>
<td>0.854</td>
</tr>
<tr>
<td></td>
<td>B - A</td>
<td>0.031</td>
<td>0.025</td>
<td>1.223</td>
<td>0.440</td>
</tr>
<tr>
<td></td>
<td>D - A</td>
<td>0.043</td>
<td>0.024</td>
<td>1.831</td>
<td>0.160</td>
</tr>
<tr>
<td>Pelagic</td>
<td>B - D</td>
<td>0.002</td>
<td>0.015</td>
<td>0.148</td>
<td>0.988</td>
</tr>
<tr>
<td></td>
<td>B - A</td>
<td>-0.047</td>
<td>0.015</td>
<td>-3.150</td>
<td>0.005 *</td>
</tr>
<tr>
<td></td>
<td>D - A</td>
<td>-0.045</td>
<td>0.015</td>
<td>-3.095</td>
<td>0.006 *</td>
</tr>
<tr>
<td>Shallow</td>
<td>B - D</td>
<td>-0.067</td>
<td>0.050</td>
<td>-1.353</td>
<td>0.366</td>
</tr>
<tr>
<td></td>
<td>B - A</td>
<td>-0.016</td>
<td>0.046</td>
<td>-0.360</td>
<td>0.931</td>
</tr>
<tr>
<td></td>
<td>D - A</td>
<td>-0.083</td>
<td>0.050</td>
<td>-1.653</td>
<td>0.224</td>
</tr>
</tbody>
</table>
4.4 Discussion

The effect of high ship noise exposure events on the diving behaviour of grey seals was examined using predicted ship noise exposure values and dive metrics from Fastloc® GPS/GSM telemetry tags. Median maximum SPL levels during high noise exposure events were 111 and 122 dB re 1 $\mu$Pa in the Celtic Sea and English Channel respectively. These values were between 7 and 15 dB louder than noise exposure before and after these events. Noise category and their interaction with dive class were a significant explanatory factor in the ascent rate of dives undertaken by the English Channel adults. Ascent rate was significantly faster after high noise exposure when compared to before noise exposure when seals were undertaking benthic dives, but was significantly higher during exposure for shallow dives. In the Celtic Sea, the interaction with dive class was significant for the descent rate of certain pup dive classes. Descent rate for pelagic dives after high noise exposure was significantly lower than before or during high noise exposure.

A number of other studies have reported changes in the ascent rate and descent rate of diving seals in response to noise. Observations of juvenile northern elephant seals (*Mirounga angustirostris*) exposed to the acoustic thermometry of climate (ATOC) sound source suggested descent rate was the most sensitive of the diving parameters studied to the noise source (Costa et al. 2003). There was a significant positive correlation between sound pressure level and descent rate at exposure levels ranging from 118 to 137 dB re 1$\mu$Pa (60-90 Hz) (Costa et al. 2003). This was also reflected in an examination of the diving behaviour of grey seals with respect to pile driving. A decrease in the descent speed of dives was the most commonly observed reaction to pile driving noise (~137 dB re 1 $\mu$Pa$^2$s single strike SEL) (Aarts et al. 2017). In relation to shipping noise, Mikkelsen et al. (2019) describes anecdotal examples of seals interrupting the ascent of dives and suddenly descending to the sea floor when encountering high levels of shipping noise (113 dB re 1 $\mu$Pa RMS 0.1-50 kHz). Controlled exposure to the
sounds of predators and sonar also resulted in a reversal of the ascent phase of
dives for northern elephant seals (Fregosi et al. 2016). The results presented
here support current evidence that exposure to noise can result in changes in
the descent rate and ascent rate diving seals, and extends this to include
shipping noise exposure. However, the direction and magnitude of change
varies between studies.

In the Celtic Sea, pups decreased the descent rate of pelagic dives after noise
exposure. This response was similar to the observed responses of grey seals to
pile driving in the North Sea, where the descent speed of dives decreased.
However, this response occurred during noise exposures. A decrease in descent
rate could occur because the seals swim the same length of path through the
water at a slower speed or they undertake a longer path with more horizontal
movement. This could suggest that seals are swimming away from noise
sources. In noise response studies featuring grey seals or harbour seals in
similar habitats to those studied here, movement away from the noise source
has been recorded several times (Russell et al. 2016; Aarts et al. 2017; Hastie
et al. 2018). However, Costa et al. (2003) reported a highly significant positive
correlation between sound pressure level and descent rate in juvenile northern
elephant seals, and interpreted this as a startle response similar to anti-predator
behaviour where seals dive rapidly to the seafloor. It is possible northern
elephant seals and grey seals would have different responses to a threat
because they have different life histories, live in different habitats, and could
have different experiences and motivations (Gotz and Janik 2010). Furthermore,
the noise characteristics of each noise source are different.

In the English Channel, adult seals increased the ascent rate of shallow dives
during noise exposure, and for benthic dives increased ascent rate after noise
exposure compared to before noise exposure. An increase in ascent rate could
occur as a result of seals surfacing more quickly or by reducing the amount of
horizontal movement in a dive. Juvenile northern elephant seals showed
variable responses in ascent rate to the ATOC noise exposure with ascent rate
increasing in one animal and decreasing in another (Costa et al. 2003). Dive
inversions, where ascent is interrupted by a descent have been reported in
response to predator, sonar and shipping noise (Fregosi et al. 2016; Mikkelsen
et al. 2019). This would have been consistent with a decrease in ascent rate,
which was not reported here. However, Fregosi et al. (2016) utilised a tag which
emitted the noise exposures itself and therefore, noise was emitted at the
location of the seal; an unrealistic scenario that would possibly elicit more
extreme responses. Furthermore, Mikkelsen et al. (2019) reported this
behaviour on one occasion for one seal and given the variability in individual dive
responses it is likely that some seals may respond this way and others may not
depending on the context of the exposure.

The driver of the different responses seen in the results for different dive classes
is unclear. There was a significant response in the descent rate of Celtic Sea
pups undertaking pelagic dives. This was also the most common type of dive for
pups included in this study. The results may therefore, have been influenced by
the greater number of this dive type included in the study from which to detect a
response. This could be driven in part by the sampling of dives in Chapter 3
which excluded shallow areas, but the dive behaviour of pups also changes
through time as they develop effective movement and foraging strategies, and
undergo ontogenetic changes in factors such as buoyancy and oxygen storage
capacity (Carter et al. 2017). In these seals, the proportion of benthic dives and
the bottom time of dives was shown to increase with time since departure from
the colony (Carter et al. 2017). As diving behaviour is under development and
benthic diving is less established in these animals, it may have made it difficult to
detect a response in benthic dives, and the behavioural repertoire available to
the animals could be more limited. It also could account for differences seen in
the number of dives for each dive class in the two regions (Fig. 4.2). For English
Channel adults, there was a similar number of benthic and pelagic dives
included in the study but only a small number of shallow dives. A change in
ascent rate was seen while seals were undertaking shallow dives during noise
exposure and benthic dives after noise exposure. When visually examining the track data, the English Channel seals appear to be undertaking directed travel. A higher ascent speed could be indicative of travelling faster and undertaking shallower dives especially given the number of benthic dives also decreases after noise exposure. However, the classification of dives includes some uncertainty due to mismatches between the tidal cycle during which seals are diving and the datum against which bathymetric data is reported, and any error in the GPS locations. Section 3.2.2 and 3.2.5.2 describe how these factors were limited as much as possible.

It is common in behavioural response studies of pinnipeds to noise that there is a high level of variation in individual responses (Costa et al. 2003; Gotz and Janik 2010; Kastelein et al. 2006; Fregosi et al. 2016; Aarts et al. 2017). This highlights the importance of context and the difficulty in concluding a general response strategy for seals to a particular noise type. Ascent rate both increased and decreased in northern elephant seals in response to ATOC (Costa et al. 2003), and a decrease in descent speed, no response and an increase in descent speed were seen in individual responses to pile driving activities (Aarts et al. 2017). The results show that there was variation in the ascent rate and descent rate not only for each individual seal but also for each high noise exposure event suggesting high levels of behavioural plasticity when responding to noise exposure events. Ascent rate changed in the English channel adults and descent rate changed in the Celtic Sea pups at lower exposure levels, and both groups showed variation in responses. However, it is not possible to say whether this was due to location, the naivety of the pups to shipping noise, the differential sound exposure levels or some combination of each factor. This variation makes it more difficult to detect population level effects. The most pertinent point is, therefore, that there is strong evidence for some response to noise exposure from pinnipeds across a number of studies looking at different noise sources particularly in relation to the ascent and descent rate of dives (Mikkelsen et al. 2019; Aarts et al. 2017; Russell et al. 2016; Hastie et al. 2018).
The results here further support this for both adult grey seals and pups in response to shipping noise in two different areas. Changes in diving behaviour, as demonstrated here, may impact vital rates through mechanisms such as missed foraging opportunities and changes in the metabolic cost of behavioural activities, regardless of whether they surface to investigate the exposure or dive to avoid the exposure. In the English Channel, the ascent rate of benthic dives increased after noise exposure, and the descent rate of Celtic Sea pups decreased after noise exposure for pelagic dives. The results suggest that changes in diving behaviour related to noise exposure are potentially not limited to the noise exposure event. This could be a latent behavioural change or a behavioural change that has lasted much longer than the exposure event. This may have consequences when considering whether such behavioural changes will have an impact on individual vital rates and population stability. Costa et al. (2003) found a significant decrease in ascent rate and a significant increase in descent rate in the hour after exposure when compared to the mean rates for a period lasting 18 hours after exposure. This suggests that changes in diving behaviour lasted longer than the exposure duration. The results here suggest behaviour only changed after noise exposure suggesting a latent behavioural response. The drivers of altered dive behaviour after rather than during noise exposure is not clear. Shipping is a continuous noise source that gradually increases and decreases as distance changes between the seals and ships. Therefore, the end of the before category and beginning of the after category are similar in noise levels to the start and end of the during category respectively. These factors may account for differences seen in the timing of behavioural responses between studies. However, there may also be ecological reasons why behavioural change occurs after a noise exposure. It is possible that a loud dive could sensitise seals to noise altering the dives that come after as a result of the experience (Götz and Janik 2011), or the analysis was not sensitive enough to detect fleeting responses just the recovery from the response which may be
more long term and hence detectable. The results point to the complexity of detangling behavioural responses when the normal behavioural ethogram of any species is so varied and uncertain.

The deviations in ascent rate and descent rate reported here appear to be small (< 0.3 $ms^{-1}$). The extent to which such small changes in individual diving behaviour could result in population level consequences is difficult to determine. Likewise the extent to which such exposure compromises individual animal welfare is unknown (Papastavrou et al. 2017). The population consequences of disturbance is an area of active research mainly constrained by a lack of data to parameterise models of disturbance (Nabe-Nielsen et al. 2018). The results presented here contribute information for inputs into models to assess these questions. Further research on the connections between behavioural change and life functions, and life functions and vital rates (Fig. 2.5) will be required to determine if shipping noise could have a meaningful impact on populations.

Disturbance that leads to reduced foraging can lead to reduced pup recruitment especially in pinnipeds that adopt an income breeding strategy (McHuron et al. 2017). However, the interim Population Consequences of Disturbance model estimated an increase in vessel numbers from 40 to 470 per year in the Moray Firth would have no impact on bottlenose dolphin populations because the behavioural changes did not result in a change in the health of individuals (New et al. 2013). It should be noted that disturbance from shipping noise is not the only stressor facing pinniped populations. When considering the population consequences this should be done in the context of exposure to other noise types (Hastie et al. 2014; Hastie et al. 2015; Kastelein et al. 2006; Götz and Janik 2013), by-catch (Cosgrove et al. 2016; Bjorge et al. 2002), culling (Thompson et al. 2007) and disease (Yon et al. 2019).

It is important to note that the behavioural response of seals to ships may not be driven by noise exposure alone. The physical presence of the ship may also be a factor in the impact of shipping traffic on seals. In air, sight plays a more central role in sensing the environment than when seals are underwater, and
approaching ships elicit alert and fleeing responses in hauled-out seals, although such studies also do not distinguish between noise and the presence of the ship (Jansen et al. 2015; Anderwald et al. 2013). Evidence suggests that dolphin foraging behaviour was reduced in the presence of boats but not in relation to noise level (Pirotta et al. 2015). As a result, future models should include the distance to ships in order to explore this possible interaction. The modelling approach utilised in this study, which determined ship locations every 15 minutes, made it difficult analyse the distance between the seal and ship at a sufficiently fine scale. Furthermore, predator behaviour could also be driven by prey responses to noise as opposed to seal responses to shipping noise. It is not known to what extent prey reactions could influence predators but this could be examined as more research becomes available regarding responses of marine animals other than mammals. For example, European eels (*Anguilla anguilla*) exposed to playbacks of shipping noise were 50% less likely to startle to an ambush predator and twice as likely to be caught by a pursuit predator (Simpson et al. 2015). Such changes in prey behaviour could drive responses in predator behaviour. Burgeoning technology such as acoustic telemetry tags that include acoustic recordings and accelerometry could help elucidate this further by allowing consideration of prey capture attempts using the accelerometer data when validated by concurrent camera deployments (Mikkelsen et al. 2019).

There are a number of methodological factors that should be considered when examining the results. As mentioned there are some uncertainties in the noise predictions as described in Chapter 3. These include ships missing from AIS data and incomplete AIS transects, ship source model error and error in defining the GPS locations of dives. The definition of before, during and after noise could also impact the results. Costa et al. (2003) found that behaviour was different up to 1 hour after noise exposure. However, in this study, time between high noise exposure events was too small to monitor changes before and after exposure for such long periods. Dives classified in overlapping categories were removed but due to the short length of time between events this may still be a factor in some
events. Further exploration of recovery times from behavioural disturbance might help define a suitable length for before, during and after noise level categories. Consideration of high noise exposure events that occur close together in time to look at cumulative impacts could also be useful. These uncertainties could contribute to difficulties detecting changes in diving behaviour given that the effect size may be small. The use of tags which record behaviour and acoustic exposure concurrently could help address these limitations in the future (Mikkelsen et al. 2019).

This chapter has demonstrated that high ship noise exposure events result in an increase in the ascent rate of adult grey seals and a decrease in the descent rate of grey seal pups for certain dive types. A significant change in pelagic diving behaviour was found after high noise exposure for Celtic Sea pups and after exposure for English Channel adults undertaking benthic dives. Diving behaviour changed during ship noise exposure for English Channel adults undertaking shallow dives. There was variation in the responses of individual seals and between high noise events. The results suggest that shipping noise does have an impact on the diving behaviour of grey seals, possibly consistent with a surfacing or swimming away response. The results provide useful data in the future assessment of the impacts of shipping noise. The small and transient nature of the changes suggests that they may not have a biologically meaningful effect on vital rates and population stability. However, the results will help inform future assessments of such effects.
An adaptive grid to improve the efficiency and accuracy of modelling underwater noise from shipping

5.1 Introduction

Key environmental protection legislation worldwide seeks to regulate noise from shipping (MMPA 1972; ESA 1973; European Commission 2008; European Commission 2017; Lucke et al. 2013), and as a result, industry and regulatory bodies are often required to robustly quantify the levels of underwater noise emissions associated with shipping for monitoring purposes, and in some circumstances, environmental impact assessment (Merchant et al. 2016). Underwater acoustic propagation models are an essential tool to predict noise for these regulatory and research activities (Dekeling et al. 2014; Farcas et al. 2016; Sertlek et al. 2016).

Specifically, acoustic propagation models are primarily used to create ship noise maps (Erbe et al. 2014; Marine Management Organisation 2015). These are important for managers because maps highlight patterns of noise in time and space. It is not practicable to measure noise over large areas using hydrophones. Therefore, to produce a map, it is necessary to predict noise, using a model, at the locations that cannot be measured directly in the environment. It is thought future trends in shipping noise could range in magnitude from 0.1 dB per year (Dekeling et al. 2014) to 3.3 dB per decade (Frisk 2012). It could take many years to detect trends of that size in measured point data. Acoustic propagation modelling can help to reduce the number of years and stations required for monitoring trends by allowing spatial averaging of noise levels (Dekeling et al. 2014). Furthermore, an understanding of noise variability in space can be used to suggest the optimum locations for underwater fixed monitoring equipment (Van der Graaf et al. 2012). Acoustic propagation models are also executed at smaller spatial scales, particularly between one or many sources and a single receiver, in order to validate acoustic propagation models against field measurements as well as benchmark the efficiency and accuracy of different acoustic propagation models (Etter 2013). Moreover, they can be useful to assess the individual exposure of animals for scientific and
regulatory procedures where animal locations are given exactly by telemetry devices or observations (Chen et al. 2017). They have been used successfully in that capacity throughout this thesis. However, the utilisation of acoustic propagation modelling to undertake such activities is known to have intensive time and computing requirements (Etter 2013; Wang et al. 2014; Marine Management Organisation 2015; Sertlek et al. 2016).

Acoustic propagation models tend to be computationally intensive to execute because they are based on a detailed physical representation of acoustic wave propagation and in many cases also account for detailed changes in the environment (range dependent models) (Etter 2013). Acoustic wave propagation is dependent on sound speed, which is determined by the temperature, hydrostatic pressure and salinity of a water mass (Etter 2013). Propagation is also influenced by absorption and reflection of waves at boundaries between the water and the surface, the water and the seafloor sediments and different water masses in the ocean (Etter 2013). However, when predicting shipping noise numerical range dependent models are often neglected in favour of simple geometric spreading laws (Etter 2013; Marine Management Organisation 2015). These spreading laws only assume acoustic energy decays logarithmically as sound propagates from source (Urick 1983). The main attraction of using geometric laws is the speed at which calculations can be conducted (Marine Management Organisation 2015; Farcas et al. 2016). However, it has been shown that geometric spreading laws can result in significant errors (Robinson et al. 2014; Farcas et al. 2016). Farcas et al. (2016) demonstrated that when compared to a more complex model (RAM (Collins 1993)), which allows environmental properties to vary with range from source, the geometric spreading laws underestimated noise close to the source and overestimated noise far from source. This is of particular concern when trying to make predictions for legislation relating to marine ecosystems as it could result in a failure to put in place appropriate mitigation strategies to protect sensitive species. Consequently, in using the geometric laws, users are often making a
compromise between computational efficiency and accuracy. As a result, there is a need for methodologies which can reduce the computational costs of executing advanced models so that users can leverage the greater level of realism they provide.

Currently, there are a number of strategies available to make acoustic propagation modelling more tractable. For example, it is most pertinent to select, from the numerous available models, an appropriate model for the specific requirements of a study (Farcas et al. 2016). The selection of a model will depend on the frequency characteristics of the noise source, the depth of the water, the variability of the environmental characteristics in the study area and the computational power available (Etter 2013). The incorrect choice of a model will compromise both the efficiency and accuracy of the results. Furthermore, an assumption of uniform sound speed, uniform sediment type and uniform bathymetry is often made to simplify propagation calculations (Sertlek et al. 2016). However, in environmentally variable regions, where there are changes in water mass properties, seafloor sediments and bathymetry, these assumptions are not valid. This is often the case in shallow shelf environments where the structure of the water column can be highly stratified (Simpson and Sharples 2012). In these environments, computationally intensive models that characterise environmental variation using a range and depth dependent approach are required (Jensen 2011).

For shipping specifically, where there are many disparate noise sources (ships), increases in efficiency can be achieved by spatially partitioning the study area into a grid. Typically, a grid will group ships in square grid cells of a fixed size (Erbe et al. 2014). Applying a grid to the ship data improves efficiency by reducing the number of times the acoustic propagation model must be executed. It is only necessary to calculate propagation loss once from the centre of a grid cell to the location of the noise receiver. This propagation loss value can then be applied to all ships in a grid cell (Erbe et al. 2012a; Erbe et al. 2014). The grid cell size selected for a study is concerned with achieving a realistic execution
time for the scale of the study area. Regional studies typically use grid cells between 2 and 5 km square (Erbe et al. 2014; Marine Management Organisation 2015), while global studies have used cells of 1° in longitude and latitude (Porter and Henderson 2013). The larger the grid cells the fewer calculations required, and therefore, the more efficient the solution. However, the larger the grid cell size, the less accurate the resulting model output (Erbe et al. 2012b). Larger grid cells do not account for environmental variation. This means that propagation loss values at different points within the cell may vary and the assumption that the propagation loss value at the centre of the cell can be applied to all ships in that cell is incorrect.

This study aims to develop a method which produces efficient and accurate noise level predictions using acoustic propagation models by designing an adaptive grid to spatially partition ship source data. The chapter presents a grid where cell size will vary with distance from the receiver. At ranges close to the receiver, where propagation loss changes very rapidly, a small grid size can be used. However, where ships are far away from the receiver, cell sizes can be much larger due to the logarithmic decay in acoustic energy with range. The study then investigates the efficiency and accuracy of this approach. Theoretically, it improves computational efficiency by reducing the number of calculations required but maintains, or improves, the accuracy of propagation loss estimations when compared to a grid with uniform cell size. Ultimately, this will improve the noise level predictions made using underwater acoustic propagation models for use in ship noise monitoring by making the implementation of more sophisticated models computationally tractable.

5.2 Methodology

This chapter presents an adaptive grid that will spatially group ships. Propagation loss can therefore, be calculated once from the centre of each grid cell to the receiver and applied to all ships in that grid cell. In order to avoid the introduction of error as a result of grouping the ships in this way, ideally propagation loss
should be uniform (not vary) across the cell (i.e. the value at the centre of the cell should be representative of the propagation loss at all the points in the cell). In this study, propagation loss was considered *uniform* when the propagation loss value from the centre of a grid cell to the receiver was approximately equal (given an error of ±1.5 dB) to the propagation loss value from each corner of the cell to the receiver. Depending on the distance between the source and the receiver, the maximum grid cell size where propagation loss is uniform will vary. This distance of uniform propagation loss was determined for a number of different grid sizes and used to predict the relationship between these two variables. This study used the relationship between grid size and distance of uniform propagation loss to produce an adaptive grid, and then demonstrated how the adaptive grid reduces computational effort and preserves the accuracy of finer more computationally expensive uniform grids.

### 5.2.1 Case study area

This study focussed on the Celtic Sea region shown by the map in Figure 5.1. It was considered preferable to use a case study, rather than an idealised site with uniform environmental properties, in order to demonstrate the efficiency and potential limitations of the new method in a real setting. The area is representative of temperate, shallow, coastal shelf waters. The Celtic Sea is seldom deeper than 120 m and is characterised by the rapid development of a strong thermocline in the summer (April to November) and its slow breakdown in autumn (Pingree 1980). The region is dynamically active and its water column properties are influenced by multiple mesoscale eddies and fronts (Pingree 1980). The adaptive grid should be transferable to areas with similar characteristics. Shallow, on-shelf seas are particularly interesting because they play a highly important role in the functioning of the global ocean including biological productivity, economic activity including shipping and the provision of social capital (Simpson and Sharples 2012).
5.2.2 Grid generation and analysis for propagation loss/distance simulations

In order to determine at what distance from the receiver the propagation loss becomes uniform across a grid cell, a series of propagation loss simulations were conducted at different grid cell sizes. The smallest grid cell size was 0.5 km and cell size was increased in 0.5 km increments up to a maximum of 20 km. This range was chosen because it is difficult for the acoustic propagation model to produce reliable results over distances shorter than 0.5 km, and a 20 km grid cell size was large enough not to result in uniform propagation loss under any of the conditions examined in this study. Figure 5.2 represents how the grid was structured for these simulations. A fixed receiver was located at one end of a transect shown on the map of the study area (Fig. 5.1). The grid boxes extended 200 km horizontally from this point but remained one grid box high vertically (Fig. 5.2). This was a computationally simple arrangement within the constraints of the study area. As shown in Figure 5.1, 200 km extends across the width of the study area, while trying to avoid the most extreme bathymetric changes. This will
increase the applicability of results to other shallow shelf seas. The first grid square was always 0.5 km from the receiver, and grid cells were overlapped horizontally by 0.5 km in order to increase the resolution of the resulting curves. If they were not overlapped in this manner the 20 km grid cells would only result in 10 data points compared to the 400 generated by a grid with 0.5 km cells. Propagation loss was calculated between sources located at each corner and at the centre of every grid cell to the receiver (Fig. 5.2).

![Diagram of grid and source/receiver placement](image)

**Figure 5.2:** Schematic of generated grid for calculating the distance between source and receiver at which propagation loss is uniform (i.e. approximately equal at all points in a grid cell). The receiver is shown as a square point located at one end of the transect. A hypothetical source was placed at the corner of each grid cell and at the centre of the cell. The grid extended the length of the 200 km transect. Each new grid cell was placed 0.5 km further along the transect than the last to increase the resolution of the resulting propagation loss curves. Propagation loss was calculated from every source to the receiver directly as shown by the dash-dot lines. The value of propagation loss at the receiver (square) is compared and when the values at the corner are within ±1.5 dB of the centre, the propagation loss within that cell is considered uniform.

Five propagation loss values were generated for each grid square along the transect - each of the four corners and the centre of the cell. These were plotted against the distance from the centre of the cell to the receiver to produce a propagation loss curve (e.g. Fig. 5.4). The propagation loss curves from each individual source/receiver pair and the resulting propagation loss and grid square distance curves were smoothed using a low pass second order Butterworth filter (N=2, f=0.01). This removes signal noise. The noise is an artefact of the coherent nature of the model used to predict propagation loss and can be removed by smoothing the signal in this manner (Robinson et al. 2014).
Propagation loss was considered uniform across a grid cell when the difference in propagation loss from the four corners fell consistently below a threshold value of 1.5 dB from the centre. The distance at which this occurred for each grid cell size was plotted to show the relationship between distance of uniform propagation loss and grid cell size. A second or third-order polynomial was fitted to the data points. The polynomial fit was chosen because it minimised the sum of the squared residuals for the datasets.

The 1.5 dB threshold was derived from information on the hearing capabilities of marine mammals and the known error associated with acoustic propagation models. The staircase methodology is used to determine auditory thresholds in marine mammals (Kastak and Schusterman 1998; Kastak and Schusterman 1999; Popov et al. 2013; Cunningham and Reichmuth 2016). This process plays back sound at different frequencies to determine when the animal responds. It is common for playback amplitude to be decreased in steps of 4 dB for every correct response and then increased in steps of 2 dB after the first missed response (Kastak and Schusterman 1998; Kastak and Schusterman 1999; Cunningham and Reichmuth 2016). This suggests, therefore, that at some frequencies marine mammals will be able to discriminate between sounds that differ in amplitude by 2 dB. As a result, some threshold level less than 2 dB would be appropriate. Hanna and Rost (1981) compared a parabolic equation model (the type of model used in this study) to measurements taken in the ocean. They reported mean errors of 1.5 dB, whilst Jensen (2011) compared different types of acoustic propagation model using a standard problem and reported 1.1 to 1.6 dB mean differences between the models. As a result, it could be expected that propagation loss values may vary by that magnitude as a result of model error rather than non-uniformity within a cell, consequently, 1.5 dB was considered an appropriate threshold.

The simulations took place along Transect A (Fig. 5.1) in two directions under a number of different conditions. Firstly, the receiver was placed at the westerly end of the transect and the grid was generated, as shown in Figure 5.2, to the
east. In this configuration the bathymetry shallows in the east and sound propagation is downslope. In the second configuration the sound propagation was reversed so the receiver was placed at the eastern extent of the transect and the grid was generated in a westerly direction. Sound would travel upslope in this configuration. Simulations are also repeated under summer and winter conditions because the water column is strongly stratified in the summer due to the development of a thermocline. This can significantly influence propagation loss (Shapiro et al. 2014). As a result, in summer the results are shown for a receiver depth of 20 m and 60 m to reflect conditions above and below the thermocline. The source depth was 7 m for all conditions. This is a typical estimated source depth for a large commercial ship (McKenna et al. 2012).

5.2.3 Acoustic propagation model

The parabolic-equation model RAMSurf\(^1\) (Collins 1993) was used to calculate propagation loss between each sound source and the receiver. This model is widely used for range dependent, low frequency, shallow water scenarios (Etter 2013). In this study, the horizontal and vertical step parameters for the acoustic model were fixed at 50 m and 0.5 m respectively for all simulations. These values ensure a convergent solution to the model given the frequency tested and ensure that all simulations are comparable. Simulations were conducted at a frequency of 125 Hz. This frequency is one of the centre frequencies for the 1/3 octave bands that are given by the EU Marine Strategy Framework Directive as important for monitoring shipping noise (European Commission 2008; European Commission 2017).

The three-dimensional oceanographic model POLCOMS was used to provide temperature and salinity data along each transect (Holt and James 2001). The oceanographic model had a horizontal resolution of 2 km and 30 vertical layers. This was used to calculate sound speed profiles in 2 km increments along the transects. This model, its implementation and associated bathymetric data are

\(^{1}\text{http://oalib.hlsresearch.com/PE/ramsurf/}\)
described in detail by Chen et al. (2013). The speed of sound through the water, given the temperature, salinity and depth was calculated using the nine term equation given by Mackenzie (1981) (Sec. 3.2.5.3). Seabed sediment data were provided by the EMODnet Geology project (http://www.emodnet-geology.eu).

The distribution of sediment types throughout the study area are shown in Figure 5.3. The geoacoustic parameters for each sediment type were selected from known geoacoustic values in the following sources, Hamilton (1980), Lurton (2002) and NURC (2008). The sediment grain size, and percentage of clay, gravel and sand for each EMODnet sediment type (Long 2006) was used to select an appropriate geoacoustic value from the above sources given a similar sediment type description. The selected geoacoustic parameters for each sediment type are given in Table 5.1.

Figure 5.3: Map of the sediment types in the Celtic Sea using five sediment classes as described by Long (2006). Information modified from EMODnet Europe seabed substrate data, scale 1:250000 (©EMODnet Geology, European Commission, 2016, downloaded 2016-07-21)
Table 5.1: Geoacoustic parameters passed to RAMSurf model. Appropriate values were selected from published geoacoustic studies as indicated below. \( C_p \) - P-wave sound speed, \( \alpha \) - P-wave attenuation.

<table>
<thead>
<tr>
<th>Sediment</th>
<th>Density ( (g\text{cm}^{-3}) )</th>
<th>( C_p ) ( (\text{ms}^{-1}) )</th>
<th>( \alpha ) ( (dB^{-1}) )</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mud/Muddy Sand (Hamilton 1980; NURC 2008)</td>
<td>1.740</td>
<td>1615</td>
<td>1.00</td>
</tr>
<tr>
<td>Sand (Hamilton 1980; NURC 2008)</td>
<td>1.941</td>
<td>1749</td>
<td>0.80</td>
</tr>
<tr>
<td>Coarse Sediments (NURC 2008)</td>
<td>2.000</td>
<td>1800</td>
<td>0.60</td>
</tr>
<tr>
<td>Mixed Sediments (Hamilton 1980; Lurton 2002)</td>
<td>2.034</td>
<td>1836</td>
<td>0.90</td>
</tr>
<tr>
<td>Rock (NURC 2008)</td>
<td>2.200</td>
<td>2400</td>
<td>0.20</td>
</tr>
</tbody>
</table>

5.2.4 Development of adaptive grid

The adaptive grid was generated for a case study area of 160 × 160 km (80 km from receiver to edge of grid) around a receiver located centrally in the Celtic Sea (latitude: 51.0, longitude: -6.7). The grid cell size used at a particular distance from the receiver was determined using the grid cell size/distance of uniform propagation loss relationship developed in Methods Section 5.2.2. To keep the computational development of the grid simple and to follow the convention of typical ship noise studies, a square grid was developed using square grid cells (Erbe et al. 2014; Marine Management Organisation 2015). This places constraints on how closely the resulting grid will adhere to the distance/grid size relationship. The relationship is maintained along the north/south and east/west axis. However, the distance between the source and receiver is greater over a diagonal axis. Therefore, the adaptive grid does not reflect the relationship at all points on the grid. Furthermore, when creating the adaptive grid it is not possible to fit certain grid cell sizes into the grid before the relationship developed in Methods Section 5.2.2 indicates the next grid size should be implemented. For example, the December upslope relationship (Sec. 5.3.1, Fig. 5.5a) shows that a grid cell size of 4.0 km should be used when source and receiver are between 33.7 km and 35.4 km apart but this distance is only separated by 1.7 km. A single row of 4.0 km grid cells would extend to 37.7 km. There are two approaches to this problem. Firstly, when this situation arises skip to the next grid size that can be drawn in the available distance i.e. skip using 4.0 km cells and use 4.5 km grid cells. The relationship shows these
should extend to 42.7 km, and therefore, two rows can be used between 33.7 and 42.7 km. Secondly, the next grid size (in half kilometre increments as used to develop the relationship) can be used regardless and a check can be implemented after each grid row is drawn to see which grid size is appropriate, i.e. a 4.0 km grid would be drawn from 33.7 km to 37.7 km and then a 4.5 km grid would begin at 37.7 km rather than 35.4 km. The second approach prioritises accuracy over efficiency because the optimal or smaller than optimal grid cell size is always selected. If a larger than optimal grid size was chosen propagation loss would vary across the cell potentially compromising accuracy. The second method was used to draw the adaptive grids as part of this study.

5.2.5 Assessment of efficiency and accuracy

The performance of the adaptive grid was tested by comparing it to a grid of the same size (160 × 160 km) where all grid cells are a uniform 5 × 5 km and a grid where all grid cells are 1 × 1 km. These grids were generated around the same receiver as the adaptive grid located at (latitude: 51.0, longitude: -6.7). A 5 km grid has been used in noise mapping studies (Erbe et al. 2014) indicating this grid size can achieve adequate computational efficiency for implementation in real world studies. A 1 km grid is a finer resolution than that typically used in ship noise mapping. It is, therefore, considered as the reference in terms of accuracy. The computational cost of executing the model for each grid was assessed by calculating the number of transects connecting the centre of the grid cells to the fixed receiver. The number of transects is indicative of how many input files are required, how many times the model would need to be executed and how many output files are created and must be processed, and therefore, is representative of computational efficiency.

Accuracy was determined by comparing the propagation loss at the location of the receiver from the sources located at the centre of each grid cell for the 1 km, 5 km and adaptive grid. These results are displayed using two techniques. Firstly, the propagation loss value at the receiver was plotted against the
distance between each source/receiver to produce a propagation loss distance curve. These data were smoothed using a Butterworth filter (N=2, f=0.01) to remove coherent noise from the signal. Additionally, the results of the simple geometric spreading models \((15 \log(r))\) and \(20 \log(r)\) where \(r\) is range from source in metres) are plotted for comparison. Secondly, the propagation loss for the adaptive grid and the 5 km grid were linearly interpolated to the points of the 1 km grid. As a result it was possible to calculate the absolute difference in the smoothed propagation loss between each point on the grid. The interpolation was a computational exercise to ensure that the matrices containing the results had the same dimensions. This ensured they could be easily compared but did not influence the resulting propagation loss values.

5.3 Results

5.3.1 Grid size and distance relationship

In order to determine the distance of uniform propagation loss for grid cell sizes between 0.5 and 20 km a number of propagation loss simulations were conducted. The resulting propagation loss curves are shown in Figure 5.4. The figure shows the propagation loss from each corner and the centre of the grid cells for the upslope transects with a receiver depth of 20 m and grid cell sizes of 1 km, 5 km and 20 km. The results for the downslope condition and plots for August at a receiver depth of 60 m are shown in Supplementary Material Figures C.1 and C.2. Figure 5.4 demonstrates that when the source and receiver are close together (0-25 km) the propagation loss at each of the corners and at the centre of the grid cell differs, at grid cell sizes of 1 km, by \(\sim 3\) dB and at larger grid sizes by \(\sim 15\) dB. This difference in propagation loss decreases as the distance between the source and receiver increases. As described in Methods (Sec. 5.2.2), propagation loss within the grid cell is considered uniform when propagation loss from the centre of the grid cell to the receiver and each corner to the receiver is approximately equal (\(\pm 1.5\) dB from centre value). The results (Fig. 5.4) show that as the grid cell size becomes larger the distance at which
propagation loss becomes uniform across the cell increases. This is illustrated in Figure 5.4 for both December and August. The distance of uniform propagation loss (the point at which each line comes together as one) is \( \sim 25 \text{ km} \) for the 1 km cells in December and August (Fig. 5.4a, 5.4b), and \( \sim 50 \text{ km} \) for 5 km grid cells in December (Fig. 5.4c). In August there was no point at which propagation loss became uniform for the 5 km grid cells. Figure 5.4d appears to be uniform for a short distance but then this uniformity breaks down again. This is most likely a result of the variability in water column properties in the Celtic Sea during summer. This was not the case for the downslope condition suggesting that the distance of uniform propagation loss is further influenced by the bathymetry of the transect. Figure 5.4e and 5.4f show that at grid cell sizes of 20 km the difference in propagation loss between each point does not come below the 1.5 dB threshold even when source and receiver are separated by 200 km.

The distance at which propagation loss becomes uniform across the grid cell was extracted for all conditions at each grid cell size. This is shown in Figures 5.5 and 5.6 for December upslope and downslope at a receiver depth of 20 m, and August upslope and downslope at a receiver depth of 20 m and 60 m. In December at 200 km the indicated grid box size is between 10 and 14 km (Fig. 5.5a, 5.5b). However, in August the indicated grid box size at 200 km is between 4 and 8 km for upslope and downslope at 20 m and 60 m (Fig. 5.5c, 5.5d, Fig. 5.6a, 5.6b). The smaller grid cell sizes in August are most likely due to the variability in ocean properties during August in the Celtic Sea. Sound propagation is determined by oceanographic conditions, and changes in properties such as temperature and salinity will result in changes in propagation loss values. Small grid cell sizes are required to capture this variation. The Celtic Sea is well mixed in the winter months (Pingree 1980). A more uniform sea in December allows the use of larger grid cells.

In order to predict the expected grid cell size at different distances, the relationship between distance of uniform propagation loss and grid size was characterised by a second or third-order polynomial. This generally indicates an
initial rapid increase in cell size over a short distance, followed by a period of less rapid change where a certain cell size can be used for greater distances. The maximum distance considered here was 200 km. The graph could be extended to greater distances in larger seas. However, the contribution of ships over 200 km distance from the receiver is likely to have negligible impact on the receiver. For example, in this study at 200 km propagation loss levels are between approximately 110 - 140 dB and sound pressure levels of ships are typically between 170 - 190 dB re $1\mu Pa$ (McKenna et al. 2012; Veirs et al. 2015) resulting in possible noise level contributions only between 30 and 80 dB re $1\mu Pa$ at this distance.
Figure 5.4: Propagation loss at each corner and the centre of grid cells for the upslope condition for grid cell sizes of 1, 5, and 20 km in December and August at a receiver depth of 20 m. When the corner values come to within 1.5 dB of the centre consistently, propagation loss is considered uniform (vertical black line). As distance between the source and receiver increases the difference in propagation loss between each corner and centre decreases until uniform. As the grid sizes become larger the distance of uniform propagation loss becomes much greater. For the 20 km grid cell sizes (e,f) there is still a large difference between each corner and centre at 200 km and at no point is propagation loss considered uniform. It is also possible to note that in December maximum propagation loss is $\sim 110$ dB but in August this value is $\sim 140$ dB.
Figure 5.5: The distance at which the difference in propagation loss between each corner and the centre of the cell is below 1.5 dB for different grid cell sizes. Plots are shown for the upslope and downslope conditions at a receiver depth of 20 m for December and August. Points are smoothed using a 2nd or 3rd order polynomial. The maximum grid cell size shown (x axis), is the maximum grid cell size for which propagation loss becomes uniform for that condition. This is greater in December than August. The dashed line indicates the maximum grid cell size that would be used when the source and receiver are separated by a distance of 80 km for the two example adaptive grids shown in this study.

Figure 5.6: The distance at which the difference in propagation loss between each corner and the centre of the cell is below 1.5 dB for different grid cell sizes. Plots are shown for the upslope and downslope conditions in August for a receiver depth of 60 m. Points are smoothed using a 3rd order polynomial. The maximum grid cell size shown is the maximum grid cell size for which propagation loss becomes uniform for that condition (x axis). This is greater for the downslope condition than the upslope condition.
5.3.2 Adaptive grid

The adaptive grids were created based on the relationships between distance and grid size shown in Figures 5.5 and 5.6. However, as discussed in Methods Section 5.2.4, these relationships change very rapidly over the first 50 km and consequently, the resulting grids are conservative representations of these relationships. This conservative approach has meant the resulting adaptive grids tend to take on one of two forms despite the different relationships generated for each condition. The adaptive grids for the upslope December and August conditions at a receiver depth of 20 m are shown in Figure 5.7 as examples of these two forms. The adaptive grids for the remaining conditions are shown in Supplementary Material Figures C.3 and C.4. The adaptive grid shown in Figure 5.7a is based on the relationship shown in Figure 5.5a. The adaptive grid commences with 1 km grid cells and steps in half kilometre increments up to 3.5 km grid cells. However, at 80 km in Figure 5.5a the dashed line indicates that the grid box size should be closer to \( \sim 6 \) km. It would be possible to achieve this by using an approach for grid development (See Sec. 5.2.4), which moves to larger grid sizes more quickly. This approach was not taken here in order to preserve, as much as possible, the accuracy of the adaptive grid, which is important for applications such as environmental impact assessment and decision making. It was often not possible to just skip to a larger grid cell size because the size used needed to be a multiple of the total length of each grid edge. For example, one edge of the final adaptive grid produced here is 160 km. The next grid size used would need to divide this distance exactly (e.g. 32, 5 km boxes) to create a complete grid. As a result, the achievable grid cell size was an interplay between maximum cell size indicated by the grid size/distance relationship and the constraints of generating a continuous grid of square grid cells.

Figure 5.7b shows the adaptive grid for the upslope August relationship as displayed in Figure 5.5c. This grid covers the same area as the other adaptive grids developed, and used the same conservative grid development approach. The maximum grid cell size achieved is only 2 km with the majority of the grid
using a 1.5 km grid cell size. When compared with December, these smaller grid cell sizes reflect the different relationships produced in summer and winter as a result of the different environmental conditions in the Celtic Sea at this time. Furthermore, the dashed line (Fig. 5.5c) indicates that at 80 km a 2.5 km grid cell size would be optimal suggesting that the conservative approach has again led to the implementation of smaller grid cell sizes.
Figure 5.7: Example adaptive grids for a 160 × 160 km area of the Celtic Sea for the upslope conditions in December and August at a receiver depth of 20 m. Each dot indicates the centre of a cell, the size of which is shown in the key above the grid. Red dot indicates the receiver and each color indicates a new grid size.
5.3.3 Computational efficiency of adaptive grid

In order to assess the computational efficiency of the adaptive grid, it was compared to two grids with $1 \times 1$ km cells and $5 \times 5$ km cells respectively. Table 5.2 shows the number of points in these two grids and the adaptive grids under the different conditions over a $160 \times 160$ km area. The $1$ km grid has $25$ times more points than the $5$ km grid. The adaptive grid achieves a $5$-fold reduction in the number of points in a $1$ km grid for December and a $2$-fold reduction in the number of points in a $1$ km grid in August. However, the adaptive grid has approximately five times the number of points as the $5$ km grid in December and approximately twelve times the number of points in August.

Table 5.2: The number of points and hence model executions required for the adaptive, $1$ km and $5$ km grids over a $160 \times 160$ km area. Depth refers to receiver depth and direction to the direction of sound propagation. The maximum distance between the receiver and edge of the grid is $80$ km in a straight line north, south, east or west. The number of points is used as a proxy for computational cost. The number of points does not change with month, receiver depth or direction of sound propagation for the grids with $1$ km and $5$ km grid cell sizes.

<table>
<thead>
<tr>
<th>Cell Size</th>
<th>Month</th>
<th>Depth</th>
<th>Direction</th>
<th>Points</th>
</tr>
</thead>
<tbody>
<tr>
<td>$1$ km</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>25600</td>
</tr>
<tr>
<td>$5$ km</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1024</td>
</tr>
<tr>
<td>Adaptive</td>
<td>December 20</td>
<td>-</td>
<td>downslope</td>
<td>5356</td>
</tr>
<tr>
<td>Adaptive</td>
<td>August 20</td>
<td>-</td>
<td>downslope</td>
<td>5356</td>
</tr>
<tr>
<td>Adaptive</td>
<td>August 60</td>
<td>-</td>
<td>downslope</td>
<td>5056</td>
</tr>
<tr>
<td>Adaptive</td>
<td>December 20</td>
<td>-</td>
<td>upslope</td>
<td>5356</td>
</tr>
<tr>
<td>Adaptive</td>
<td>August 20</td>
<td>-</td>
<td>upslope</td>
<td>12752</td>
</tr>
<tr>
<td>Adaptive</td>
<td>August 60</td>
<td>-</td>
<td>upslope</td>
<td>13132</td>
</tr>
</tbody>
</table>

5.3.4 Accuracy of adaptive grid

The adaptive grid provides an important computational saving over the grid with $1 \times 1$ km uniform grid cells but is not more efficient than using a grid with $5 \times 5$ km cells. As a result, it is necessary to examine more closely the advantages of using the adaptive grid, the most pertinent of which, is the accuracy of the grid. Variation in propagation loss within the cell can result in over- or under-estimation of the total propagation loss. The potential advantage of the adaptive grid is that it can achieve computational efficiencies without a decrease in accuracy by using a smaller grid cell size close to the receiver and a larger cell
size further away. Figures 5.8a and 5.8b show the smoothed propagation loss at the receiver for each transect in the grid. It is possible to see that over the first 25 kilometres the 1 km and adaptive grid differ by a maximum of 0.5 dB. This is due to the adaptive grid taking the form of the 1 km grid for much of this range. The 1 km and adaptive grid differ from the 5 km grid by between 3 and 13.8 dB in this range. At greater distances the difference between all three grid sizes is reduced and all vary by not more than 2 dB. Figures 5.8a and 5.8b also show the propagation loss calculated using the geometric spreading model. The difference between this and the RAMSurf model, which is a typical model for shallow low frequency problems, is \( \sim 10 \) dB in December close to the receiver but as large as \( \sim 20 \) dB far from the receiver in August. The 15 log model for both conditions consistently underestimates propagation loss and hence overestimates noise. The opposite is true of the 20 log model in December (Fig. 5.8a). It consistently overestimates propagation loss and underestimates noise levels. However, in Figure 5.8b the 20 log model overestimates propagation loss and hence underestimates noise close to the receiver but at greater distances the opposite is true.

It is also possible to view how the accuracy of the propagation loss compares over the spatial grid. The absolute difference in the propagation loss between the 1 km grid and the 5 km grid for December and August are shown in Figures 5.9a and 5.9b respectively. To demonstrate the change in propagation loss clearly, values greater than or equal to the threshold of 1.5 dB are shown in black and those less than 1.5 dB are shown in white. It is possible to see that some of the greatest differences in propagation loss calculations can be found when source and receiver are close together at the centre of the map. Figures 5.9c and 5.9d show the absolute difference in propagation loss at the receiver between the 1 km grid and adaptive grid for December and August respectively. When compared with the previous figures of the same month 5.9a and 5.9b, it is possible to see that the adaptive grid reduces the number of points that differ from the 1 km grid by 1.5 dB or greater. This is particularly evident when source
and receiver are close together. This demonstrates the improvement in accuracy achieved by the adaptive grid. In December and August 3.5% and 16.1% of the 25600 points in the 1 km grid have a difference of greater than or equal to 1.5 dB respectively when compared to the 5 km grid. This is in contrast to 1.4% and 9.2% when comparing the 1 km and the adaptive grid in December and August respectively.

Figure 5.8: Smoothed propagation loss (Butterworth filter N=2, f=0.01) for all points in the 1 km, 5 km and Adaptive grid for the upslope condition in December and August at a receiver depth of 20 m. Over the first 25 km, the 5 km grid differs by up to 13.8 dB from the 1 km grid while the adaptive grid only differs by a maximum of 0.5 dB.

5.4 Discussion

This study aimed to reduce the computational cost and improve the accuracy of modelled ship noise level predictions by replacing the aggregation of ships using a uniform grid with an adaptive grid. The results demonstrate that, over a 160 × 160 km area the adaptive grid reduces the computational cost 5 fold in December and between 2 and 5 fold in August when compared to a 1 km grid. The 5 km grid reduces computational cost 5 fold again. However, over the first 25 km the 5 km grid produces errors of up to 13.8 dB when compared to the accurate but inefficient 1 km grid. The newly developed adaptive grid generates much smaller errors of less than 0.5 dB while demonstrating high computational efficiency. At greater distances the difference in propagation loss between the 1 km grid and the adaptive grid increases to similar levels as the difference
Figure 5.9: The absolute difference in propagation loss at the receiver between the 1 km grid and the 5 km grid, and the 1 km grid and the adaptive grid. Black points indicate where the difference in absolute propagation loss at the receiver is greater than or equal to 1.5 dB. Receiver was located at centre of grid. Grid lines mark uniform 5 km cells. There are 25600 points in the grid. The percentage of these with error greater than or equal to 1.5 dB are (a) 3.5%, (b) 16.1% (c) 1.4% and (d) 9.2%.

between the 1 km and 5 km grids. The adaptive grid reduced the computational cost of obtaining noise level predictions while maintaining a higher level of accuracy in the area close to the receiver when compared to the 5 km grid.

The reduction in computational cost achieved by the adaptive grid is potentially a realisable reduction in the monetary cost of completing environmental impact assessments, monitoring activities and scientific studies that can be recouped by the businesses or universities that fund such activities. This saving is most likely to be found due to a reduction in the amount of time taken to complete the activity, which could also increase the speed of decision making by management organisations. It is difficult to define exactly how much of a time saving can be achieved by using the adaptive grid because baseline times are dependent on many factors such as the model used, computational power available and the
efficiency of code used to generate input files and execute the model. However, the time savings are likely to be of the same magnitude as the reduction in the number of transects that must be calculated. For example, if it is assumed that to complete a single transect takes two seconds, the 1 km grid with 25600 transects would be complete in approximately 14 hours, the 5 km grid with 1024 transects would take 34 minutes and for a typical adaptive grid of 5356 transects (Tbl. 5.2) it would take 3 hours. In this scenario the adaptive grid would reduce the time required in comparison to the 1 km grid by 11 hours.

Furthermore, the ability to improve the efficiency of acoustic propagation model predictions is important to facilitate the implementation of more sophisticated models as part of regular ship noise mapping and assessment activities (Marine Management Organisation 2015). Farcas et al. (2016) demonstrated that when compared to the model RAM, geometric spreading laws, which are widely used as part of legislative compliance assessments because they are quick, underestimate noise close to the source but overestimate noise far from the source. The results here show that the geometric spreading laws ($PL = N\log(r)$) followed similar trends when compared to the results from the RAMSurf model, although whether it under- or over-estimated noise varied depending on the value of $N$ chosen. The implementation of the spreading laws, in this Celtic Sea setting, may have negative implications for sensitive marine species and industry. Where noise is underestimated, even a precautionary approach to mitigation, may not be sufficient to put in place the correct protection for a species. This is particularly important at close range where noise levels, and hence potential harm are greatest (Richardson et al. 1995; Farcas et al. 2016). In general it is also important not to overestimate noise to avoid negative impacts such as restricting the growth of economically important industries or the cost of implementing mitigation strategies that are not required (Farcas et al. 2016). In shallow coastal environments where the source and receiver are very close together the introduction of more accurate models, which take account of important environmental variation, is a key mechanism by which the accuracy of
these noise level predictions can be improved (Farcas et al. 2016). The results of this study demonstrate that the adaptive grid can be utilised as part of a methodological toolbox for ship noise modelling that can make models based on advanced physical representations of propagation (e.g. RAM and Bellhop) more practical. This will only be enhanced by continued efforts to improve the acoustic propagation models themselves and the development of new models with reduced computational execution times (Sertlek and Ainslie 2014).

For ship noise mapping, sources are often represented using a density map, which spatially and temporally partitions the data (Erbe et al. 2012a; Porter and Henderson 2013; Marine Management Organisation 2015). The resulting maps are usually based on annual or monthly averages of ship numbers, which involve considerably fewer calculations than weekly or daily maps for the same time frame. However, there is known temporal variability in ship noise at these finer scales (Merchant et al. 2014b; Neenan et al. 2016). This can be addressed using the adaptive grid, which improves the efficiency and accuracy of spatial partitioning, potentially allowing a greater range of temporal variability to be captured for the same level of computational effort.

The level of accuracy required by acoustic propagation models can be considered, to some extent, dependent on the acceptable levels of uncertainty as defined by the user. The results of modelling activities are generally used to assess the impact of noise on marine ecosystems and may feed into planning applications, legislative compliance reporting and scientific conclusions regarding noise impacts (Hastie et al. 2015; Farcas et al. 2016; Chen et al. 2017). As a result, the over- or under-estimation of noise levels can have real consequences for marine life or industry. Crucially, the adaptive grid is more accurate than the 5 km grid when source and receiver are within 25 km of each other. This spatial relationship between a noise source and a receiver is important for assessing the impact of noise on marine life (Ellison et al. 2012). Richardson et al. (1995) explained the ‘zones of influence’ concept based on the distance between a receiver and a single source. The theoretical zones suggest
at what distances sound is audible, can cause behavioural disturbance, avoidance, hearing loss and injury in marine mammals. While this concept has its critics (Ellison et al. 2012; Van der Graaf et al. 2012), it is useful for highlighting the importance of accurate noise level predictions when source and receiver are very close together. Erbe (2002) suggested that a received level of 120 dB re $1 \mu Pa$ would cause a behavioural response in 50% of cetaceans. To use the adaptive grid as an example, propagation loss estimates when source and receiver are within 25 km is between approximately 50 and 80 dB. Typical broadband source levels for large ships are between 170 and 190 dB re $1 \mu Pa$ (McKenna et al. 2012; Veirs et al. 2015) resulting in approximate received ship noise levels for the adaptive grid presented here between 90 and 140 dB re $1 \mu Pa$ within 25 km, which is firmly in the region of the 120 dB re $1 \mu Pa$ predicted to cause behavioural disturbance (Erbe 2002). The rate of propagation loss at these close distances will change relative to the environmental conditions, but it serves to highlight that this zone is important for marine mammal protection when considering the impact of underwater noise from shipping. In addition, Erbe and Farmer (2000b) estimated zones of influence for beluga whales (*Delphinapterus leucas*) in relation to icebreaker ships and reported a zone of audibility between 38 and 78 km, masking between 14 and 71 km and temporary auditory threshold shift (TTS) between 1 and 4 km. Corresponding estimates were also presented by Erbe (2002). They examined whale watching boats travelling at high speed (51 km/h). These boats were audible to whales over ranges of 16 km, masked calls over ranges of 14 km and could produce TTS over ranges of 400 m. The appropriate zones of influence will be specific to noise source, species and environmental conditions but accurate noise level predictions are a vital ingredient to make sure these zones are appropriate. The proposed zones reported above show that the close distances (0 - 25 km) over which the adaptive grid offers improved noise level predictions are key for the protection of marine mammals.

The results demonstrate that the grid size/distance curves are generally
characterised by a 3rd order polynomial curve. The shape of each curve is determined by the underlying bathymetry, and the structure of the fronts and thermocline in the water column (Urick 1983; Lurton 2002; Shapiro et al. 2014). It is possible to see that the overall shape of each curve is influenced by the gentle upslope or downslope bathymetry and its interplay with water column structure. There is a marked difference between the relationships in August and those in December, specifically, for the upslope condition. In August, the variation in the water column is much greater, and therefore, much smaller grid sizes are required to capture this variation. The maximum grid size is 4 km in August compared to 14 km in December. There are similar albeit less extreme differences visible under the downslope condition. In December grid cell sizes reach a maximum of 10 km but only 8 km in August. This difference in the downslope condition is not reflected in the results for computational efficiency (Tbl. 5.2) because the difference is absorbed during the conservative approach to grid development (Sec. 5.2.4). As a result there is little difference in the adaptive grids under the downslope condition. In contrast, the difference in the upslope condition is quite marked, resulting in an opportunity to use separate grids for December and August allowing greater accuracy to be preserved in August and efficiency to be maximised in December. These observations have important implications for the applicability of these relationships to settings outside the Celtic Sea.

In order for the results to be applicable to a new area, this location should have the same general shallow water characteristics. In addition, careful consideration should be given to the structure of the water column. If the area develops a thermocline in summer, it is important to switch to a grid with finer grid cells. As described above, the conservative approach to grid development has, to some extent, removed the differences between the grids for the downslope condition. Despite the different grid size/distance relationships (Fig. 5.5 and 5.6) for all the downslope conditions and the December upslope condition, Table 5.2 shows that the number of points in all cases are between
5036 and 5356. This similarity suggests that the adaptive grids developed here may be applicable to a number of other temperate shallow water settings. However, given the influence of upslope propagation, they would not account for very steep or sudden changes in bathymetry, and therefore, there must be careful consideration of the physical properties of any new site before the adaptive grid is applied. In an area that is very different to the Celtic Sea where many projects in the same area are likely to be required, developing specific relationships between grid cell size and distance may be warranted.

There are a number of possible applications for the adaptive grid. It can be utilised to assess the exposure of individual animals or around a single receiver to validate field measurements (Robinson et al. 2014; Chen et al. 2017). Alternatively, it can be implemented around multiple receivers to undertake ship noise mapping. The test scenarios in this study have placed a fixed receiver at the centre of the grid. If the receiver was to move or there was more than one receiver, as would be the case when examining marine fauna or creating a ship noise map, it would be necessary to regenerate the adaptive grid around each receiver at each location. It is important to note that the grid generation process does not require the acoustic propagation model to be executed. In this study, grid generation at a new receiver location took on average 0.09 seconds. In comparison to the estimated theoretical time saving of 11 hours per receiver achieved by implementing an adaptive grid, there is still a considerable improvement in efficiency. When creating a ship noise map, as with maps using traditional grids, propagation loss must be calculated between each source/receiver pair (Erbe et al. 2012b). In traditional mapping activities, one transect may pass through several receivers, reducing the number of transects required. This will occur less often when using an adaptive grid because grid cells will be offset. Despite differences in the methodologies, there will still be net savings in computational efficiency because the adaptive grid also reduces the number transects required. It is difficult to suggest a general level of efficiency that can be achieved because the reduction in transects depends on the size of
the area mapped. The largest cell sizes in the adaptive grid provide the greatest benefit (i.e. reduction in the number of transects compared to smaller uniform grids) but these are only used at greater distances. Therefore, the larger the area mapped, the greater the improvement seen in efficiency. Importantly, when noise source and receiver move around the environment, the regeneration of the grid would always result in the smallest grid sizes around the receiver, and therefore, provides the finest estimations of propagation loss when the source and receiver are close together regardless of the oceanographic variation in an area. The ability to maintain a fine grid structure when source and receiver are close together improves accuracy when compared to uniform grid sizes, as shown in this study.

Nevertheless, depending on the execution time of the model for a single transect, such ship noise mapping can still be a time consuming process. Potentially, there is an opportunity to find a greater level of efficiency by increasing grid cell size more quickly over certain ranges. The variation in propagation loss in space is determined by range from the source (i.e. geometry) and absorption, over which environmental properties can have a significant influence (Urick 1983). It is environmental influences such as temperature and bathymetry that result in deviations from the general geometric spreading laws that describe propagation in homogeneous waters (Urick 1983). In such environmentally uniform areas, there is the potential for the use of much coarser adaptive grids, which change to larger grid cells more quickly, based on geometry alone. It may be possible to use coarser grids and account for environmental variation by using some factor to weight the propagation loss values. However, it is likely to require significant work to determine this relationship. Alternatively, in a region where the environmental variables do not differ in space the implementation of range independent models with faster execution times may improve the efficiency of the results and make mapping exercises with multiple receivers more tractable (Porter and Henderson 2013). However, in environmentally variable regions using such models and coarser grids, even though faster than RAM, will often
produce inaccurate results. Specifically, in the Celtic Sea it is known that spatial and seasonal variation in the temperature structure of the water column results in marked differences in propagation loss calculations (Shapiro et al. 2014).

As mentioned above, the adaptive grid presented here is a conservative implementation of the distance and grid size relationship. There is also the possibility to find greater computational efficiencies by using different approaches to adaptive grid development. The method of grid generation used in this study was governed by the shape of the adaptive grid and the individual cells, which are at present, square. As a result, the distance/grid size relationship is only maintained along the north/south and east/west axis. Where the relationship breaks down, the accuracy of the results may be compromised, which could account for some of the variation seen between the 1 km grid and adaptive grid when source and receiver were separated by larger distances (25 - 80 km). In the future it may be possible to implement curved grids that maintain the relationship throughout the 360 degree axis. Particularly, grids generated from triangular cells have the ability to be flexible and could smoothly migrate between cell sizes avoiding the use of 0.5 km steps in cell size, and maintaining the observed relationship between grid cell size and distance at all points on the grid (Chen et al. 2006).

This study has been concerned with modelling approaches to predict underwater noise levels. However, acoustic propagation models are not a perfect reflection of reality. Hanna and Rost (1981) compared a parabolic equation model to measurements in the ocean and found mean errors of 1.5 dB. Ship noise predictions are also a multi-stage process and this study does not consider the errors that arise during these other stages, such as difficulties in characterising ship source levels (Wales and Heitmeyer 2002; Wittekind 2014; Farcas et al. 2016) and uncertainties in the environmental input data. However, the ability to predict noise levels for management purposes is invaluable (Boyd et al. 2011; Van der Graaf et al. 2012). It is not logistically possible to deploy hydrophones for real-world measurements at all points in the ocean. The use of
a model allows managers and regulatory bodies to make informed decisions about the likely impact of shipping noise and determine noise hotspots and quiet zones. The adaptive grid can assist in realising the potential of acoustic propagation models in management settings by helping to make it a more efficient and practical process with the highest levels of accuracy possible.

This study aimed to improve the efficiency and accuracy of ship noise predictions using acoustic propagation models by developing a method which uses an adaptive grid to spatially partition ships. Over an area of $160 \times 160$ km the adaptive grid reduced the number of model executions 5 fold in December and between 2 and 5 fold in August. A similar level of computational efficiency was achieved with a coarse 5 km grid. However, over the first 25 km the 5 km grid produces errors of up to 13.8 dB when compared to the 1 km grid. The newly developed adaptive grid generated much smaller errors of less than 0.5 dB but also demonstrated high computational efficiency. As a result, the adaptive grid provides the ability to maintain or improve the accuracy of noise level predictions, and at the same time, increase the efficiency of the modelling process. This is a potentially important reduction in the cost of undertaking modelling activities and can help management stakeholders use the most accurate and sophisticated modelling approaches. This can help safeguard sensitive marine ecosystems from noise pollution and impose fair restrictions on industry by improving the underwater noise predictions that inform management activities.
6.1 Summary of main research findings

The thesis presents three-dimensional predictions of ship noise exposure for grey seal adults and pups. The pups were primarily tagged in the Celtic Sea, an area with dynamic oceanographic properties. Vessels in this area were largely constrained to shipping lanes in the east of the study area away from seal tracks. The seals experienced median m-weighted 24-hr cumulative sound exposure levels ($cSEL_{24}$) of 143 dB re $1\mu Pa^2s$. The adult seals were primarily tagged in the English Channel, a pinch point for busy shipping lanes from the Atlantic Ocean. They experienced higher median $cSEL_{24}$ of 159 dB re $1\mu Pa^2s$. The predictions of noise exposure, which accounted for horizontal and vertical changes in seal movement and water column properties, were used to assess if seals would experience auditory damage in the form of temporary threshold shift, and to investigate if the diving behaviour of the seals was influenced by exposure to high levels of shipping noise. The efficiency of the methodology utilised in this thesis was improved by developing a new method to aggregate ships into a grid for ship noise modelling. The main findings of the thesis are highlighted in the following sections.

6.1.1 The influence of shipping traffic and seal behaviour on predicted exposure values

It was not possible to give a direct comparison of exposure between English Channel adults and Celtic Sea pups due to the confounding factors of region and age. However, it is reasonable to suggest that differences in shipping traffic, the at-sea movement of seals and their habitat use was, to some extent, driving the received exposure levels of the two groups.

An analysis of shipping traffic contributing to the received noise levels of seals every 15 minutes, demonstrated that exposure was influenced by the maximum ship source level and closest point of approach of ships in that 15 minutes as well as the multivariate interaction of these values and the number of ships within
120 km of the seal. The most influential variable in the model was maximum ship source level (i.e. the highest source level of any ship within 120 km of the seal in a 15 minute period). The results highlighted that close approaches by ships with higher estimated source levels occurred more frequently for the English Channel seals leading to higher estimated exposure levels in those individuals.

Seals were highly mobile within the sound field and experienced lower noise levels at the surface and at the bottom of dives due to scattering and absorption at these boundaries during some dives. The seals studied in the English Channel conducted directed travel from haul-out sites across the busy shipping lanes, but seals studied in the Celtic Sea tended not to utilise the busiest shipping areas in that region. However, it is not known if shipping traffic had any influence on this pattern of habitat use.

6.1.2 Auditory damage in grey seals exposed to shipping noise

The predicted exposure of seals to shipping noise with or without the consideration of effective quiet did not exceed best evidence thresholds for temporary threshold shift (TTS) and hence, permanent threshold shift. This suggests that, given our current knowledge of auditory damage in phocid seals, grey seals in the Celtic Sea and English Channel were not at risk from auditory damage during the study. Only 9 of 18 seals on 18 of 86 days experienced sound pressure level values above effective quiet when set at 124 dB re 1 µPa, and cSEL_{eq} for these seals was between 141 and 169 dB re 1 µPa^2 s. The mean exposure duration above effective quiet was 38.57 (SD = 47.86) minutes. This is considerably less than the 24-hr cumulative period currently used in legislation.

6.1.3 Behavioural changes shown by grey seals exposed to shipping noise

While the results suggest no auditory damage occurs in the seals studied, changes in the diving behaviour of grey seals were seen during the most acute exposures to shipping noise experienced by seals. To date, at-sea behavioural changes by seals in response to shipping noise has been understudied.
However, the results presented here show that the English Channel seals increased the ascent rate of benthic dives *after* high ship noise exposure, and *during* high noise exposure for shallow dives. However, the Celtic Sea pups decreased the descent rate of pelagic dives *after* exposure to periods of high ship noise. This may be consistent with a surfacing or swimming away response. As expected, the differences were subtle and subject to individual variation between the seals but the results support anecdotal observations by Mikkelsen et al. (2019) and Chen et al. (2017) of changes in the diving behaviour of seals in response to shipping noise.

### 6.1.4 Modelling underwater noise from shipping

The approach utilised in the study produced fine-scale estimates of received noise levels at the location of the seal. However, the process was time consuming (8 hour execution time for each 24 hour of track) due to the range-dependent propagation model, large number of ships and the many depth levels required for modelling. The adaptive grid presented in Chapter 5 improves the efficiency and maintains the accuracy of ship noise modelling that utilises detailed range-dependent models. Efficiency increased 5 fold in December and between 2 and 5 fold in August. The adaptive grid resulted in errors of less than 0.5 dB when compared to a fine-scale grid with 1 km grid cells. A grid with 5 km cells resulted in errors up to 13.8 dB when compared to the 1 km grid. The adaptive grid makes more accurate models potentially viable for the purpose of risk assessment and the monitoring of shipping noise.

### 6.2 Study limitations

It is recognised that the methodology has some limitations and the results should be viewed in this context. The modelling approach produced detailed predictions of noise with a sophisticated range-dependent acoustic propagation model ideally suited for shallow water, low frequency propagation scenarios (Etter 2013). This approach allowed the use of existing data from seal tags to conduct
an historic reconstruction of noise levels experienced by seals (Thompson 2012; Huon et al. 2015). This was a logistically efficient use of data and ensured that the lowest number of wild animals possible were subject to potentially stressful scientific interference. The use of the modelling approach allowed the calculation of noise at the location of the animal, which links noise levels and behaviour more closely than measurements with fixed hydrophones. However, there is still some error associated with this methodology. Particularly, the interpolation of dive locations between GPS fixes from tags introduces uncertainty regarding the true location of the seal and hence noise levels. This was minimised by restricting the amount of time allowed between GPS fixes (Section 3.2.2). The development of a long-term acoustic recording tag for use on seals offers opportunities to measure noise levels and behaviour simultaneously (Mikkelsen et al. 2019). This would reduce such errors from modelling noise levels, and would be useful in exploring the behavioural responses of seals to noise more closely. However, the utility of modelling noise should not be underestimated and complements the use of acoustic recording tags.

Acoustic tags (e.g. DTAGs) are useful to record the whole soundscape, encompassing a number of noise sources (anthropogenic and natural) depending on the frequency range of the tag. However, it can be difficult to isolate different noise sources and to remove flow noise from water passing the tag on the seal (Benda-Beckmann et al. 2016). Modelling noise provided the opportunity to isolate one noise source and predict its contribution to the soundscape and impact on seal behaviour. Modelling also allows noise in the area around the seal to be calculated, placing seal exposure in the wider context of shipping noise. Furthermore, the DTAG utilised by Mikkelsen et al. (2019) has not been deployed on many occasions, and therefore, studies utilising this device will require that new animals are tagged. In contrast, modelling approaches can easily be transferred to new areas and existing tag datasets, especially with the implementation of the adaptive grid produced in Chapter 5. This is a more cost-effective and efficient way to include a consideration of
animal behaviour in regulatory processes.

The DTAG provides an exciting opportunity to validate the approach used in this study against measured data. This would improve confidence in the predicted exposure values for seals. The sensitivity analysis was used to estimate uncertainty associated with a limited number of errors. However, the complete workflow, which includes all elements of the modelling process, would benefit from validation against measured data. This would need to be comprehensive because noise is highly variable in time and space. Point comparisons are unlikely to give a true understanding of the overall errors in the model. It was not possible to validate the methodology used in this thesis because there was no available data. However, the inter-quartile range of predicted $cSEL_{24}$, given the sensitivity analysis and uncertainty estimates, was between 3 and 7 dB for all seals. Furthermore, there are reasons to be positive about the methodology utilised here. Parabolic equation propagation models, particularly RAM, have been validated and benchmarked in a number of scenarios and general confidence in these models is high (Hanna and Rost 1981; Davis et al. 1982; Etter 2013; Wang et al. 2014). The results from the English Channel were also similar to hydrophone data in Falmouth Bay, suggesting that noise level estimates are within reasonable ranges of measured data (Merchant et al. 2012).

As discussed in Chapter 3, the estimation of ship source level could be improved. It is a priority for ship noise modelling that accurate source level estimates or measurements are available for the modern fleet (Brooker et al. 2015). Generally, acoustic propagation models take monopole source levels as input because it is most computationally tractable (Robinson et al. 2014). However, current noise measurement standards generate radiated source levels that are influenced by surface reflections (Gassmann et al. 2017). Therefore, measurements are often not suitable for input to models or as validation datasets for source level models which estimate monopole levels (Ainslie et al. 2009). The AIS data also contains errors from missing ships, incomplete transects and manual alteration of transmitted data by ship crews.
(Harati-Mokhtari et al. 2007). It is most accurate for large commercial ships but also includes a high number of fishing vessels. The results, therefore, most accurately reflect the low frequency noise from large ships. It is recognised that smaller boats are generally missing from AIS datasets but these emit noise at higher frequencies (Hermannsen et al. 2016). The results should be considered within this context because phocid seals are thought to hear at frequencies up to 86 kHz (National Marine Fisheries Service 2018). However, this would be of greater consequence for odontocetes with specialised high frequency hearing (160 kHz) (National Marine Fisheries Service 2018). Recreational boating is also likely to be more important for areas not assessed within this thesis such as the islands of the Iroise Sea Marine Park. In the English Channel, particularly, there can be confidence in the AIS data given the short distance to coastal AIS receivers and dominance of the commercial fleet.

Furthermore, there are many other noise sources in the ocean such as pile driving and seismic surveys, which could confound the assessment of the impacts of noise on behaviour. The impact of this on the results is expected to be minimal. The AIS showed no active survey vessels in the vicinity of the high noise events utilised in Chapter 4. Furthermore, the high noise events are spread in time and space, reducing the likelihood that concurrent noise events, if they did occur, were present in all high noise events. It should also be noted that additional noise sources could increase the overall noise exposure of seals increasing the likelihood of TTS as a result of exposure to the whole soundscape (Hastie et al. 2014; Hastie et al. 2015).

High noise events generally occurred when ships were close to seals. Therefore, it is not possible to completely separate the role of the presence of ships and the role of the noise emitted by the ship in catalysing behavioural reactions to noise. Approaching ships are known to disturb seals from haul-outs but vision plays an important role in-air (Jansen et al. 2010; Andersen et al. 2012). Underwater the noise emitted by ships is likely to be detectable before the visual cues of its presence due to the nature of light and sound in water (Au and Hastings 2008).
This suggests underwater noise has a significant role in detections and reactions to shipping traffic. However, it is necessary to explore such factors further.

### 6.3 Assessment of risk to grey seals from shipping noise

Overall, the results contribute to knowledge of the risks posed by shipping noise to grey seals. This is summarised in a risk matrix which estimates the likelihood of a potential impact occurring under different shipping noise conditions (Fig. 6.1). The matrix was developed using existing literature with the addition of the results produced in this thesis. Categories of potential impact were generated with reference to the impacts explored by Southall et al. (2007). They distinguished between brief, minor and biologically unimportant impacts, and sustained, meaningful responses. As such short-term impacts are thought to occur only a few times in a 24-hr period and do not recur on subsequent days (Southall et al. 2007). The matrix illustrates how the results from this thesis, which are summarised in Section 6.1, make a contribution to our understanding of the impacts of shipping noise in three categories; permanent physical injury, temporary physical injury and transient behavioural reactions. These contributions are highlighted in Figure 6.1 using blue boxes. The severity of these potential impacts was estimated based on the current literature (Southall et al. 2007; New et al. 2013; Harwood et al. 2016; Nabe-Nielsen et al. 2018). However, there is little understanding of how non-lethal impacts effect the long-term health and welfare of individual seals or population stability as a whole. This makes it very difficult to order the impacts by severity.

The m-weighted 24-hr cumulative sound exposure levels calculated for the English Channel and Celtic Sea indicate that noise conditions for these seals would be below the level of noise exposure required for permanent injury (Southall et al. 2007). This was also true of harbour seals in the Moray Firth (Jones et al. 2017). The exposure levels also did not reach levels high enough for temporary auditory damage for grey seals in the English Channel or Celtic Sea. In contrast, 20 harbour seals in the Moray Firth were exposed to predicted...
### Likelihood of Response

<table>
<thead>
<tr>
<th>Potential Impact</th>
<th>Likelihood of Occurrence</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mortality</td>
<td>No specific literature</td>
</tr>
<tr>
<td>Permanent Physical Injury (e.g. PTS)</td>
<td>24-hr cSEL values were not (\geq 201) dB re 1µPa for any seals</td>
</tr>
<tr>
<td>Temporary Physical Injury (e.g. TTS)</td>
<td>EC and CS did not exceed 181 dB re 1µPa (2) s; MF exceeded on some occasions</td>
</tr>
<tr>
<td>Long-term Habitat Abandonment</td>
<td>No specific literature</td>
</tr>
<tr>
<td>Short-term Habitat Abandonment</td>
<td>Evidence for abandonment of hauled-out seals but no specific literature for at-sea seals</td>
</tr>
<tr>
<td>Sustained Behavioural Reactions</td>
<td>No specific literature</td>
</tr>
<tr>
<td>Transient Behavioural Reactions</td>
<td>CS pups decrease dive descent rate at median maximum SPL levels of 111 dB re 1µPa; EC adults increase dive ascent rate at median maximum SPL levels of 122 dB re 1µPa</td>
</tr>
<tr>
<td>Call masking</td>
<td>Some evidence for frequency overlap between calls and shipping noise</td>
</tr>
<tr>
<td>Chronic stress</td>
<td>No specific literature</td>
</tr>
</tbody>
</table>

### References

1. Jones et al. 2017

### Figure 6.1: Matrix showing the potential likelihood of occurrence for potential impacts on grey seals from shipping noise. Blue borders indicate where the results produced in this thesis have contributed to knowledge on the impacts of shipping noise on grey seals. The majority of the matrix remains highly uncertain. Particularly the decreasing order of severity for impacts. This was estimated based on current knowledge. EC - English Channel; CS - Celtic Sea; MF - Moray Firth
levels above thresholds for temporary auditory damage when considering upper confidence limits (Jones et al. 2017). However, there is still some uncertainty associated with the likelihood of temporary injury because Jones et al. (2017) utilised a different methodological approach when compared to this thesis, and there may be differences between seal exposures in different locations. Figure 6.1 presents a single risk matrix based on results from several different species at different locations. However, the difference in the results presented in this thesis based on the region/age of the seals, highlights that risk is likely to vary by location and by species. It could potentially also differ by season depending on the oceanic conditions for sound propagation at the location of the seal and the seasonality of shipping traffic (Kavussanos and Alizadeh-M 2001; Fleming 2010; Jensen et al. 2015; Chen et al. 2017).

The results show that transient behavioural responses to shipping noise are likely under noise conditions between 111 and 122 dB re 1 $\mu$Pa in grey seals. This was similar to observations by Mikkelsen et al. (2019) that observed changes in the behaviour of seals during an approach by a ship that reached 113 dB re 1 $\mu$Pa. The behavioural reactions studied here were limited to high noise exposure events, that while transient, occurred on average 1.12 (SD = 0.92) times per day across all seals. The results showed that behavioural change could start after or last longer than the exposure events but the length of time for which a behaviour occurred after an event was not studied and requires further exploration. There is still little information about the likelihood of sustained behavioural responses such as the cessation of foraging or habitat abandonment in response to shipping noise in seals (Sec. 2.5.1). This thesis presents only subtle changes in diving behaviour and hence no new evidence to suggest that such sustained changes are likely in response to shipping noise. Furthermore, in studies of pile driving noise the exclusion of seals from habitat close to the noise source did not last longer than the pile driving activities (Russell et al. 2016), and studies of seals that abandon haul-outs due to approaching ships show that they often return to the haul-out (Andersen et al. 2017).
However, shipping traffic has been present in the environment of seals for many decades (Ross 1976; Andrew et al. 2002), and therefore, it is difficult to study the influence of shipping on habitat use in seals because any change may have already persisted for some time and be the norm.

Moreover, the risk of mortality, call masking and chronic stress have not been addressed in this thesis but nevertheless seals may be vulnerable to these negative impacts. Initial evidence shows that the calls of captive grey seals overlap with the frequencies of shipping noise suggesting that call masking by shipping traffic is likely, but there is scope for further research regarding change in the communication space of seals (Asselin et al. 1993; Bagocius 2014). For example, calls can be associated with breeding, which in the context of the English Channel seals, may occur near haul-out sites at greater distance from shipping lanes than the trips studied in this thesis (Van Parijs and Kovacs 2002).

There is no specific evidence in the literature to suggest that shipping noise could result in the direct mortality of seals. The risk of mortality may instead come from ship strike of which noise may be a warning to seals rather than a contributing factor (Baker et al. 1998; Osinga et al. 2012). Shipping noise could become a factor indirectly in the mortality of seals through mechanisms related to energetic balance such as the loss of foraging opportunities, changes in the stress response and changes in the energetic cost of diving due to changes in behaviour (McHuron et al. 2017).

As highlighted at several points throughout this thesis, the link between impacts and the long-term health of individuals is not yet known. Reduced foraging opportunities can result in reduced pup recruitment in income breeders such as seals (McHuron et al. 2017), but initial studies of bottlenose dolphins (Tursiops truncatus) suggest that increases in boat traffic would have no population consequences because it resulted in no change in the health of individuals (New et al. 2013). However, it is also necessary to consider stressors from other sources which may have cumulative effects (Hastie et al. 2014; Cosgrove et al. 2016; Yon et al. 2019). For grey seals, metabolic challenges are not only related
to anthropogenic activities but also breeding, diving, lactating and moulting (Bennett et al. 2012). Pups particularly undergo a post-weaning fast that can be physiologically demanding (Noren et al. 2008). The hypothalamic-pituitary-adrenal (HPA) axis mediates the stress response but is also important to maintain energy balance. For example, glucocorticoids promote gluconeogenesis by facilitating the mobilisation of fat and protein reserves (Bennett et al. 2012). The stress response can increase glucocorticoid concentration, and therefore, alters energy allocation influencing fitness and survival (Blas et al. 2007; Shallin Busch and Hayward 2009). If stress is chronic, as may be the case with shipping noise, basal levels of stress hormones can increase (Creel et al. 2002). Sustained elevated levels of stress hormones have been linked to negative impacts such as immunosuppression, influencing the overall health of an individual (Svensson et al. 1998; Padgett and Glaser 2003). While there have been some studies on anthropogenic stressors and the stress response in phocid seals (Engelhard et al. 2002; Lidgard et al. 2008), there is no literature that deals specifically with chronic stress and shipping noise in this group.

6.4 Recommendations for policy on underwater noise from shipping

As highlighted in the literature review, underwater noise from shipping must be addressed as part of legislation such as the EU Marine Strategy Framework Directive (MSFD). However, there is still uncertainty around quantifying the impact of shipping noise for individual species and ecosystems, especially when trying to set targets for acceptable levels of noise, and when weighing the costs of mitigation measures against the negative ecological impacts (Merchant 2019). In the UK, this was evident in the most recent update on progress towards the ‘Good Environmental Status’ (GES) of UK waters as part of the MSFD (DEFRA 2019). For grey seals, a positive trend in the population suggests GES (SCOS 2018; DEFRA 2019). However, the report highlighted ongoing uncertainty
around what levels and frequencies of anthropogenic noise lead to negative impacts on populations, and therefore, it was still not possible to determine if GES had been achieved with respect to noise for any species or habitat (DEFRA 2019). The response to this from signatories to the MSFD is to establish ambient noise monitoring and mapping (Merchant et al. 2016). The results of this thesis highlight that the acute peaks in noise, as a result of close approaches by ships or groups of ships at greater distances, are important factors in the exposure of grey seals and in eliciting behavioural responses. The sound pressure levels only exceeded effective quiet under these conditions, and the seals exhibited changes in behaviour at these peak noise levels. Therefore, it is necessary to ensure that metrics for monitoring noise are sensitive to changes in the number and intensity of these peaks in noise. Metrics that temporally average noise may not be sensitive to changes in the number of these peaks experienced by marine life (Merchant et al. 2018). Although from a policy perspective, it is difficult to monitor changes in noise at the level of the individual animal, the use of exceedance levels, which indicate the percentage of time sound is above a certain level, could be helpful in capturing changes in peak noise levels that exceed background ambient noise (Merchant et al. 2018). Furthermore, risk mapping would provide opportunities to look at the changing spatial relationship between seals and ships, and therefore, the likelihood of experiencing high noise levels that may result in changes in the diving behaviour of the seals. In the North Sea and Atlantic seas around the UK and Ireland the JOMOPANS and JONAS projects respectively, aim to develop standardised systems to monitor ambient noise using measurements and modelling, and JOMOPANS supports the use of percentiles as a metric to summarise sound for monitoring purposes (Merchant et al. 2018).

The outputs of research and monitoring programmes addressing shipping noise are often depth averaged or single depth two-dimensional maps (Maglio et al. 2015; Sertlek et al. 2016; Sertlek et al. 2019). These are useful for monitoring trends and predicting underwater noise for policy purposes. However, the results
presented here indicate that in dynamically active oceanic regions the exposure of diving animals changes with depth. Therefore, two-dimensional maps, while useful for generalised basin scale monitoring, should be used with caution for species that utilise the complete depth profile of the water column, especially in areas where temperature and salinity are very variable. At locations of high spatial overlap between seals and shipping such as those identified by Jones et al. (2017) within 50 km of the coast, more fine-scale predictions that include depth would be required to make a complete assessment of the exposure and behavioural responses of pinnipeds to shipping noise. This could be particularly relevant for licensing and consent, where industrial projects such as wind farm construction, result in increased vessel traffic along defined routes and in particular zones which may overlap with seal populations at important sites such as those used for foraging (Anderwald et al. 2013). Modelling underwater noise at different depths increases the computational effort (Marine Management Organisation 2015). In this thesis the depth resolution was 1 m. With further research on the influence of depth on exposure levels, it may be possible to use coarser depth resolutions to make the inclusion of depth more efficient.

Similarly, the results show that the adaptive grid presented in Chapter 5 reduced the number of model executions required 5 fold in December and between 2 and 5 fold in August. Crucially, it achieved these improvements in efficiency while maintaining the accuracy of a finer grid with 1 km grid cells. This could facilitate the use of more sophisticated acoustic models in a regulatory setting. This is particularly relevant for areas where changes in ocean properties result in changes in the propagation of underwater noise (Shapiro et al. 2014). If regulators are able to make more accurate predictions and reduce uncertainty, they will be able to implement more targeted regulation that balances the needs of conservation and industry more exactly (Merchant 2019). Furthermore, the reduction in time required to complete a model of shipping noise for an area is potentially a realisable reduction in the cost of licensing for industry and regulators and could also help the decision making process to be more dynamic.
The results suggest that exposure levels are not high enough to induce TTS in grey seals in the English Channel or Celtic Sea but shipping is projected to increase in the coming years (UNCTAD 2018). Therefore, it is vital that such species are still considered in ongoing monitoring programmes (Allen et al. 2011). However, the results also highlight that seals changed their diving behaviour. Therefore, for grey seals, changes in diving behaviour are likely to be more important than TTS when looking to mitigate the risks of shipping noise. This is particularly relevant for policymakers. Regulatory organisations have focussed on quantifiable thresholds for temporary threshold shift (National Marine Fisheries Service 2018). Behavioural responses are often transient and vary widely between and within individuals based on context, experience and behavioural state, and therefore, are more difficult to quantify in terms of thresholds (Southall et al. 2007; Gotz and Janik 2010). However, policy that neglects the behavioural responses of seals to shipping noise will fail to mitigate the negative impacts that may result. In the service of informing policy, the predicted exposure values, and the magnitude of the exhibited behavioural responses to given sound pressure levels could be used as input for models which aim to predict the impact of individual effects on population stability (Nabe-Nielsen et al. 2018).

The 15 minute cumulative sound exposure levels were modelled using explanatory variables describing shipping traffic in relation to the seal location. The most influential variable determining exposure to shipping noise in the model was maximum ship source level (i.e. the highest source level of any ship within 120 km of the seal in a 15 minute period). The results suggest that to achieve the greatest decrease in the exposure of seals to shipping noise, it is best to direct efforts towards decreasing the source levels of the ships. Distance to the seal was also an important factor but given the far-ranging and diverse spatial usage of individual seals, changing the spatial relationship between seals and shipping is a more uncertain solution and it may be difficult to achieve consistently lower exposure levels (Thompson et al. 1991). In contrast, there are
a number of solutions for lowering the source levels of individual ships. Reductions in ship source levels can be achieved by changing the design of new ships, retrofitting old ships with quieting technology, decreasing the speed of ships and improving ship maintenance (IMO 2014). It is possible to achieve between 2 and 30 dB reductions in noise from cavitating propellers through the use of coating surfaces, changing the number of propeller blades, correcting propeller pitch and appropriate propeller design (Ebrahimi et al. 2019). However, the easiest method by which to decrease ship source level is to introduce a speed limit. A reduction in speed limit is a trade off with the increased exposure due to the ship travelling through an area more slowly. McKenna et al. (2013) reported the quietest noise levels of a ship recorded at different speeds was 8 knots, which was a 65% reduction in operational speed, a considerable drop in efficiency. In the Haro Strait, a speed limit of 11.8 knots was estimated to result in a 3 dB reduction in noise levels (Williams et al. 2019). However, vessel slowdowns as part of the Enhancing Cetacean Habitat and Observation (ECHO) program by the Vancouver Fraser Port Authority reported a reduction of 1.2 dB compared to baseline periods despite longer transit times (Joy et al. 2019). This was a voluntary program and only 37% of vessels achieved the desired speed of 11 knots. This highlights that speed reductions could be an effective method for reducing the exposure of grey seals in the Celtic Sea and English Channel but that vessel compliance is important for achieving reductions in noise and this needs co-operation from international and regional regulatory organisations such as the International Maritime Organisation (IMO), European Union and OSPAR Commission.

The adoption of these strategies assume that an absolute reduction in ship source levels is the only conservation goal for an ecosystem. There are in fact many competing priorities for both the shipping industry and for ecosystem managers. Ship quieting measures will impose a cost on the industry either through the purchase of new equipment or through losses in efficiency with speed reductions. In southern California, speed restrictions (10 - 12 knots) were
estimated to increase costs by 1.3 - 2.0 % due to an increase in transit time and higher fuel consumption (Gonyo et al. 2019). The shipping industry is also vulnerable to global economic fluctuations (Frisk 2012). The 2008 global economic downturn influenced volumes of trade and shipping rates, and companies have been operating in an increasingly tough market (UNCTAD 2018). The industry must also invest to meet binding regulations on atmospheric and ballast water emissions (IMO 1973; IMO 2004). The IMO has issued non-binding guidelines to help the industry address shipping noise but these are not necessarily a priority for the industry (IMO 2014; UNCTAD 2018). For ecosystems, the presence of ships may also be of consideration in marine planning for species at risk from ship strike. For example, significant mortality of the endangered North Atlantic right whale is caused by ship strike and fishery entanglements (Kraus 1990; Knowlton et al. 2012). In this scenario the relocation of shipping lanes and speed restrictions have shown to be effective in reducing mortality from ship strike (Vanderlaan and Taggart 2009; Laist et al. 2014). These management actions would also have influenced noise levels in the area. Noise may not always be a priority in such situations, and there are other issues facing grey seals in the English Channel and Celtic Sea such as by-catch which should be part of holistic policy planning (SCOS 2018).

The results show that the impacts of shipping noise on grey seals are subtle at present. However, the magnitude of the exposure and occurrence of a behavioural response do suggest that shipping noise could become a greater issue if traffic increases as predicted in the coming years (UNCTAD 2018). Noise levels should be addressed proactively given current evidence, especially because, as highlighted by the risk matrix (Fig. 6.1), impacts such as chronic stress are still unknown. It has been well documented that environmental policies serve ecosystems and societies best if they address problems proactively before they become a crisis (Davies and Brillant 2019). Box 6.1 summarises the recommendations arising from this thesis.
1. Shipping traffic was a key driver in the exposure level of seals. Given the model of 15 minute cSEL, the greatest reductions in the exposure of grey seals to shipping noise could be found by reducing the source level of ships.

2. Metrics to monitor shipping noise should be sensitive to changes in the number and sound pressure level of peaks in noise generated by high ship source levels and the spatial relationship between the seals shipping traffic.

3. The behaviour and life stage of seals should be central to assessing the risk of grey seals to shipping noise. At-sea movement patterns, age and habitat use played a role in determining exposure to shipping noise. However, this was linked to spatial variation in shipping traffic suggesting that regional variation in these properties should be factored into management plans and assessments at the scale of seal habitat use.

4. The results provide evidence of seals changing their behaviour in response to shipping noise but there was variation across this and other behavioural response studies in seals. Given current evidence it may be most important to consider the occurrence of a response rather than the direction of change (e.g. increase or decrease in dive ascent rate).

5. Shipping traffic is expected to increase, therefore, the impact of shipping noise on grey seals should be closely monitored. The behaviour of seals, the closest point of approach of ships, number of ships, ship source levels and the duration of exposure were important factors in determining if exposures reached levels high enough for auditory damage.

6. The use of an adaptive grid to aggregate ships for noise modelling purposes could improve the efficiency of such activities and facilitate the inclusion of depth profiles in noise exposure modelling for diving animals.
6.5 Suggested future research

This thesis has addressed gaps in our understanding of the exposure and behavioural response of grey seals to shipping noise and improved the efficiency of modelling shipping noise with range dependent acoustic propagation models. However, the results also highlight areas of future research that would assist policymakers and researchers. Firstly, in Chapter 3 the cumulative sound exposure levels were assessed in reference to a value of effective quiet. This value represents a sound pressure level to which the seals can be exposed continuously without inducing TTS or impairing recovery from TTS (Ward et al. 1976). The value utilised in Chapter 3 was an estimate based on the lowest value known to result in TTS, and is the only published value of effective quiet for phocid seals (Finneran 2015). However, examination of the results shows that $cSEL_{24}$ was sensitive to this parameter. Only 18 days across 9 seals had sound pressure level values above effective quiet when set at 124 dB re $1\mu Pa$, but this increases to 27 days across 10 seals if the value of effective quiet is reduced by 3 dB to 121 dB re $1\mu Pa$. The NOAA technical guidelines for assessing the effect of anthropogenic sound on marine mammal hearing does not implement the use of effective quiet because of this uncertainty (National Marine Fisheries Service 2018). This is a useful precautionary approach but resolving a value of effective quiet for phocid seals and other marine mammal species could help reduce uncertainty around estimates of TTS onset. Moreover, a better understanding of phocid hearing including the development and recovery of hearing from shipping noise exposures would be beneficial. At present, values are based on only a small number of data points from a limited number of fatiguing stimuli that may not represent the frequency and duration of shipping noise exposure (Kastak and Schusterman 1998; Kastak et al. 1999; Kastelein et al. 2012; Kastelein et al. 2013). Therefore, more audiograms and TTS onset data related to phocid seals would be useful. Particularly, data related to the hearing of pups and adults would help elucidate how vulnerable the younger population is to shipping noise and if they warrant separate consideration in regulatory threshold setting.
Thresholds for the onset of TTS are set at cumulative sound exposure levels over 24-hr periods. This cumulative period is arbitrary (Southall et al. 2007; National Marine Fisheries Service 2018) and has little meaning in the context of continuous noise exposure from shipping for seals. The results reveal that mean exposure above effective quiet was only 38.57 (SD = 47.86) minutes in 24 hours. Therefore, the 24-hr cumulative exposure period may result in overestimates of TTS by not considering periods of time where the noise levels are low enough to allow recovery from TTS inducing exposures (Finneran 2015). This is particularly relevant for pinnipeds that haul-out on land. During haul-out bouts seals can escape underwater noise exposure reducing the length of time they are continuously exposed to noise. However, the exposure of seals to noise from shipping when hauled-out is unknown, as is how the growth/recovery of TTS occurs across different mediums. The cumulative period has implications in policy settings, and the results imply that the 24 hour period is a precautionary approach for seals in the English Channel and Celtic Sea. Further research is required to resolve this issue. Particularly, assessing exposure over longer continuous track segments could elucidate patterns of exposure. Industry, particularly, would benefit from a more realistic cumulative period that is potentially less precautionary in some circumstances.

The thesis has demonstrated that seals change their diving behaviour in response to underwater noise from shipping. Chapter 4 examined several diving metrics but seals could also show horizontal changes in behaviour such as an alteration in swimming speed or directional bearing in response to noise (Aarts et al. 2017). Due to the variable nature of the results, behaviour should also be examined under different scenarios such as different geographical regions with different shipping characteristics. Particularly, the results highlighted that behavioural change may not be easily separated into before, during and after a ship noise exposure and hence the timing and length of the changes in diving behaviour could be examined further to isolate the onset of behavioural changes and elucidate the duration of the responses shown by seals.
As highlighted in the literature review, the next step in providing meaningful information for policymakers on acceptable levels of noise, is the link between the individual impacts, health and the wider population and ecosystem (Pirotta et al. 2018). The most feasible approach to addressing this is the implementation of various models of population consequences (New et al. 2013; Harwood et al. 2016; Pirotta et al. 2018). However, these models are difficult to parameterise. For grey seals or more generally phocid seals with respect to shipping noise, the results of this thesis could make a contribution to parameterising these models for future work. Particularly, such work would also be helped by elucidation of the risks not yet understood in the risk matrix (Fig. 6.1) such as chronic stress, mortality and call masking. These were discussed in more detail in Section 6.3.

6.6 Concluding remarks

The thesis investigated the exposure and behavioural response of diving grey seals to shipping noise and improved the methodology by which underwater noise from shipping can be modelled for such activities. In brief, the preceding chapters have found that grey seal adults and pups change their diving behaviour in response to underwater noise from shipping but that exposure levels were not high enough to cause auditory damage in the seals studied. The ship source level, number of ships and closest point of approach of ships were important factors in determining the exposure of the seals to shipping traffic in combination with a consideration of depth and seal movement. An adaptive grid resulted in a 2 to 5 fold improvement in the efficiency of the modelling methodology. The results contribute to an improved understanding of the impact of shipping noise on populations of grey seals that are dealing with multiple anthropogenic stressors, and can inform regulators in the development of effective policy in the future.
A.1 Model parameters for RAMSurf

Table A.1 gives the fixed parameter values utilised in the RAMSurf model as part of Chapter 3. As described in the main text, the parameter values for the RAMSurf model were selected to ensure the model converged at all frequencies utilised in the study. The values also ensured the model was as efficient as possible to execute.

Table A.1: Fixed parameter values for RAMSurf model. Frequency, maximum range, bathymetry, sound speed and sediment data are all variable for each input file and are determined as described in the main text.

<table>
<thead>
<tr>
<th>Parameter Name</th>
<th>Abbreviation</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Source Depth</td>
<td>zs</td>
<td>3 or 6 m</td>
</tr>
<tr>
<td>Receiver Depth</td>
<td>zr</td>
<td>all</td>
</tr>
<tr>
<td>Range Step</td>
<td>dr</td>
<td>50 m</td>
</tr>
<tr>
<td>Range Decimation Factor</td>
<td>ndr</td>
<td>1</td>
</tr>
<tr>
<td>Maximum Depth</td>
<td>zmax</td>
<td>500 m</td>
</tr>
<tr>
<td>Depth Grid Spacing</td>
<td>dz</td>
<td>0.5 m</td>
</tr>
<tr>
<td>Depth Decimation Factor</td>
<td>ndz</td>
<td>1</td>
</tr>
<tr>
<td>Maximum Depth of Output</td>
<td>zmpkt</td>
<td>200 m</td>
</tr>
<tr>
<td>Reference Sound Speed</td>
<td>c0</td>
<td>$1500 \text{ ms}^{-1}$</td>
</tr>
<tr>
<td>Number of Terms in Rational Approximation</td>
<td>np</td>
<td>6</td>
</tr>
<tr>
<td>Number of Stability Constraints</td>
<td>ns</td>
<td>1</td>
</tr>
<tr>
<td>Maximum Range of Stability Constraints</td>
<td>rs</td>
<td>0</td>
</tr>
</tbody>
</table>
A.2 Comparison of measured ship source levels and RANDI source levels estimates

The measured source levels of 22 ships were compared to RANDI source level estimates over the same frequency band. The broadband source levels (20 - 1000 Hz) were available on the SONIC Ship Underwater Radiated Noise Database (http://vesselnoise.soton.ac.uk/) and the RANDI model was used as described in Section 3.2.4. The predicted 1 Hz bands were integrated (approximated by summation) to generate a comparable source level figure. The speed and length of the ship were given in the SONIC database. All ship measurements included in the database were derived from the McKenna et al. (2012) study. The mean deviation between the two values was 9.48 (SD = 8.43) dB. The results of the comparison are shown in Table A.2. The results are influenced, however, by the methodology with which measured source levels were obtained. The measured levels may be as much as 12 to 27 dB lower than source levels measured using the ANSI prescribed conditions and values corrected for surface interference effects (Gassmann et al. 2017). Therefore, the comparison does not necessarily present like with like. A comparison of the RANDI model and ships measured using ANSI standards with a complex acoustic propagation model Kraken-C had median estimation errors of 0 (±7.1) (Peng et al. 2018). Therefore, the results presented here and hence uncertainty estimates based on this comparison may be an overly critical assessment of RANDI source level estimates.
### Table A.2: Comparison of measured ship source levels and estimated ship source levels using the RANDI source model.

<table>
<thead>
<tr>
<th>Vessel Name</th>
<th>Length (m)</th>
<th>Length (ft)</th>
<th>Speed (Kts)</th>
<th>Measured Source Level (db re $1 \mu$Pa)</th>
<th>RANDI Source Level (db re $1 \mu$Pa)</th>
<th>Difference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jabal Ali-1</td>
<td>148</td>
<td>486</td>
<td>8.94</td>
<td>184.9</td>
<td>173</td>
<td>-11.9</td>
</tr>
<tr>
<td>Energy Protector</td>
<td>182</td>
<td>597</td>
<td>13.8</td>
<td>182.4</td>
<td>184</td>
<td>1.6</td>
</tr>
<tr>
<td>NS Century</td>
<td>243</td>
<td>797</td>
<td>12.8</td>
<td>182.1</td>
<td>190</td>
<td>7.9</td>
</tr>
<tr>
<td>Chemtrans Sky</td>
<td>229</td>
<td>751</td>
<td>14.6</td>
<td>181.3</td>
<td>193</td>
<td>11.7</td>
</tr>
<tr>
<td>Star Express</td>
<td>182</td>
<td>597</td>
<td>13.8</td>
<td>178.5</td>
<td>184</td>
<td>5.5</td>
</tr>
<tr>
<td>Nave Ariadne</td>
<td>228</td>
<td>748</td>
<td>14.6</td>
<td>182.7</td>
<td>193</td>
<td>10.3</td>
</tr>
<tr>
<td>Yayoi Express</td>
<td>180</td>
<td>591</td>
<td>15.6</td>
<td>181.8</td>
<td>188</td>
<td>6.2</td>
</tr>
<tr>
<td>Global Ace</td>
<td>189</td>
<td>620</td>
<td>13.8</td>
<td>185.8</td>
<td>191</td>
<td>5.2</td>
</tr>
<tr>
<td>Red Lotus</td>
<td>229</td>
<td>751</td>
<td>13.8</td>
<td>185.1</td>
<td>191</td>
<td>5.9</td>
</tr>
<tr>
<td>Pansolar</td>
<td>225</td>
<td>738</td>
<td>14.2</td>
<td>185.9</td>
<td>192</td>
<td>6.1</td>
</tr>
<tr>
<td>CSL Virginia</td>
<td>294</td>
<td>965</td>
<td>20.6</td>
<td>184.7</td>
<td>207</td>
<td>22.3</td>
</tr>
<tr>
<td>Ever Reward</td>
<td>294</td>
<td>965</td>
<td>20.8</td>
<td>184.5</td>
<td>207</td>
<td>22.5</td>
</tr>
<tr>
<td>OOCL Halifax</td>
<td>294</td>
<td>965</td>
<td>20.8</td>
<td>186.6</td>
<td>207</td>
<td>20.4</td>
</tr>
<tr>
<td>Sun Road</td>
<td>294</td>
<td>965</td>
<td>21.4</td>
<td>184.2</td>
<td>208</td>
<td>23.8</td>
</tr>
<tr>
<td>MSC Nora</td>
<td>244</td>
<td>801</td>
<td>18</td>
<td>175.5</td>
<td>188</td>
<td>12.5</td>
</tr>
<tr>
<td>Heijin</td>
<td>180</td>
<td>591</td>
<td>16.5</td>
<td>178.1</td>
<td>189</td>
<td>10.9</td>
</tr>
<tr>
<td>Topeka</td>
<td>199</td>
<td>653</td>
<td>16.5</td>
<td>180.8</td>
<td>195</td>
<td>14.2</td>
</tr>
<tr>
<td>United Spirit</td>
<td>175</td>
<td>574</td>
<td>17.7</td>
<td>182.2</td>
<td>191</td>
<td>8.8</td>
</tr>
<tr>
<td>Saga Frontier</td>
<td>199</td>
<td>653</td>
<td>13</td>
<td>181.8</td>
<td>187</td>
<td>5.2</td>
</tr>
<tr>
<td>Star Grip</td>
<td>197</td>
<td>646</td>
<td>13</td>
<td>178.8</td>
<td>187</td>
<td>8.2</td>
</tr>
<tr>
<td>Tritonia</td>
<td>170</td>
<td>558</td>
<td>14.2</td>
<td>183.8</td>
<td>185</td>
<td>1.2</td>
</tr>
<tr>
<td>Hardanger</td>
<td>213</td>
<td>699</td>
<td>14.2</td>
<td>181.1</td>
<td>192</td>
<td>10.9</td>
</tr>
</tbody>
</table>
A.3 Uncertainty estimates for cumulative sound exposure levels

The uncertainty of the 24-hr cumulative sound exposure levels were based on the outputs of the sensitivity analysis. The greatest sources of error for the input variables were the source level of the ships and the sediment type. As a result, these variables were used to generate bootstrapped samples for each 15 minute section of each day for each seal. Section A.2 compares measurements of ship source levels (McKenna et al. 2012) and the estimated source levels using the RANDI ship source model (Breeding et al. 1996). The mean and standard deviation of the difference in the two values was used to generate a distribution (mean = 9.48, SD = 8.43) from which to draw error levels for each 15 minute cumulative sound exposure level. A distribution was also created for sediment type based on the execution of 5 runs where random sediment type was selected and compared to the baseline values in the sensitivity analysis (mean = -1.05, SD = 3.11). The predicted $cSEL_{15}$ were adjusted by drawing from the distributions for error in sediment and source level. A distribution of 1000 samples were generated for each 15 minute section. The distribution for each $cSEL_{24}$ based on these samples, the summary statistics and the original predicted $cSEL_{24}$ value are given below for each day (Fig. A.1 and A.2). The results suggest that the range in $cSEL_{24}$ values for each day can be greater than 20 dB. However, the interquartile-range of each distribution is between 3 and 7 dB. The results show the level of uncertainty which may be associated with the predictions of $cSEL_{24}$. However, when measured data becomes available, which may be relatively soon given the success of recent DTAG deployments (Mikkelsen et al. 2019), additional validation is still required. This would help account for ships that are missing from the AIS data and characterise sounds present in the soundscape that are not related to shipping. The uncertainty estimates could also be improved by a detailed understanding of the error distributions used as input for resampling. For example, as discussed in Section
A.2 the comparison between RANDI source levels and measured source levels are themselves uncertain.
Figure A.1: Uncertainty distributions for all days in the Celtic Sea generated using bootstrapped samples.
Figure A.2: Uncertainty distributions for all days in the English Channel generated using bootstrapped samples.
A.4 Model validation plots for shipping traffic model

Validation of the GAMM model predicting 15 minute cSEL developed in Chapter 3 was conducted by visually examining the normalised residuals of the model (Fig. A.3). There were no patterns indicating heterogeneity of the residuals in the fitted vs residuals plots (Fig. A.3a), and the residuals showed a normal distribution (Fig. A.3b). The sampling of datapoints was sufficient to ensure no autocorrelation in residuals (Fig. A.3c).

![Figure A.3: Model validation plots for GAMM model of 15 minute cSEL and explanatory variables closest point of approach, maximum source level and number of ships.](image)

(a) fitted vs residuals plot, (b) histogram of residuals to examine normality, (c) plot of autocorrelation between the residuals.
A.5 Results of sensitivity analysis

The following plots give the results of the sensitivity analysis for variables that resulted in variations of less than 2 dB, and therefore, were not included in the main text.
Figure A.4: Mean absolute deviation from baseline cumulative sound exposure levels (cSEL) for variations in model input parameters.
B Supplementary Material for Chapter 4

B.1 Model outputs for English Channel

The tables below give the model selection results for response variables not included in the main text. They indicate that noise was not a significant explanatory variable in the model of each response variable. Negative $\Delta$ AIC values indicate that the model fit was improved by removing the explanatory variable $\textit{noise}_{\text{cat}}$ from the model.

Maximum Depth

Table B.1: The structure of the maximal model and the final minimum adequate model for the response variable maximum depth.

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>$R^2$</th>
<th>AIC</th>
<th>$\Delta$ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Full $^1$</td>
<td>12</td>
<td>0.846</td>
<td>7115</td>
<td></td>
</tr>
<tr>
<td>Final $^2$</td>
<td>8</td>
<td>0.852</td>
<td>7112</td>
<td>-3</td>
</tr>
</tbody>
</table>

$^1 \textit{maxdepth} \sim s(\text{bathy}) + \textit{noise}_{\text{cat}} + \textit{dive}_{\text{c}} + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal})$

$^2 \textit{maxdepth} \sim s(\text{bathy}) + \textit{dive}_{\text{c}} + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal})$

Descent Rate

Table B.2: The structure of the maximal model and the final minimum adequate model for the response variable descent rate.

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>$R^2$</th>
<th>AIC</th>
<th>$\Delta$ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Full $^1$</td>
<td>16</td>
<td>0.439</td>
<td>-357</td>
<td></td>
</tr>
<tr>
<td>Final $^2$</td>
<td>10</td>
<td>0.442</td>
<td>-365</td>
<td>-8</td>
</tr>
</tbody>
</table>

$^1 \textit{descent}_{\text{rate}} \sim s(\text{bathy}) + \textit{noise}_{\text{cat}} + \textit{dive}_{\text{c}} + \textit{noise}_{\text{cat}} : \textit{dive}_{\text{c}} + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal}) + \text{varIdent}(1|\text{sex})$

$^2 \textit{descent}_{\text{rate}} \sim s(\text{bathy}) + \textit{dive}_{\text{c}} + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal}) + \text{varIdent}(1|\text{sex})$
### Dive Duration

**Table B.3:** The structure of the maximal model and the final minimum adequate model for the response variable dive duration.

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>$R^2$</th>
<th>AIC</th>
<th>$\Delta$ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Full</td>
<td>15</td>
<td>0.402</td>
<td>10262</td>
<td></td>
</tr>
<tr>
<td>Final</td>
<td>9</td>
<td>0.403</td>
<td>10257</td>
<td>-5</td>
</tr>
</tbody>
</table>

1 $dive_{dur} \sim s(\text{bathy}) + noise_{cat} + dive_c + noise_{cat} : dive_c + (1|\text{seal/event}) + cor\text{Spher}(1|\text{seal})$
2 $dive_{dur} \sim s(\text{bathy}) + dive_c + (1|\text{seal/event}) + cor\text{Spher}(1|\text{seal})$

### Bottom Time

**Table B.4:** The structure of the maximal model and the final minimum adequate model for the response variable bottom time.

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>$R^2$</th>
<th>AIC</th>
<th>$\Delta$ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Full</td>
<td>17</td>
<td>0.24</td>
<td>9930</td>
<td></td>
</tr>
<tr>
<td>Final</td>
<td>11</td>
<td>0.24</td>
<td>9922</td>
<td>-8</td>
</tr>
</tbody>
</table>

1 $\text{bottom time} \sim s(\text{bathy}) + noise_{cat} + dive_c + noise_{cat} : dive_c + (1|\text{seal/event}) + cor\text{Spher}(1|\text{seal}) + var\text{Ident}(1|\text{dive_c})$
2 $\text{bottom time} \sim s(\text{bathy}) + dive_c + (1|\text{seal/event}) + cor\text{Spher}(1|\text{seal}) + var\text{Ident}(1|\text{dive_c})$

### Inter-Dive Interval

**Table B.5:** The structure of the maximal model and the final minimum adequate model for the response variable inter-dive interval.

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>$R^2$</th>
<th>AIC</th>
<th>$\Delta$ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Full</td>
<td>15</td>
<td>0.24</td>
<td>8852</td>
<td></td>
</tr>
<tr>
<td>Final</td>
<td>9</td>
<td>0.24</td>
<td>8844</td>
<td>-8</td>
</tr>
</tbody>
</table>

1 $\text{IDI} \sim s(\text{bathy}) + noise_{cat} + dive_c + noise_{cat} : dive_c + (1|\text{seal/event}) + cor\text{Spher}(1|\text{seal}) + var\text{Ident}(1|\text{dive_c})$
2 $\text{IDI} \sim s(\text{bathy}) + dive_c + (1|\text{seal/event}) + cor\text{Spher}(1|\text{seal}) + var\text{Ident}(1|\text{dive_c})$
B.2 Random effects coefficients for GAMM models

Figure B.1 shows the random effect coefficients from the English Channel adult ascent rate and Celtic Sea pups decent rate models. It shows variation between seals and within seals between high ship noise events.

**Figure B.1:** Random effect coefficients by seal and event for (a) English Channel adults ascent rate, and (b) Celtic Sea pup descent rate. It shows variation in behaviour between seals and for each seal between different events. Seals reacted to some events by decreasing ascent rate/descent rate and others by increasing ascent rate/descent rate. Graph b is given in transformed log(y+1) scale of model.
B.3 Model outputs for Celtic Sea

Dive Duration

Table B.6: The structure of the maximal model and the final minimum adequate model for the response variable dive duration.

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>$R^2$</th>
<th>AIC</th>
<th>△ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Full $^1$</td>
<td>17</td>
<td>0.801</td>
<td>9784</td>
<td></td>
</tr>
<tr>
<td>Final $^2$</td>
<td>11</td>
<td>0.803</td>
<td>9779</td>
<td>-6</td>
</tr>
</tbody>
</table>

1 $\text{dive}_{\text{dur}} \sim s(\text{bathy}) + \text{noise}_{\text{cat}} + \text{dive}_{c} + \text{noise}_{\text{cat}} : \text{dive}_{c} + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal}) + \text{varIdent}(1|\text{dive}_{c})$

2 $\text{dive}_{\text{dur}} \sim s(\text{bathy}) + \text{dive}_{c} + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal}) + \text{varIdent}(1|\text{dive}_{c})$

Maximum Depth

The △ AIC value indicates that the explanatory variable $\text{noise}_{\text{cat}}$ was making a significant contribution to the model. However, this was not included in the results because there was significant heterogeneity in the model residuals that could not be dealt with sufficiently while still achieving model convergence.

Table B.7: The structure of the maximal model and the final minimum adequate model for the response variable maximum depth.

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>$R^2$</th>
<th>AIC</th>
<th>△ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Full $^1$</td>
<td>17</td>
<td>0.801</td>
<td>7082</td>
<td></td>
</tr>
<tr>
<td>Final $^2$</td>
<td>11</td>
<td>0.803</td>
<td>7104</td>
<td>22</td>
</tr>
</tbody>
</table>

1 $\text{max}_{\text{depth}} \sim s(\text{bathy}) + \text{noise}_{\text{cat}} + \text{dive}_{c} + \text{noise}_{\text{cat}} : \text{dive}_{c} + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal}) + \text{varIdent}(1|\text{dive}_{c})$

2 $\text{max}_{\text{depth}} \sim s(\text{bathy}) + \text{dive}_{c} + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal}) + \text{varIdent}(1|\text{dive}_{c})$

Ascent Rate

Table B.8: The structure of the maximal model and the final minimum adequate model for the response variable ascent rate.

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>$R^2$</th>
<th>AIC</th>
<th>△ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Full $^1$</td>
<td>17</td>
<td>0.230</td>
<td>-542</td>
<td></td>
</tr>
<tr>
<td>Final $^2$</td>
<td>11</td>
<td>0.218</td>
<td>-541</td>
<td>1</td>
</tr>
</tbody>
</table>

1 $\log(\text{ascent}_{\text{rate}} + 1) \sim s(\text{bathy}) + \text{noise}_{\text{cat}} + \text{dive}_{c} + \text{noise}_{\text{cat}} : \text{dive}_{c} + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal}) + \text{varIdent}(1|\text{dive}_{c})$

2 $\log(\text{ascent}_{\text{rate}} + 1) \sim s(\text{bathy}) + \text{dive}_{c} + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal}) + \text{varIdent}(1|\text{dive}_{c})$
Bottom Time

Table B.9: The structure of the maximal model and the final minimum adequate model for the response variable bottom time.

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>$R^2$</th>
<th>AIC</th>
<th>$\triangle$ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Full</td>
<td>17</td>
<td>0.402</td>
<td>1548</td>
<td></td>
</tr>
<tr>
<td>Final</td>
<td>11</td>
<td>0.399</td>
<td>1544</td>
<td>-4</td>
</tr>
</tbody>
</table>

1  $\log(\text{bottom time}) \sim s(\text{bathy}) + \text{noise}_{\text{cat}} + \text{dive}_c + \text{noise}_{\text{cat}} : \text{dive}_c + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal}) + \text{varIdent}(1|\text{dive}_c)$
2  $\log(\text{bottom time}) \sim s(\text{bathy}) + \text{dive}_c + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal}) + \text{varIdent}(1|\text{dive}_c)$

Inter-Dive Interval

Table B.10: The structure of the maximal model and the final minimum adequate model for the response variable inter-dive interval.

<table>
<thead>
<tr>
<th>Model</th>
<th>df</th>
<th>$R^2$</th>
<th>AIC</th>
<th>$\triangle$ AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Full</td>
<td>15</td>
<td>0.037</td>
<td>2040</td>
<td></td>
</tr>
<tr>
<td>Final</td>
<td>9</td>
<td>0.036</td>
<td>2033</td>
<td>-7</td>
</tr>
</tbody>
</table>

1  $\log(\text{IDI} + 1) \sim s(\text{bathy}) + \text{noise}_{\text{cat}} + \text{dive}_c + \text{noise}_{\text{cat}} : \text{dive}_c + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal})$
2  $\log(\text{IDI} + 1) \sim s(\text{bathy}) + \text{dive}_c + (1|\text{seal/event}) + \text{corSpher}(1|\text{seal})$
B.4 Model validation plots for behavioural analysis models

Validation of the GAMM models was completed using visual inspection of the model residuals. Figure B.2a and b show the residuals vs fitted values of the English Channel ascent rate and Celtic Sea descent rate models respectively. The addition of a variance structure improved the heterogeneity of the Celtic Sea model as far as possible while still allowing the model to converge (Fig. B.2b). There are some minor deviations from the normality of the residuals but they generally show acceptable levels of normality to meet the model assumptions (Zuur 2009). In both models the residuals show very little remaining autocorrelation (Fig. B.2e and f).

![Figure B.2](image)

Figure B.2: Residual vs fitted plots for each model (a,b). Q-Q plot looking at the normality of residuals (c,d). Autocorrelation plot of the residuals (e,f). Plot a, c and e refer to the English Channel adults ascent rate model. Plot b, d and f refer to the Celtic Sea descent rate model.
C  |  Supplementary Material for Chapter 5

The following supplementary material contains the results from propagation loss simulations for conditions not reported in the main text. It also includes the adaptive grids for conditions not shown in the text because they take a similar form to the examples used in the manuscript.
Figure C.1: Propagation loss at each corner and the centre of grid cells for the downslope condition for grid cell sizes of 1, 5, and 20 km in December and August at a receiver depth of 20 m. When the corner values come to within 1.5 dB of the centre values consistently, propagation loss is considered uniform (vertical line). As distance between the source and receiver increases the difference in propagation loss between each corner and centre decreases until uniform. As the grid sizes become larger the distance of uniform propagation loss becomes much greater. For the 20 km grid cell sizes (e,f) there is still a large difference between each corner and centre at 200 km and at no point is propagation loss considered uniform. It is also possible to note that in December maximum propagation loss is $\sim 110$ dB but in August this value is $\sim 140$ dB.
Figure C.2: Propagation loss at each corner and the centre of grid cells for the upslope and downslope condition for grid cell sizes of 1, 5, and 20 km in August at a receiver depth of 60 m. When the values come to within 1.5 dB of each other consistently, propagation loss is considered uniform (vertical line). As distance between the source and receiver increases the difference in propagation loss between each corner and centre decreases until uniform. As the grid sizes become larger the distance of uniform propagation loss becomes much greater. For the 20 km grid cell sizes (e,f) there is still a large difference between each corner and centre at 200 km and at no point is propagation loss considered uniform.
Figure C.3: Example adaptive grids for a 160 × 160 km area of the Celtic Sea for the downslope conditions in December and August for a receiver depth of 20 m. Each dot indicates the centre of a cell, the size of which is shown in the key above the grid. Red dot indicates the receiver and each color indicates a new grid size.
Figure C.4: Example adaptive grids for a 160 × 160 km area of the Celtic Sea for the upslope and downslope conditions in August for a receiver depth of 60 m. Each dot indicates the centre of a cell, the size of which is shown in the key above the grid. Red dot indicates the receiver and each color indicates a new grid size.
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