OSPAR COMMISSION

Overview of the impacts of anthropogenic underwater sound in the marine environment

Biodiversity Series

2009

OSPAR Convention

The Convention for the Protection of the Marine Environment of the North-East Atlantic (the "OSPAR Convention") was opened for signature at the Ministerial Meeting of the former Oslo and Paris Commissions in Paris on 22 September 1992. The Convention entered into force on 25 March 1998. It has been ratified by Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Luxembourg, Netherlands, Norway, Portugal, Sweden, Switzerland and the United Kingdom and approved by the European Community and Spain.

Convention OSPAR

La Convention pour la protection du milieu marin de l'Atlantique du Nord-Est, dite Convention OSPAR, a été ouverte à la signature à la réunion ministérielle des anciennes Commissions d'Oslo et de Paris, à Paris le 22 septembre 1992. La Convention est entrée en vigueur le 25 mars 1998.
La Convention a été ratifiée par l'Allemagne, la Belgique, le Danemark, la Finlande, la France, l'Irlande, l'Islande, le Luxembourg, la Norvège, les Pays-Bas, le Portugal, le Royaume-Uni de Grande Bretagne et d'Irlande du Nord, la Suède et la Suisse et approuvée par la Communauté européenne et l'Espagne.

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In memory of our colleague

Wolfgang DINTER

who passed away too early and while this report was being compiled. His deep commitment resulted in increased general perception of the problems that human impacts such as underwater noise pose to marine life.

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Module 1: Introduction and Terms of Reference

1.1 Introduction

Many marine organisms, including most marine mammals (whales, dolphins, porpoises and pinnipeds), many marine fish species, and even some invertebrates, use sound for a variety of purposes, for example in communication, to locate mates, to search for prey, to avoid predators and hazards, and for short- and long-range navigation (for reviews see for example Tyack & Clark 2000; Popper *et al*. 2001; Würsig & Richardson 2002; Popper *et al*. 2004). Depending on the intensity (sound pressure level) at the source, the pitch (frequency) and the distance between source and receiver, sound can potentially affect marine organisms in various ways. In this regard, it is also important to consider the cumulative effect when a sound source is long-lasting or repeated in time (Richardson *et al*. 1995).

Concerns on the potential adverse effects of anthropogenic noise on marine life have been raised from within the scientific community since the 1970s, and research on the topic expanded in the 1980s (*e.g.* Payne & Webb 1971; Richardson *et al*. 1985). During the last decade the topic has been investigated extensively by a number of scientific institutions, governmental agencies and intergovernmental bodies, with major reviews dealing with the effects of sound on fish and marine mammals (e.g. Richardson *et al.* 1995; Würsig & Richardson 2002; Popper *et al.* 2004; Hastings & Popper 2005; Hildebrand 2005; ICES 2005; NRC 2003, 2005; Wahlberg & Westerberg 2005; Thomsen *et al*. 2006; Madsen *et al*. 2006; Southall *et al.* 2007; Nowacek *et al*. 2007). These studies have documented both the presence and absence of physiological effects and behavioural responses of marine mammals, fish, and even invertebrates to various sound signals and have set the scene for discussions among scientists, stakeholders and policy makers on how to address potential impacts of underwater noise and how to develop meaningful mitigation measures within regulatory frameworks.

The issue has also been taken up by the OSPAR Convention for the Protection of the Marine Environment of the North-East Atlantic. The OSPAR Convention defines pollution in Article 1(d):

> *"Pollution" means the introduction by man, directly or indirectly, of substances or energy into the maritime area, which results, or is likely to result, in hazards to human health, harm to living resources and marine ecosystems, damage to amenities or interference with other legitimate uses of the sea.*

Sound is one form of energy and depending on frequency, geometry and the acoustic properties of water and sediment it may spread over long distances, sometimes crossing national boundaries. Above certain thresholds, sound can constitute pollution in the sense of Article 1(d).

Consequently, at the meeting of the OSPAR Commission in 2004, Contracting Parties recognised noise as a form of pollution. At the meeting of the OSPAR Biodiversity Committee (BDC) in 2005, Germany offered to present a draft document on a comprehensive overview of the underwater acoustic impact from all relevant activities emitting underwater sound.

The first draft of the document was posted on the OSPAR website in autumn 2006 for discussion at the meeting of the working group on the Environmental Impact of Human Activities (EIHA) in 2006. A second draft was posted on the OSPAR website in March 2007 and included annotated comments from Contracting Parties and observers. Following extensive discussions at BDC 2007 it was agreed (see the BDC 2007 Summary Record, paragraph 4.37 (a-d)) to create an intersessional correspondence group (ICGN), consisting of Norway, Spain, the Netherlands and the International Association for Oil and Gas Producers (OGP), under the co-lead of Germany, the EC, the UK and the National Resources Defense Council (NRDC). It was also agreed that the ICGN should finalise the preliminary overview and present it for discussion to EIHA 2007 in order to be published as a living background document to be revised periodically in the light of further developments and research available. It was further noted that the ICGN should develop proposals to prepare by 2008/2009 the draft assessment of the impact of underwater noise on the marine environment for the purpose of product BA-5 of the Joint Assessment Monitoring Programme (JAMP) and the preparation of the next Quality Status Report (QSR).

The ICGN made considerable progress but was unable to complete the task. Given the difficulties that the ICGN and working group were facing in resolving the issues and the time that this was taking, the UK suggested an alternative approach for the paper, based on a 'modular' structure as a way forward, as this was favoured by the various UK bodies who had provided comments on the previous drafts of the paper (Defra, MoD, BERR, JNCC and Cefas). EIHA 2007 agreed that this issue should be referred to the OSPAR Heads of Delegation (HoD) meeting to be held in London in November 2007, to agree how to proceed and that the UK would submit detailed proposals for a modular approach for consideration by HoD.

De nombreux organismes marins, notamment la plupart des mammifères marins (baleines, dauphins, marsouins, pinnipèdes), de nombreuses espèces halieutiques marines et même certains invertébrés, utilisent le son à diverses fins, par exemple pour communiquer, localiser des opportunités d'accouplement, chercher des proies, éviter des prédateurs et dangers, et se déplacer sur des distances courtes et longues (voir par exemple les revues de Tyack & Clark 2000; Popper et al. 2001; Würsig & Richardson 2002; Popper et al. 2004). Selon l'intensité (niveau de pression acoustique) à la source, la hauteur (fréquence) et la distance entre la source et le récepteur, le son peut potentiellement affecter les organismes marins de diverses manières. A cet égard, il est également important de considérer l'effet cumulatif d'une source acoustique qui dure ou se répète dans le temps (Richardson et al. 1995).

La communauté scientifique déclare ses inquiétudes, depuis les années 1970, quant aux effets préjudiciables potentiels du bruit anthropique sur la vie marine et la recherche dans ce domaine s'est développée dans les années 1980 (par exemple Payne & Webb 1971; Richardson et al. 1985). Au cours des dix dernières années un certain nombre d'institutions scientifiques, d'agences gouvernementales et d'organes intergouvernementaux ont étudié de manière extensive ce domaine, en produisant des revues importantes sur les effets des sons sur les mammifères marins (par exemple. Richardson et al. 1995; Würsig & Richardson 2002; Popper et al. 2004; Hastings & Popper 2005; Hildebrand 2005; CIEM 2005; NRC 2003, 2005; Wahlberg & Westerberg 2005; Thomsen et al. 2006; Madsen et al. 2006; Southall et al. 2007; Nowacek et al. 2007). Ces études documentent aussi bien la présence que l'absence d'effets physiologiques et réactions comportementales des mammifères marins, du poisson et même des invertébrés aux divers signaux acoustiques et ont préparé la voie pour des discussions parmi les scientifiques, les parties prenantes et les décideurs politiques sur la manière de *traiter les impacts potentiels des bruits sous-marins et de développer des mesures de mitigation significatives au sein des cadres de travail réglementaires.*

Cette question a été également reprise par la Convention OSPAR pour la protection du milieu marin de l'Atlantique du Nord-est. La Convention OSPAR définit la pollution dans l'Article 1(d):

> *On entend par "pollution" : l'introduction par l'homme, directement ou indirectement, de substances ou d'énergie dans la zone maritime, créant ou susceptibles de créer des risques pour la santé de l'homme, des dommages aux ressources biologiques et aux écosystèmes marins, des atteintes aux valeurs d'agrément ou des entraves aux autres utilisations légitimes de la mer.*

Le son est une forme d'énergie et selon la fréquence, la géométrie et les propriétés acoustiques de l'eau et des sédiments il peut se répandre sur de longues distances, traversant quelquefois les frontières nationales. Le son peut constituer une pollution dans le sens de l'article 1(d) s'il dépasse certains seuils.

Lors d'OSPAR 2004, les Parties contractantes sont donc convenues de reconnaître le bruit comme une forme de pollution. Lors de la réunion du Comité biodiversité d'OSPAR (BDC) en 2005, l'Allemagne a offert de présenter un projet de document sur une synthèse exhaustive de l'impact des bruits sous-marins provenant des diverses activités émettant des sons sous-marins.

L'avant projet de ce document a été publié sur le site internet OSPAR en automne 2006, pour que la réunion du Groupe de travail OSPAR sur l'impact environnemental des activités humaines (EIHA) en 2006 puisse en discuter. Un second projet a été publié sur le site internet OSPAR en mars 2007, il comporte les commentaires annotés des Parties contractantes et des observateurs. A la suite de discussions approfondies lors du BDC 2007 il a été convenu (voir le compte rendu du BDC 2007, paragraphe 4.37 (a-d)) de mettre en place un Groupe intersessionel par correspondance sur les bruits sous-marins (ICGN), se composant de la Norvège, de l'Espagne et de l'OGP et copiloté par l'Allemagne, la CE, le Royaume-Uni et le Conseil de défense des ressources naturelles (NRDC). Il a également été convenu que l'ICGN terminerait le récapitulatif préliminaire et le présenterait à l'EIHA 2007 afin qu'il soit publié en tant que document de fond dynamique qui sera révisé périodiquement à la lumière de nouveaux développements disponibles et des recherches effectuées. Il a de plus été noté que l'ICGN devra élaborer des propositions pour la préparation du projet d'évaluation de l'impact des bruits sous-marins sur le milieu marin, et ce en 2008/2009, aux fins du produit BA-5 du Programme conjoint d'évaluation et de surveillance continue (JAMP) et la préparation du prochain Bilan de santé (QSR).

L'ICGN a fait d'énormes progrès mais n'a pas été en mesure de terminer cette tâche. Etant donné les difficultés auxquelles l'ICG et le groupe de travail ont fait face pour résoudre la question et le temps pris, le Royaume-Uni a suggéré une approche alternative, fondée sur une structure 'modulaire' permettant de faire avancer ce document. Cette démarche est préférée par les divers organes du Royaume-Uni qui ont communiqué des commentaires sur les projets précédents du document (Defra, MoD, BERR, JNCC et Cefas). L'EIHA 2007 est convenu que cette question devrait être renvoyée à la réunion des chefs de délégation OSPAR (HOD) se tenant à Londres en novembre 2007, pour convenir de la procédure à suivre et que le Royaume-Uni communiquerait des propositions détaillées pour une approche modulaire au HOD pour qu'il les étudie.

1.2 Terms of Reference

During the meeting of Heads of Delegation (HoD) in 2007, the UK presented their proposal for a modular approach (see document HoD(2) 07/2/1-Add.3). Several Contracting Parties welcomed this initiative by the UK on the basis that the intention was to build on information already produced. It was also noted that as this is an area to be considered within the Marine Strategy Directive, the timeframe for finalising the document would be important.

In detail HoD agreed:

- to welcome the UK proposal and asked the UK to take over as lead country for ICGN on the understanding that work would build upon the existing text;
- to encourage quick progress, stressing that this was not a research project but a compilation of existing knowledge. To this end the UK would, in co-operation with the ICGN, assign teams leads for each module by the end of the year 2007;
- to instruct the Secretariat to fund an external consultant using the QSR Special Budget, for one or more areas (modules) where no lead country was forthcoming;
- to request that ICGN present an update to BDC 2008 and OSPAR 2008, and a draft background document to EIHA 2008.

1.3 Structure of the Paper

This paper presents the Overview of the Impacts of Anthropogenic Underwater Sound in the Marine Environment in the modular approach. Besides this introductory chapter, it is comprised of the following seven modules: 2) Background on underwater sound, 3) Background on impacts of sound on aquatic life, 4) Marine construction and industrial activities, 5) Shipping, 6) Sonar, 7) Seismic surveys, and 8) Other activities.

The second module provides a comprehensive background on underwater sound, dealing with issues that are closely related to the overview: a) the nature of sound and basic concepts, b) measurement of sounds, c) physical units, d) biological units, e) sound vs. noise, f) source level measurements, g) sound propagation and transmission loss, and h) background noise.

The third module provides a background on general aspects of the impact of sound on marine life and aims to set the scene for the following modules. It outlines the approach of Richardson *et al.* 1995 in describing zones of noise influences (masking, behavioural response, injury, death) and discusses levels on which an impact assessment can be performed. In this context, the recently developed PCAD model by the National Research Council NRC 2005 ('population consequences of acoustic disturbance') is introduced and discussed.

Each of the five latter modules deals with one activity or closely related activities and their documented acoustic impacts on marine life. They comprise a description of sound sources and a review of documented impacts. The impact review follows the outline put forward in module 3 by describing results looking at masking, behavioural responses, injury and death. Taxa looked at should be marine mammals (cetaceans and pinnipeds), marine fish, and other marine life (*e.g.* turtles, invertebrates). Whenever possible, the overview focuses on peer-reviewed papers and widely accessible reviews (e.g. ICES 2005 and COWRIE papers). This part shall be followed by a detailed description of mitigation measures that are suggested or already applied.

This document summarises what is known about the impacts of various noisy activities on marine life to date. It should be viewed as a living background document, aiding decision makers and scientists on the issue. Given the fast progress of the scientific field, it is open for revision and regular updates.

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Module 2: Background on Underwater Sound

2.1 Nature of sound and basic concepts

Sound is a mechanical disturbance that travels though an elastic medium (*e.g.* air, water or solids). Sound is created if particles in such a medium are displaced by an external force and start oscillating around their original position. These oscillating particles will also set neighbouring particles in motion as the original disturbance travels through the medium. Sound waves are therefore compressional (longitudinal) waves that propagate through the interior of the material as pressure fluctuations, such that some particles will temporarily be further away from each other (local rarefaction) or closer together (local compression). A characteristic of longitudinal waves is that the motion of the particles goes back and forth along the same direction in which the wave travels. The rate of change of these pressure fluctuations determines the frequency of the sound which is measured in hertz (Hz); defined as the number of complete vibration cycles per second. A vibration cycle encompasses the entire event of a positive pressure variation from baseline followed by a negative pressure variation and its return to baseline. A high frequency tone (f) is generally perceived as having a high pitch (e.g. treble voice) while one with a lower frequency has a lower pitch (e.g. bass voice). Humans can hear frequencies between 20 Hz to 20 kHz, but for marine mammals and other species the audible spectrum can extend much outside the human hearing range. Sounds outside the human hearing range are often referred to as infrasound (below 20 Hz) and ultrasound (above 20 kHz).

In an acoustic wave, the pressure between particles will be higher during a temporary compression and lower during a rarefaction. Figure 2.1 displays the pressure variation in an acoustic wave in space and time. The magnitude of these pressure differences defines the amplitude of a sound. Amplitude is commonly measured as the difference between equilibrium and maximum positive pressure (peak) or as the difference between maximum positive and maximum negative pressure (peak to peak) of the waveform (see Figure 2.1a).

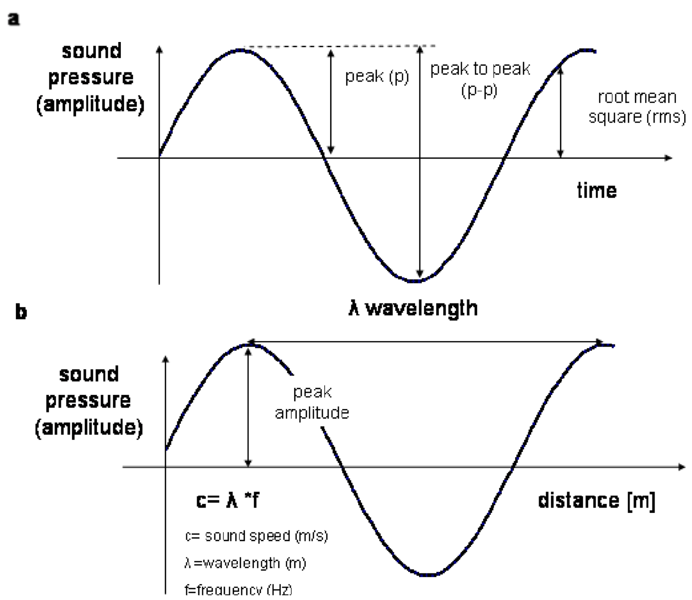

Figure 2.1: *Spatial pressure variation of an acoustic wave (of an acoustic wave at a given time (a) and time-behaviour of an acoustic wave at a given location (b, waveform). The graph also shows the most common amplitude measurements used to describe acoustic signals*

As previously mentioned, the frequency of a sound relates to the number of vibration cycles per second. The wavelength (λ) of sound is the distance a wave travels during one of these cycles (see Figure 2.1b). Wavelength and frequency are inversely related and depend upon the speed of sound in the respective medium (see Figure 2.1; $c = \lambda * f$). Different media vary greatly in their properties to conduct sound [*e.g.* the speed of sound in water (approximately 1500 ms^{-1}) is much greater than in air (approximately 330 ms^{-1})]. While the ears of mammals primarily sense pressure changes, the lateral line systems and ears of fish can also sense movement of particles directly. Particle motion refers to the vibrations of the molecules around an equilibrium state and can be quantified by measuring either velocity or acceleration of the particles. The relation between these particle velocities and the above mentioned pressure excursions depends on the medium properties (elasticity and density) and is described by the so-called acoustic impedance. For a medium like (sea) water this impedance is approximately 3560 times larger than for air.

2.2 Measurement of sound

Sound can be measured using sensors that are either sensitive to pressure changes or to particle motion. Hydrophones usually consist of crystals or ceramic elements that output a small electrical voltage when being deformed by local pressure changes (piezoelectric effect). A good hydrophone is linear and is equally sensitive at all frequencies of interest (flat-frequency response), meaning that it always produces the same voltage for a given sound pressure irrespective of frequency.

2.3 Physical units

2.3.1 Sound pressure level and the decibel

As mentioned above, sound can be described with respect to its amplitude, frequency and duration. The most common amplitude measurements are based on sound pressure given in Pascal (Pa), defined as force (in Newton, N) per unit area (m^2) However, due to the fact that the ears of many animals are able to detect sounds over a vast range of amplitudes, sound pressure is rarely directly reported. The problem of displaying the significant range of perceivable sound pressure values is that, in many marine mammals, the sound pressure capable of causing damage to the ears is seven orders of magnitude greater than the lowest sound pressure that is audible under quiet conditions. Therefore, to compress this dynamic range into a convenient range of values, a logarithmic scale is used. This scale is also useful to approximate human hearing, since the human ear judges perceived loudness on a logarithmic scale (Kinsler & Frey, 1982).

A common unit is the Sound Pressure Level (SPL), which can be calculated as:

$$SPL(dB) = 20 \times log10\left(\frac{Sound\ pressure}{Reference\ pressure}\right)$$

Sound pressure levels are reported in decibels (dB) with respect to a specific reference pressure value. The term decibel means “tenth of a bel” and was the unit first used to model human loudness perception. When reporting the SPL of a sound, it is important to not only state "dB" but to also add the reference pressure and state the analysis bandwidth (Verboom 1992). In water, this reference pressure is generally 1 micro-Pascal (µPa), while a reference pressure in air is typically 20 µPa. The air reference pressure (20 µPa) is derived from the average human hearing threshold (being the lowest sound pressure audible under quiet conditions) at 1 kHz. The difference in air and water reference pressures equates to a difference in SPL of approximately 26 dB. However, when comparing sound measurements in air and in water, the sound velocity and density of the medium (acoustic impedance) must also be considered. It is therefore common to add 62 dB (26 dB + 36 dB (the influence of acoustic impedance)) for the conversion from air to water. It is also important to note that to compare sound pressure values in air and underwater, biologically-meaningful units should be used (*e.g.* sound pressure levels could be referenced to the species' hearing threshold at a given frequency, similar to weighting procedures used to model human hearing).

Adding 6 dB to a given sound pressure level on the decibel scale reflects a doubling of physical sound pressure. However, this does not reflect a doubling in subjective perceived loudness in humans, which requires (as a rough rule of thumb) a 10 dB increment between two sound pressure values. Sound pressure levels are usually calculated from root mean square (rms) pressure of the waveform (Figure 2.1a). However, for certain sound types other amplitude parameters can be more appropriate (see Figure 2.1a). The peak-to-peak sound pressure is the difference between the maximum positive and maximum negative pressure of a waveform (Figure 2.1). Peak-to-peak SPLs are usually used to describe short, high intensity sounds where the rms-sound pressure value could underestimate the risk of acoustic trauma. The rms value is calculated as the square-root of the mean-squared pressure of the waveform. The unit is generally used to describe the mean variance of continuous waveforms (often loosely referred to as the "mean power") of continuous waveforms, and is given by the following formula:

$$Mean\ Power = \frac{1}{T}\int_0^T |x(t)|^2\, dt$$

where *T* is the duration of the signal, and *x(t)* are the sample values describing the signal.

2.3.2 Sound exposure level

The Sound Exposure Level (SEL) is a measure of the energy of a sound; therefore it depends on both amplitude and duration. The SEL is the time-integral of the instantaneous squared sound pressure normalized to a squared reference pressure over a 1-second period, thus in our case normalised to 1 µPa²s. Sound exposure levels can therefore be calculated as:

$$Sound\ exposure\ level = \sqrt{\int_0^T |x(t)|^2\, dt}$$

where *T* is the duration of the signal, and *x(t)* is the sample values describing the signal.

This formula can also be expressed as:

$$SEL = SPL(rms) + 10*log(duration)$$

where *SPL* is measured in dB re 1 µPa and *duration* is measured in seconds.

SELs are considered useful when making predictions about the physiological impact of noise: hearing damage can be modelled as a function of the acoustic energy of a stimulus, and the onset of temporary hearing damage depends on the SPL and exposure time (e.g. a subject can be safely exposed to a weaker stimulus for a longer time, but exposure to a loud stimulus would only be acceptable for a short time). However, this model is limited, particularly with respect to very short, loud pulses. In air, SELs are usually adjusted (weighted) for human hearing, but at present no such procedure is used in underwater applications.

2.3.3 Spectrum of a sound

Most sounds consist of many different frequencies, and spectral displays are a useful means of describing the “frequency content” of a sound. The spectrum of a sound is a representation that shows amplitude (y-axis) as a function of frequency (x-axis). A spectrum may, for instance, display the measured sound pressure power within different frequency intervals. The term bandwidth describes the frequency range of sound. It is important to emphasize that a normalised bandwidth of 1 Hz is standard practice in mathematical analysis of sound, whereas 1/3 octave bandwidths are most common in physical analysis. Therefore spectra need some indication of the analysis bandwidth as well.

The sine wave sound shown in Figure 2.1 can be considered as an exceptional case in that it constitutes a simple waveform that only consists of one frequency (a pure tone). In reality, sounds often contain integer multiples (harmonics) of a fundamental frequency. In a spectral display these would be shown as lines at different frequencies and one could read on the y-axis how much each harmonic contributes. Even more complex waveforms (*e.g.* noise) might contain a wide range of frequencies, and are referred to as broadband. It is important to know the spectrum of a sound in order to predict impact of noise on marine animals, as species differ greatly in their hearing sensitivity at different frequencies.

2.4 Biological units

It is important to acknowledge that what we perceive through our sensory organs does not necessarily reflect the reality of the physical environment. Although perceived loudness is proportional to the sound pressure level in dB (*SPL*), an SPL value in isolation would be meaningless if one wanted to predict how loud another animal has perceived a sound, or whether a sound is likely to elicit a response or inflict damage. In part, this has to do with the fact that the hearing systems of animals are not equally sensitive to all frequencies. Therefore, an important concept is that of the auditory threshold (also called hearing threshold) (Johnson 1967). The hearing threshold is the average sound pressure level that is just audible to a subject under quiet conditions. When the hearing threshold is plotted as a function of frequency it is called an audiogram (Figure 2.2). To give an example, the harbour porpoise’s hearing threshold at 500 Hz is about 90 dB re 1 µPa, while its hearing threshold at 50 kHz is in the order of 35 dB re 1 µPa (see Kastelein *et al.* 2002). This would mean that a sound with an SPL of 100 dB re 1 µPa and a frequency of 500 Hz would be barely audible to the porpoise, however, the same SPL at a frequency of 50 kHz would be perceived as relatively loud. Species also differ markedly in their audiograms with respect to the frequency range they can hear, and with respect to their absolute sensitivity. A unit that takes part of the mentioned characteristics into account is the sensation level. This is the sound pressure level in dB by which a stimulus exceeds the hearing threshold at a given frequency of interest (Yost 2000). Although this is a reasonable approximation, one should note that

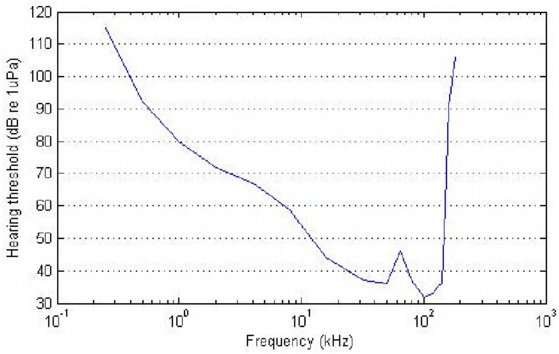

Figure 2.2*: Harbour porpoise audiogram; reproduced from (Kastelein, Bunskoek et al. 2002)*

perceived loudness is also influenced by other factors such as bandwidth and stimulus duration.

2.5 Sound versus noise

The term “noise” is colloquially used to refer to any unwanted sound. This is also reflected by the fact that criteria designed to protect humans or marine animals from anthropogenic sound sources are often referred to as “noise-exposure criteria”. In the latter sense, any unwanted sound with an amplitude exceeding a certain threshold, above which it is known to adversely affect behaviour or physiology of an organism, would be termed noise. However, noise also has a very specific acoustic meaning; it refers to a sound with a random waveform that contains energy across a broad range of frequencies. For instance “white noise” has energy which is equally distributed across all frequencies.

2.6 Source level measurements

The Source Level (SL) of an underwater source is defined as the level of sound at a nominal reference distance of 1m, expressed in dB re 1 µPa @1m. If a sound source was infinitesimally small (a so-called “ideal point-source”) then the source level could easily be measured by placing a hydrophone 1m from the source. However, real point sources do not exist, so in the case of a sound source that extends over a larger area (*e.g.* the noise from a ship), the measured sound pressure level in the immediate vicinity would be highly variable. The area around a sound source where such effects are present is called the acoustic “near-field”. The range (d) around a sound source within which near field effects occur can be approximated by the following formula (Richardson, 1995):

$$d = \frac{f \times a^2}{c}$$

where *c* is the sound speed, *f* is the frequency in Hz, and *a* is the longest dimension of the sound source.

Due to this effect, it is good practice to make source level measurements in the acoustic “far field” at sufficient distance from the source that the field has "settled down". Source levels can then be calculated back by a measured or modelled transmission loss.

2.7 Sound propagation and transmission loss

The process of sound travelling through a medium is termed sound propagation. Transmission loss (TL) refers to the loss of acoustic power with increasing distance from the sound source; an observer moving away from the source will therefore measure gradually decreasing sound pressure levels which are referred to as received levels (RL). The transmission loss at a given distance refers to the measured received level subtracted from the source level.

A major source of transmission loss is through geometrical spreading. Spreading losses occur because an emitted sound wave radiating spherically from the source contains only a limited amount of energy. As such, the energy of the sound wave will be distributed over a larger area at greater distances, and the acoustic power at a given location will be therefore be lower. However, in shallow water where sound can be reflected by the sea floor and/or water surface, transmission loss is far more complex. Transmission loss expected by spreading losses alone can be predicted as:

$$TL = 20 \times log10(d) \qquad \textit{(spherical spreading)}$$

where *d* is distance in metres.

A secondary energy loss is through absorption. This is the process by which sound energy is converted into heat and, as a result, the measured sound pressure level decreases with increasing distance from the sound source. An approximation for absorption losses (*a*) is (Richardson, 1995, p 73):

$$\alpha = 0.036 \times f^{1.5}$$

where *f* is the frequency in kHz.

Absorption losses are negligible for low frequencies (<1 kHz) but can be significant for high frequencies (*e.g.* a 40 kHz sound would have absorption losses up to approximately 10 dB per km).

In addition to absorption losses in the water, significant losses occur at the boundaries of the sea floor and the sea surface.

In conclusion, a simple model for obtaining predicted received levels at a given distance is to subtract spreading and absorption losses from the source level. However, sound propagation over large distances or in shallow water can be much more complex. For instance, temperature and salinity changes in the vertical profile of the ocean result in changes to sound speed which can lead to complex reflecting boundary layers that channel sound.

2.8 Background noise

Background noise is the combined noise of environment (ambient) and man-made (apparent) noise at a certain location that is usually present during a measurement. Ambient noise originates from a variety of different sources including wave and wind action, seismic activity, distant human activities and dispersed biological sources. In many oceans, wind-generated bubbles and spray are the primary source of ambient noise at frequencies between several hundred Hz and up to 30 kHz. In some tropical and subtropical oceans, biological sources like snapping shrimp or chorusing fish can contribute significantly to ambient noise even at high frequencies; up to values over 100 kHz. Ambient noise is biologically significant if it coincides with the frequency range of the communication or echolocation signals of marine animals. It can then potentially result in “masking”. This means that the distance over which marine animals are able to communicate (active space), navigate and detect prey or obstacles (in case of echolocating *odontocete*s) can be dramatically reduced.

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Module 3: Background on General Aspects of Impacts of Sound on Marine Life

3.1 The Importance of Sound for Marine Organisms

Sound speed in water exceeds that in air by a factor of about 4.5 and absorption is less compared to air. Not surprisingly, sound is an important sensory modality for many marine organisms, especially since other senses such as vision, touch, smell or taste are limited in range and / or speed of signal transmission. A variety of marine life - including marine mammals, fish and some invertebrates - have therefore developed special mechanisms both for emitting and detecting underwater sound (Urick 1983; Richardson *et al.* 1995; Popper *et al.* 2001, 2004; Berta *et al.* 2006; Karlsen 2007).

In marine mammals (cetaceans and pinnipeds), sound serves in communication, orientation, predator avoidance, and in foraging. It is also very likely that some cetaceans eavesdrop on sounds of their prey or conspecifics in order to obtain biologically relevant information (Riedmann 1990; Tyack 1998; Tyack & Clark 2000; Janik 2005, Gregg *et al.* 2008). Sounds range from the 10 Hz low-frequency calls of blue whales to the ultrasonic clicks of more than 200 kHz in certain offshore dolphins. Source levels of click sounds that are used by toothed whales in navigation and foraging can be as high as 235 dB re 1µPa peak-to-peak (sperm whale clicks Møhl *et al.* 2003). The communicative signals are much longer in duration, but source levels are lower (e.g. in bottlenose dolphins and in killer whales: 158 - 170 dB re 1µPa rms; Janik 2000; Miller 2006). The hearing of marine mammals spans an equally wide range of frequencies as the emitted sounds do (< 1 kHz - 180 kHz; overviews in Richardson *et al.* 1995; Perrin *et al.* 2002; Berta *et al.* 2006). However, it is not justified to define the animal's hearing range solely based on the frequency range of their acoustic emissions.

Many marine fish species produce sounds for communication, and it is suggested that they also use the surrounding acoustic environment for orientation (Montgomery *et al.* 2006). Most sounds are emitted at frequencies below 1 kHz, with source levels rarely exceeding 120 dB re 1 µPa rms. Fish exhibit a variety of hearing mechanisms, ranging from species that are probably only sensitive to particle motion, through 'generalists' that hear in the lower frequencies below 1 kHz, up to 'specialists', such as herring, that are able to perceive frequencies above 3 kHz (Fish & Mowbray 1970; Myrberg 1981; Zelick *et al.* 1999; Fay & Popper 1999 ; Popper 2003; Popper *et al.* 2003, 2004; Ladich 2004; Ladich & Fine 2006; Ladich & Myrberg 2006). However, so far only a few numbers of fish species (and sometimes only a few individuals) have been investigated. Consequently, our knowledge in this field is still very limited.

Other marine life that is sensitive to underwater sound include invertebrates such as decapod crustaceans which produce sounds and are sensitive to frequencies below 3 kHz, some cephalopods that are known to be sensitive to infrasound (< 20 Hz) with an upper hearing limit of 200 Hz, sea turtles with hearing capabilities in the lower frequency band, and perhaps aquatic birds (O' Hara & Wilcox 1990; Packard *et al.* 1990; Bartol *et al.* 1999; Popper *et al.* 2001; NRC 2003).

3.2 Sources of Sound in the Marine Environment

There are a variety of sound sources in the marine environment that occur naturally, such as vocalisations of marine mammals, fish and certain crustaceans (see above) and sounds that are induced by precipitation, wind, currents and waves. In addition there are acoustic events such as sub-sea volcanic eruptions, earthquakes and lightning strikes with a potential of affecting marine life (Bryden 2008). Some of them can reach quite high source levels of more than 200 dB re 1 μPa peak to peak (Møhl *et al.* 2003). Snapping shrimp influence ambient noise levels in tropical and subtropical waters to a high degree and might contribute to ambient noise levels in some areas of higher latitudes as well (Wenz 1962; Au *et al.* 1985, NRC 2003; Hildebrand 2005). Recent research indicates that these and other natural sounds can provide important orientation cues for marine fauna (*e.g.* Montgomery *et al.* 2006).

Man-made sound sources of primary concern in impact assessments are underwater explosions, ships, seismic exploration, offshore construction (*e.g.* for offshore wind farms or hydrocarbon production and transport facilities) and industrial activities, sonars of various types and acoustic devices designed to deter mammals from approaching (so called acoustic harassment or deterrent devices, AHDs, ADDs). Emitted frequencies range from low frequency engine noise < 100 Hz to very high frequency echo sounders of several hundred kHz. Source levels also vary widely and can reach more than 250 dB re 1 μPa peak-to-peak in the case of some offshore construction activities, seismic exploration and explosives (reviews by Richardson *et al.* 1995; Lepper *et al.* 2004, Nowacek *et al.* 2007; Thomsen *et al.* 2006).

3.3 Assessing Impacts of Underwater Sound on Marine Life

Impact assessments are generally concerned with those man-made activities that overlap in frequencies with the hearing range of marine organisms in question (reviews by Richardson *et al*. 1995; Nowacek *et al*. 2007). An exception has to be made for very loud sounds, because then the peak sound pressure is decisive and the frequency becomes less relevant. A sound is audible when the receiver is able to perceive it over background noise. The threshold of hearing that varies with frequency also determines audibility. The frequency dependant hearing sensitivity is expressed in the form of a hearing curve (audiogram), which in fish and marine mammals usually exhibits a U-shaped form. Determining if and how far an animal can hear a sound is the first important step in any impact assessment (Richardson *et al*. 1995; Popper 2003).

Table 3.1: *Overview of the acoustic properties of some anthropogenic sounds. * Nominal source, ** Higher source levels from drill ships use of bow thrusters, *** Projection based on literature data levels back-calculated at 1 m.*

Sound	Source level (dB re 1 µPa-m)*	Bandwidth (Hz)	Major amplitude (Hz)	Duration (ms)	Directionality	Source citations in module
Offshore construction						4
TNT (1-100 lbs)	272 - 287 Peak	2 - 1000	6 - 21	~ 1 - 10	Omnidirectional	4
Pile driving	228 Peak / 243 - 257 P-to-P	20 ->20 000	100-500	50	Omnidirectional	4
Offshore industrial activities						
Dredging	168 - 186 rms	30 - > 20 000	100 - 500	Continuous	Omnidirectional	4
Drilling	145 - 190 rms**	10 – 10 000	< 100	Continuous	Omnidirectional	4
Wind turbine	142 rms	16 - 20 000	30 - 200	Continuous	Omnidirectional	4
Shipping						
Small boats and ships	160 - 180 rms	20 - >10 00	> 1000	Continuous	Omnidirectional	5
Large vessels	180 - 190 rms	6 - > 30 000	> 200	Continuous	Omnidirectional	5
Sonar						
Military sonar low- frequency	215 Peak	100 - 500	-	600 - 1000	Horizontally focused	6
Military sonar mid-frequency	223 - 235 Peak	2800 - 8200	3,500	500 - 2000	Horizontally focused	6
Echosounders	235 Peak	Variable	Variable 1,500-36,000	5 - 10 ms	Vertically focused	6
Seismic surveys						
Airgun array	260 - 262 P-to-P	10 – 100 000	10 - 120	30 - 60	Vertically focused*	7
Other activities						
Acoustic deterrent / harassment devices	132 - 200 Peak	5000 -30 000	5000 – 30 000	Variable 15 – 500 ms	Omnidirectional	8
Tidal and wave energy devices***	165 - 175 rms***	10 - 50 000	-	Continuous	Omnidirectional	8

Scientific studies have documented both the presence and absence of behavioural responses of marine life to various sound signals. To date, no universal conclusion on the effect of sound can be drawn and is unlikely to emerge in the near future[1]. Certain conclusions about noise impacts are unlikely ever to be made, given the difficulty of observing marine mammals, fish and other marine life in the marine environment and the variability of responses. However, it is generally accepted that exposure to anthropogenic sound can induce a range of adverse effects on marine life, from insignificant impacts to significant behavioural changes to, in some cases, strandings and death of marine mammals. When evaluating the effects of underwater sound sources, peak pressure, received energy (received sound exposure level), signal duration, spectral type, frequency (range), kurtosis, duty cycle, directionality, and signal rise times are important. For species only sensitive to particle motion (some fish), this has to be considered as well. Possible effects can vary depending on a variety of internal and external factors, and can be broadly divided into *masking*, *behavioural disturbance*, and *hearing loss* (TTS and PTS, see below) / discomfort / injury. In extreme cases, and at very high received sound pressure levels, that are usually close to the source, very intense sounds might also lead to the *death* of the receiver (Richardson *et al.* 1995; Popper 2003; Popper *et al.* 2004; Madsen 2005; Nowacek *et al.* 2007; Southall *et al.* 2007).

Masking occurs when the noise is strong enough to impair detection of biologically relevant sound signals such as communication signals, echolocation clicks and passive detection cues that are used for navigation and finding prey. The zone of masking is defined by the range at which sound levels from the noise source are received above threshold within the 'critical band' centred on the signal (NRC 2003)[2]. It starts when the received sound level of the masking sound, for example noise from a nearby ship, equals the ambient noise in the frequency of the signal. Masking can shorten the range over which sounds can be detected and conspecifics are able to communicate for example mother and calf pairs of odontocetes (NRC 2003; Janik 2005; Madsen *et al.* 2006). However it should be noted that most animals use a range of frequencies to communicate and it is unlikely that the full range of frequencies would be masked over long time periods Nevertheless, important information can be lost even if perception is not masked over the full hearing range and the sound budget of some areas may be increased chronically, e.g. near shipping routes. If biological important functions such as foraging (see for example Aguilar de Soto *et al.* 2006) or finding mates are interrupted, masking can potentially have quite adverse effects. It also has to be considered that the communication networks - areas over which biologically important information is transferred - of some of the social groups of odontocetes (for example killer whales) are theoretically spanning several thousand square kilometres and that masking can reduce them considerably (discussion in Janik 2005).

Behavioural disturbances are changes in activity in response to a sound. These effects can be very difficult to measure and depend on a wide variety of factors such as the physical characteristics of the signal, the behavioural and motivational state of the receiver, its age, sex and social status and many others. Therefore, the extent of behavioural disturbance for any given signal can vary both within a population as well as within the same individual.

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[1] To put the difficulty of the assessment into context, a dose-response analysis of behavioural or physiological reactions has been difficult to gather even for humans in controlled experiments; while there is strong evidence of impacts of increased ambient noise on humans (Lercher *et al*. 2003, Stansfeld and Matheson, 2003).

[2] In biological hearing systems, noise is integrated over several frequency filters, called the critical bands. The width of the filter is app. 1/3 – 1/12 of an octave with Q (the bandwidth of the filter divided by the centre frequency) being relatively constant for most species investigated (see also Richardson *et al*. 1995).

Behavioural reactions can range from very subtle changes in behaviour to strong avoidance reactions. In some cetaceans, they can also be exhibited as changes in vocal activity (review by Richardson *et al.* 1995; Würsig & Richardson 2002; Miller *et al.* 2000).

Both TTS (=temporary threshold shift) and PTS (=permanent threshold shift) represent changes in the ability of an animal to hear, usually at a particular frequency, with the difference that TTS is recoverable after hours or days and PTS is not. As with masking, impairment through TTS or PTS of a marine animal's ability to hear can potentially have quite adverse effects on its ability to communicate, to hear predators and to engage in other important activities. Both TTS and PTS are triggered by the level and duration of the received signal (for extended reviews on the topics mentioned in this and the preceding paragraph see for example Richardson *et al.* 1995; Würsig & Richardson 2002; Gordon *et al.* 2004; Popper 2003; NRC 2003; Popper *et al.* 2004; Hastings & Popper 2005; Hildebrand 2005; Janik 2005; Madsen *et al.* 2006; Thomsen *et al.* 2006; Nowacek *et al.* 2007; Southall *et al.* 2007; see also Mooney *et al.* 2009).

Sound can potentially have a range of non-auditory effects such as damaging non-auditory tissues (swim-bladder, muscle tissue in fish; enhanced gas bubble growth in fish and marine mammals; traumatic brain injury/neurotrauma in fish and marine mammals etc; overviews in Richardson *et al.* 1995; Hastings & Popper 2005). Yet research on non-auditory effects of sound is still very much in its infancy.

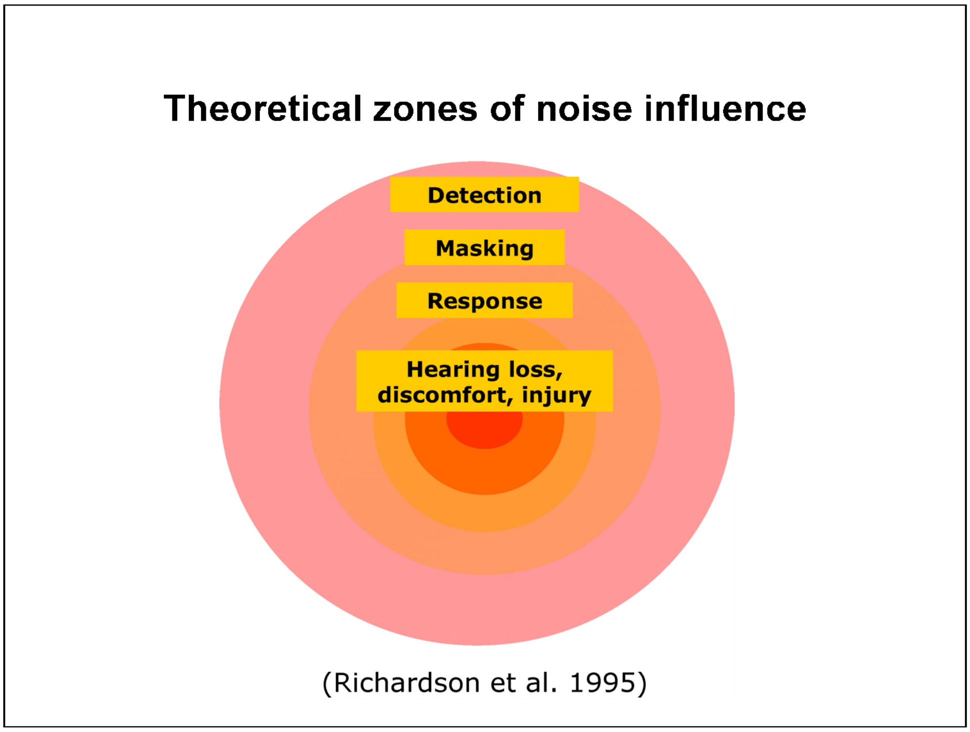

Figure 3.1: *Theoretical Zones of noise influence (after Richardson et al. 1995).*

Based on the different effects noise can have, Richardson *et al.* (1995) defined several theoretical zones of noise influence, depending on the distance between source and receiver. The zone of audibility is the largest and the one leading to the death of the receiver, the smallest. This model, shown in Figure 3.1, has been used very often in impact assessments where the zones of noise influences are determined based on noise propagation modelling or sound pressure level measurements on the one hand, and information on the hearing capabilities of the species in question on the other (see for recent examples Madsen *et al.* 2006; Thomsen *et al.* 2006). As sound spreads, in principle, omnidirectionally from the source, the zones of noise influences, given as distance from the source, indicate a radius rather than a straight line. It should be noted here that this model gives only a very rough estimate of the zones of influence as sound in the seas is always three-dimensional. The interference, reflection and refraction patterns within sound propagation will inevitably lead to much more complex sound fields than those based on the model by Richardson *et al.* (1995). This complexity may lead to effects such as increases of received sound energy with distance, especially when multiple sound sources are used simultaneously, (*i.e.* seismic surveys).

Stress and Indirect effects. Based on extrapolations from investigations in terrestrial mammals, Wright *et al.* (2007) speculate that underwater noise can act as a stressor in marine mammals with consequences to individual health and population viability. Yet this field is quite a new area and conclusions are still very preliminary. Underwater sound has also been linked to both physiological and behavioural effects on fish and some invertebrates which may serve as food sources for marine mammals. If these effects are extensive or chronic, they could have the potential to interfere with the successful feeding of marine mammals.

***Table 3.2**: Overview of observed effects of underwater noise on marine life (after Richardson et al. 1995; Wuersig and Richardson 2003; Hastings and Popper 2005; Wright et al. 2007).*

Impact	Type of Effect
physiological *non auditory*	**- damage to body tissue: *e.g.* massive internal haemorrhages with secondary lesions, ossicular fractures or dyslocation, leakage of cerebro-spinal liquid into the middle ear, rupture of lung tissue** - induction of gas embolism (Gas Embolic Syndrome, Decompression Sickness/DCS, 'the bends', Caisson syndrome) - induction of fat embolism
auditory ("Sound Induced Hearing Loss/SIHL")	- gross damage to the auditory system – *e.g.* resulting in: rupture of the oval or round window or rupture of the eardrum - vestibular trauma – *e.g.* resulting in: vertigo, dysfunction of co-ordination, and equilibrium - permanent hearing threshold shift (PTS) – *e.g.*, a permanent elevation of the level at which a sound can be detected - temporary hearing threshold shift (TTS) – *e.g.*, a temporary elevation of the level at which a sound can be detected
perceptual	- masking of communication with con-specifics - masking of other biologically important sounds
behavioural	- stranding and beaching - interruption of normal behaviour such as feeding, breeding, and nursing - behaviour modified (less effective/efficient) - adaptive shifting of vocalisation intensity and/or frequency - displacement from area (short or long term)

For impact assessments, determining or modelling the sound levels that the observed animal receives is crucial. Propagation of sound varies locally with water depths, sea bed characteristics and other factors (Urick 1983). It is therefore impossible to assess the validity of studies documenting responses without measuring or modelling received sound pressure levels. Hence, it is preferable to review only such studies that have documented received sound pressure levels in some form (see also Nowacek *et al.* 2007 for a detailed discussion on this topic).

Effect studies are usually concerned with reactions of individuals or groups to impacts. Yet, from a conservation perspective, it is further critical to assess whether anthropogenic sound has a significant effect on populations. This is also important in assessing the impacts of noise in relation / or addition to other stressors, either to assess cumulative impacts and / or to focus protection efforts. A number of anthropogenic activities exist which have proven to have a substantial effect on marine mammals. Among the most important is the by-catch of cetaceans in fisheries which cause a mortality of several hundreds of thousands of animals per year (Read *et al.* 2006). Other factors include the depletion of fish stocks thus reducing prey availability or shift of prey species and chemical pollution (reviews for marine mammals in Perrin *et al.* 2002). An obvious conclusion is that all factors impacting on populations are cumulative and must be assessed together by discussing the significance of effects. Yet so far there is no information available on the cumulative effects of the factors listed above. No agreed assessment framework for cumulative effects of diverse human activities exists.

For marine mammals, NRC (2005) developed a population consequence of acoustic disturbance model (PCAD model, Figure 3.2). The model involves different steps from sound source characteristics through behavioural change, life functions impacted, and effects on vital rates to population consequences.

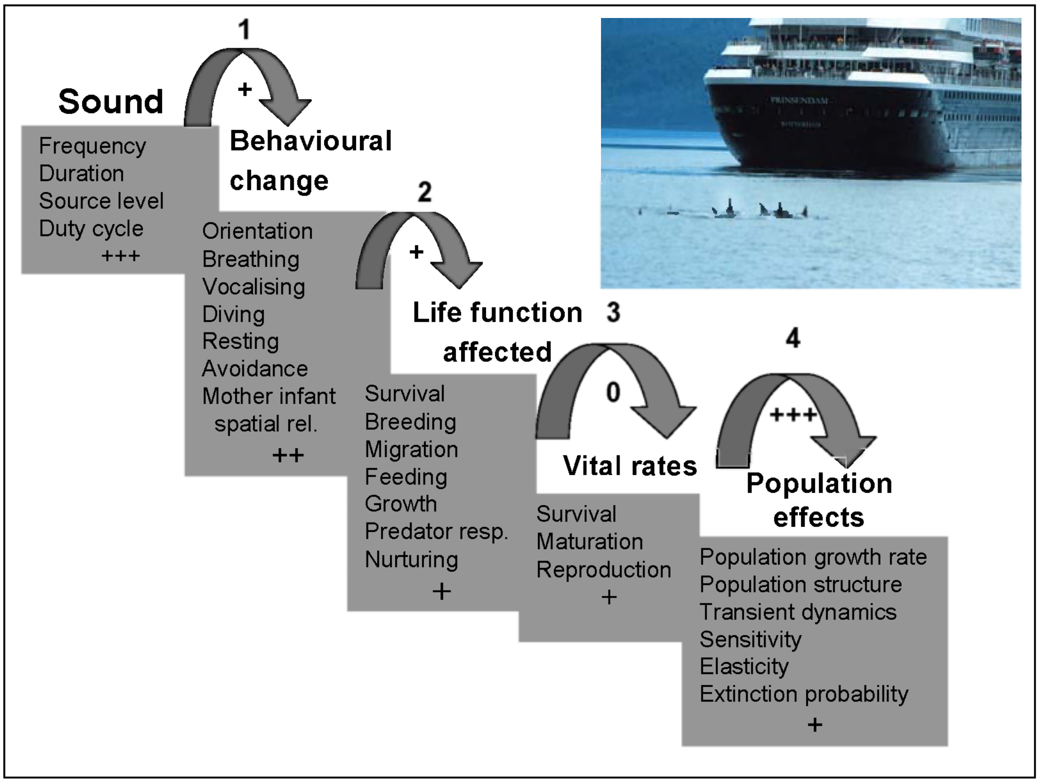

Figure 3.2*: Overview of the PCAD-model by NRC (2005).*

As can be seen in Figure 3.2, most of the variables of the PCAD model are currently unknown. Challenges to fill in gaps can come in many ways, due to uncertainties in population estimates for several species / regions, difficulties in weighting noise against other stressors (see above), difficulties in quantifying noise impacts etc. (see NRC 2005 for a detailed discussion).

Further difficulties arise from the inaccessibility of the marine environment (e.g. most carcasses of marine mammals will sink or be lost at sea, instead of stranding. This makes the assessment of lethal effects in marine life very challenging. The difficulties in estimating non-lethal effects are addressed above. Also, when discussing the impact of underwater sound on marine biota, one needs to differentiate between acute and chronic effects. Acute potential effects encompass the risks of immediate auditory damage or injury of the body from intense sound sources, while possible chronic effects entail the risks of habitat degradation or exclusion from preferred habitats for extended periods of time, even at moderate sound pressure levels. Both impacts may affect individuals, subpopulations or even populations at currently unpredictable levels.

3.4 Regulation and Mitigation

One way to regulate noisy activities is to set criteria for noise exposure that should not be exceeded. For example, based on earlier work, the U.S. NMFS (2003) used an exposure criterion of 180 dB re 1 μPa rms for cetaceans (both baleen and toothed whales) and 190 dB re 1 μPa rms for pinnipeds. Very recently, Southall *et al.* (2007) proposed sound exposure criteria for cetaceans and pinnipeds composed both of unweighted peak pressures and M-weighted sound exposure levels which are an expression for the total energy of a sound wave. It should be noted that these values are discussed critically within the scientific community as they are based on very limited data sets with respect to noise induced injury and behavioural response in marine mammals. There have been similar attempts to define exposure criteria for fish (see Popper *et al.* 2006; see also Woodbury and Stadler 2007), but none have yet been published in the peer reviewed literature.

Another way for regulation is to set safety zones within which no marine mammals should be present during sound intensive activities. For example, for marine mammals, the Joint Nature Conservation Committee (UK) recommends an exclusion zone of 500 m for the start of seismic surveys (JNCC 2004) while the Umweltbundesamt (Germany) recommends an exclusion zone of 750 m around a pile driving site where a certain sound pressure level should not be exceeded. However, it remains unclear whether or not safety zones are effective in protecting the animals from excessive sound exposure. For example, as already pointed out in section 3.3, it is not always guaranteed that sound pressure drops monotonically with increasing distance. Furthermore, exclusion zone validity is controversial if received levels are not measured in field during noise source operation in order to confirm or validate exposure levels. It should also be noted that masking and behavioural responses are possible beyond the safety zone.

The most effective mitigation measures are geographical and seasonal restrictions to avoid ensonification of sensitive species and habitats. Sound-producing activities may be designed to avoid areas and/or times where/when sensitive marine mammals and other species are usually engaged in susceptible activities such as mating, breeding, feeding, or migration.

Besides the measures listed above, there are several other measures to mitigate potential impacts of underwater noise, both dealing with the source of noise as well as the receiver. In this respect there is a difference between human activities producing noise as an unwanted side effect *e.g.* shipping and pile driving on one hand and activities deliberately producing sounds (*e.g.* seismic surveys) for specific goals on the other. The former can rather easily be made quieter by significantly reducing the most prominent noise sources without impairing their main mission objectives. The latter are potentially in a much more difficult situation when reducing their sound emission. Operational procedures can also be applied to reduce noise, for example 'soft-start / ramp-up' procedures can be undertaken during pile driving by slowly increasing the energy of the emitted sounds and thereby alerting marine life to the noise. Looking at the receiver, acoustic harassment devices have been used for both seals and harbour porpoises and have proven to be effective in scaring the animals away from the source at close ranges (Yurk & Trites 2000; Culik *et al.*, 2001; Cox *et al.* 2001). Yet, habituation is possible (Cox *et al.* 2001). Furthermore, since these devices deliberately disturb the receiver, their application needs to be discussed from a conservational viewpoint as well. Further details on mitigation measures for the various human activities generating noise will be given in modules 4 – 8.

3.5 Conclusions

Sound is important for many marine organisms, including marine mammals, fish, and perhaps some invertebrates. In many species, it is used actively for communication, navigation and foraging and perhaps also passively for orientation and eavesdropping. There are many sound sources in the ocean, some of them naturally occurring, others man-made. It must be acknowledged that our current knowledge on the impacts of underwater sound on marine life is incomplete, frequently inconclusive and occasionally contradictory. Nevertheless, it is clear that man-made underwater sound becomes a form of pollution when it harms or is likely to harm marine life.

The range of observed impacts to marine life from underwater sound is broad, from insignificant impacts to widespread behavioural disturbances to, in some cases, strandings and deaths of marine mammals. These impacts may affect a host of marine life, ranging from marine mammals, to fish, to invertebrates and perhaps sea turtles and possibly even aquatic (diving) birds. Even the disruption of commercially important fisheries, such as cod and haddock, has been shown (Engas *et al.* 1993). For the future discussion of the problem it seems to be useful to prioritise work by focusing on those high-energy sound sources which may be considered as having the highest potential for adverse impacts, and to distinguish here between the effects of (short-term) exposure to intense sound levels which might in their worst cases result in injury and death on the one hand and effects of exposure to more moderate but generally increasing continuous (background) sound which can possibly influence long-term habitat quality and therefore might cause stronger population effects.

From a conservation perspective, estimating the effects of noise disturbances on populations is critical, and there are first attempts to develop population consequences of acoustic disturbance models (PCAD) at least for marine mammals. These approaches, based on studies on disruption of individuals and investigation in population parameters such as vital rates, are very much in their infancy.

Noise exposure criteria are one way to regulate noisy activities. Another way is to set safety zones within which no marine mammals should be present during sound intensive activities. Besides these measures, there are some options for mitigating noise, dealing both with the source and the receiver.

3.6 References

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Module 4: Marine Construction and Industrial Activities

4.1 Noise Profiles of Marine Construction and Industrial Activities

Marine construction and industrial activities include pile driving, dredging, cable laying, drilling, the operation of offshore wind farms and hydrocarbon production facilities, and the use of explosives in construction and decommissioning. Stone dumping is another activity generating noise. However, there are no reports on sounds emitted by this activity.

Many studies on sound pressure levels and investigations into effects have been published in non-refereed sources [that are difficult to evaluate]. While these studies have been included in this review their conclusions should be treated with caution (for reviews, see Richardson *et al.* 1995; NRC 2003; Hastings & Popper 2005; Nowacek *et al.* 2007)

Pile-driving is undertaken in harbour works, bridge construction, oil and gas platform installations, and in the construction of offshore wind farm foundations. Most recent published work has concerned the last activity. Source levels vary (Table 4.1) depending on the diameter of the pile and the method of pile driving (impact or vibropiling). The frequency spectrum ranges from less than 20 Hz to more than 20 kHz with most energy around 100 - 200 Hz (for an extended overview see Nedwell *et al.* 2003A; Nedwell & Howell 2004; Madsen *et al.* 2006; Thomsen *et al.* 2006; see also Prins *et al.* 2008).

Table 4.1: *Overview of some studies measuring noise from impact pile driving (In most cases, peak level and rms measurements were averaged over time of the signals; for SEL averaging time = 1 s; for more details refer to the papers).*

Activity	**Diameter**	**Measurement**	**Reported exposure/pressure level**	**Reference**
Construction of aviation fuel receiving facility	Unknown	Rms pressure at various distances from source	> 170 dB re 1 µPa rms at 250 m	Würsig *et al.* (2000)
Offshore wind farm construction in Sweden	3 m	SEL at various distances	~ > 200 dB re 1 µPa^2 · s at 1m	McKenzie-Maxon (2000)
Oakland Bay Bridge Construction	2.4 m	Peak-to-peak and rms pressure at various distances from source	185 - 196 dB re 1 µPa rms at 100 m 197 - 207 dB re 1 µPa peak to peak	Caltrans (2001)
Offshore wind farm construction in German North Sea	1.5 m	Peak pressure and SEL at various distances	228 dB re 1 µPa zero to peak at 1 m	Thomsen *et al.* (2006)
Offshore wind farm construction at four different sites in the UK	4.0 - 4.7 m	Peak-to-peak pressure at various distances	243 - 257 dB re 1 µPa peak to peak at 1m	Nedwell *et al.* (*in press*)

Dredging, for example to extract geological resources such as sand and gravel, to maintain shipping lanes, and to route seafloor pipelines, emits continuous broadband sound during operations, mostly in the lower frequencies. In one investigation, estimated source levels ranged from 160 to 180 dB re 1 µPa at 1 m (maximum ~ 100 Hz). Bandwidth was between 20 Hz and 1 kHz (limited by the recording equipment; most energy was below 500 Hz; Richardson *et al.* 1995). In a recent study Defra (2003) measured sound spectrum levels emitted by an aggregate dredger at different distances and found most energy to be below 500 Hz.

Drilling is undertaken from a) natural or man-made islands, b) platforms, and c) drilling vessels (semi-submersibles and drillships). Underwater noise levels from natural or man-made islands are reported to be rather low (source level~ 145 dB re 1 µPa at 1 m or lower) with the main frequency content below 100 Hz (Richardson *et al.* 1995). In a recent investigation, Blackwell *et al.* (2004a) found broadband (10 Hz – 10 kHz) levels reaching a maximum of 124 dB re 1 µPa at 1 km, mainly at 700 Hz – 1.4 kHz. Drilling noise from rig-caissons used in the Beaufort Sea was approx. 150 dB re 1 µPa at 1 m at 30 - 40 Hz (Richardson *et al.* 1995). Data on underwater noise from drilling platforms is rather sparse: Richardson *et al.* (1995) indicated that conventional drilling platforms are not very noisy. This was confirmed by McCauley (1998), who reported only 115 - 117 dB re 1 µPa at 405 and 125 m respectively (main energy was in the 31 - 62 Hz 1/3 octaves; analysis in 1/3 octave band levels, range: 10 Hz - 10 kHz). Drilling from drillships produces the highest noise levels with a maximum broadband source level of about 190 dB re 1 µPa rms at 1 m (10 Hz - 10 kHz). Sound levels are comparably high as sound is transmitted very efficiently through the hull. The ships generally use their thrusters to remain in location, resulting in a mixture of propeller and drilling noise (Richardson *et al.* 1995; NRC 2003).

Operation of offshore wind farms. Operational source levels of offshore wind farms are lower compared to those during construction and depend on construction type, size, environmental conditions (i.e. depth, topography, sediment structure, hydrography, type of foundation), wind speed, and probably also the size of the wind farm. Noise during operation has been measured from single turbines (maximum power 2 MW) in Sweden and Denmark (review in Madsen *et al.* 2006). Thomsen *et al.* (2006) reported sounds during the operation of an offshore turbine (1.5 MW) in shallow (5-10 m) waters of Utgrunden, Sweden at moderate to strong wind speeds of 12 m/s. The 1/3-octave sound pressure levels ranged between 90 and 112 dB re 1 µPa Leq at 110 m with most energy at 50, 160 and 200 Hz[3]. Recent measurements by Nedwell *et al.* (in press) on four offshore wind farms (2 MW - 3 MW) off the UK confirmed rather low broadband received sound pressure levels (114 - 130 dB re 1 µPa) inside wind farm areas with a maximum difference in SPL to outside the wind farm of 8 dB re 1 µPa. In addition there will be noise from maintenance (shipping) and repair work.

Explosions are used in construction and occasionally in the removal of unwanted subsea structures. Underwater explosions are one of the strongest point sources of anthropogenic sound in the seas. Sound from explosives can also travel tremendous distances (Richardson *et al.* 1995). Underwater transmission of explosions is complex with an initial shock pulse followed by a succession of oscillating bubble pulses. Source levels vary with the type and amount of explosives used, the water depth at which the explosion occurs and can range from 272 to 287 dB re 1 µPa zero to peak at 1 m distance (1 - 100 lb. TNT). Frequencies are

[3] For continuous and sinusoidal sounds Leq = rms.

rather low (range 2 - ~ 1 kHz; main energy between 6 – 21 Hz; duration < 1 - 10 ms; Richardson *et al.* 1995; NRC 2003).

4.2 Effects on Marine Mammals

4.2.1 Responsiveness

Pile driving did not affect ringed seals (*Phoca hispida*) off Alaska, probably due to habituation to industrial noise in the study area (received sound pressure levels ~ 150 dB re 1 µPa rms (Blackwell *et al.* 2004b). Using satellite telemetry, Tougaard *et al.* (2003a) could show that harbour seals continued to transit Horns Reef (Danish North Sea) during pile-driving for an offshore wind farm. However, the resolution of positional information was too low for a detailed study on the effects of the construction phase on seals. At Nysted (Danish Baltic Sea), Edren *et al.* (2004) found a 10 – 60% decrease in the number of hauled out harbour seals on a sandbank 10 km away from the construction during days of ramming activity compared to days when no pile driving took place. This effect was of short duration.

Of particular relevance here also, are the results of the recent empirical studies by Tougaard *et al.* (2003a,b, 2005) and Carstensen *et al.* (2006) during the construction of the offshore wind farms at Horns Reef (North Sea) and Nysted (Baltic). At Horns Reef, acoustic activity of harbour porpoises (*Phocoena phocoena*) decreased shortly after each ramming event and went back to baseline conditions after 3 - 4 h (Tougaard *et al.* 2003b pp 53. This effect was not only observed in the direct vicinity of the construction site but also at monitoring stations approximately 15 km away indicating that porpoises either decreased their acoustic activity or left the area during ramming periods (Tougaard *et al.* 2003b). It was also found that densities of porpoises in the entire Reef area during ramming were significantly lower than during baseline. During ramming, porpoises exhibited more directional swimming patterns compared to observations obtained on days without construction where more non-directional swimming patterns were observed. This effect was found at distances of more than 11 km and perhaps also 15 km from the construction site (Tougaard *et al.* 2003a). It is important to mention that these distances represent the radius of observations rather than the zone of responsiveness, as no observations / acoustic logging happened at greater distances (Tougaard *et al.*2003b). These reaction distances might therefore be viewed as the minimum zone of responsiveness. Similar effects were found during the construction (combination of pile driving and vibropiling) of the Nysted offshore wind farm[4]. There was no return to baseline levels after construction (Carstensen *et al.* 2006; Tougaard *et al.* 2005). However, since absolute abundance of porpoises near Nysted was low from the start, this finding might be incidental and is difficult to attribute to the construction activity (Tougaard *et al.* 2005). Further, both studies should be interpreted with caution, as there was no documentation of received sound pressure levels.[5, 6]

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[4] It should be noted here that there are no visual observations from Nysted comparable to the ones from Horns Rev. The observed effect at Nysted is the reduced habitat use - while porpoises were detected again at Horns Rev already a couple of hours (Tougaard et al. 2003B)after the piling ended. In this context it is important to note, that it can't be determined whether the ensonified animals returned to the Horns Rev area or other porpoises entered the area after the end of the piling activity.

[5] Modelling of pile driving noise by Thomsen et al. (2006) and Madsen et al. (2006) indicated that the signals from pile driving in conditions typical for the North Sea and Baltic, where both wind farms are located, might be audible to porpoises over at least 80 km and perhaps over several hundreds of km underwater. Source levels at both installations were reportedly similar to the ones used in the modelling exercises, so the reported zones of responsiveness are not totally unrealistic.

Future investigations, modelling or measuring received sound pressure levels, should give a better understanding of the effects of pile driving noise on porpoises and other marine mammals and other aquatic life.

Richardson *et al.* (1995) provide an overview of investigations into behavioural responses of cetaceans to **dredging.** Bowhead whales (*Balaena mysticetus*) did not apparently respond to a suction dredge in one study, but individuals avoided these dredges when exposed to 122 - 131 dB re 1 µPa (or 21-30 dB above ambient noise) in another investigation (see also Richardson *et al.* 1990). Gray whales (*Eschrichtius robustus*) ceased to use a particular breeding lagoon after an increase in industrial activities, including shipping and dredging (Bryant *et al.* 1984). However, it is not clear if this was due to sound or the increased presence of ships; no studies were made of the increase in sound or of received sound pressure levels. There are, to our knowledge, no recent studies (post 1995) on the effects of dredging noise on marine mammals.

Moulton *et al.* (2003) found no effects of the development of an artificial island including construction and **drilling** noise on the abundance of ringed seals nearby. Received levels were ~ 120 dB re 1 µPa. Activities included pile driving during construction and the authors did not look at effects separately (see also Blackwell *et al.* 2004b for results on pile driving on the same population). Richardson *et al*'s (1995) account of studies relating to behavioural effects of drilling in toothed whales is very equivocal showing both avoidance and attraction. Also, in some circumstances it is not apparent if dredging or some other factor induced the observed behaviour. Finally, in most cases, no received sound pressure levels are being reported so the results are very difficult to assess. In an experimental study using four animals, and also documenting received sound pressure levels, Thomas *et al.* (1990) could not detect any short-term behavioural or physiological effects of drilling noise playbacks on captive beluga whales (*Delphinapteras leucas*). Bowhead whales did show avoidance in some cases at received levels around 115 dB re 1 µPa (~20 dB above ambient noise level; for a detailed discussion, see Richardson *et al.* (1995). Gray whales showed avoidance responses to drilling sounds at received levels of 120 dB re 1 µPa, with considerable variation between individuals (Malme *et al.* 1984).

Operation of offshore wind farms. Koschinski *et al.* (2003) reported behavioural responses in harbour porpoises and harbour seals to playbacks of simulated offshore turbine sounds at ranges of 200 – 300 m, indicated by theodolite tracking and recordings of acoustic activity (porpoises). However, they did not model or measure received sound pressure levels and Madsen *et al.* (2006) discussed other potential pitfalls of the study such as the introduction of artefacts by using a pre-recorded CD for playbacks. Looking at the sound pressure levels emitted from operating wind farms, reactions, if any, might only occur at relatively short ranges in any cetacean species (see Thomsen *et al.* 2006). In line with that, Lucke *et al.*

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[6] Another note of caution applies to use of pingers and seal-scarers before ramming to deter porpoises and seals from the vicinity of the construction sites (Tougaard et al.2003b, 2005). In particular, the seal scarers might have caused avoidance response in porpoises at some distances, since the source levels used were reportedly rather high with carrier frequencies well within the hearing range of porpoises (SL = approx. 189 dB peak to peak re 1 µPa broadband; carrier frequencies of 13 – 15 kHz; Lofitech, Norway, pers. comm.). Since harbour porpoises have very acute hearing in that frequency range, it cannot be ruled out that effects were caused by a combination of the mitigation measures employed, along with the pile-driving. On the other hand, decrease of acoustic activity was also found during pile-driving in a harbour close to the Nysted site where no mitigation measures were employed.

(2007) found that simulated offshore wind turbine noise (1.5 MW) was only able to mask the detection of low frequencies up to 2 kHz by a harbour porpoise. The received level necessary for masking was 128 dB re 1 µPa. This would result in a masking zone of 20 m around smaller turbines (Lucke *et al.* 2007). However, it is important to remember that these conclusions are only valid for relatively small turbines. It is likely that bigger turbines – for example the 4-5 MW ones planned for most offshore wind farms – will be noisier with the sound most likely shifted to lower frequencies[7]. Another factor that has to be considered is the tonal content of the noise emitted by turbines in operation (Dewi 2004; Wahlberg & Westerberg 2005; Madsen *et al.* 2006). In larger turbines, narrow tones with clearly defined peaks might considerably exceed background noise levels, and the zone of audibility of these rather discrete frequencies, might be much larger than for relatively broadband noise (Dewi 2004).

Explosions. There is very little published data on the behavioural reaction of marine mammals to explosions caused by offshore construction or demolition activities. Some results from other studies are cited here to give an impression of possible reactions: 'Seal bombs' often used to prevent seals feeding around fishing operations caused startle and flight responses in some pinnipeds with habituation to repeated exposure. They have also been used to scare dolphins away from fishing operations (Richardson *et al.* 1995). Madsen and Møhl (2000) found no acoustic reactions of five sperm whales to distant detonators at received sound pressure levels of 180 dB re 1 µPa rms, possibly because the detonator noise resembled sperm whale clicks and might have been therefore perceived as signals from conspecifics. Behavioural responses at the surface were lacking in one observed individual (Madsen and Møhl 2000). Todd *et al.* (1996) did not find any changes in behaviour of humpback whales to blasts during the development of an offshore oil platform (received SPL = 140-153 dB re 1 µPa rms at 1.8 km). The same authors indicated that the higher number of entanglements of the whales in nets recorded in the area at the time may have been influenced by the long-term effects of exposure to deleterious levels of sound. However, cause-effect relationships could not be established and controls were not provided and therefore the results of the study are difficult to interpret (for a critical discussion of the study see Nowacek *et al.* 2007) Finneran *et al.* (2000) exposed two trained dolphins and one beluga to sounds resembling distant blast explosions. They observed that disruptions of the animals' trained behaviours began to occur at exposures corresponding to 5 kg at 9.3 km and 5 kg at 1.5 km for the dolphins (sound flux density of 153 and 169 dB re 1 µPa2s, respectively) and 500 kg at 1.9 km for the beluga whale (sound flux density of 177 dB re 1 µPa2s).

4.2.2 Hearing loss and injury

Given the comparatively low source levels, injuries from either **dredging** or **drilling** operations are unlikely in marine mammals, except very close to the source (see Southall et al (2007) for suggested noise exposure criteria). There is no documented case of injury caused by **pile driving** in the wild, yet this should be interpreted with caution since studies

[7] For example, Dewi (2004) simulated sound emissions of a 2.5 MW turbine based on their measurements of a 1.5 MW offshore windmill in operation. They estimated that the sound pressure levels of the simulated 2.5 MW turbine would be between < 10 to 30 dB higher compared to the 1.5 turbine, depending on frequency. Nedwell et al's (*in press*) recent results on comparably low operational noise levels from wind farms up to 3 MW do not necessarily contradict these simulations as their ambient noise levels were relatively high and measurements were undertaken from a whole wind farm and not a single turbine. Further evaluation, especially *in situ* measurements from larger turbines is needed to assess the impact from bigger turbines.

are very limited and observation of injury are almost impossible to obtain under natural conditions. Temporary threshold shift (TTS) caused by signals resembling **explosives** has been investigated in captive bottlenose dolphins and beluga whales. No TTS was observed to levels up to 221 dB re 1 µPa peak to peak (Finneran *et al.* 2000). Again, as this is the only study looking at TTS induced by sounds resembling construction noises, general conclusions are limited. Richardson *et al.* (1995) reports some rather poorly documented cases of injury and death of marine mammals thought to have been caused by explosions. Ketten (1993) reports injury in the ears of two humpback whales stranded after underwater explosions.

4.3 Effects on Fish

There is only a very limited number of investigations on the effects of marine construction sounds on fish. To our knowledge, these have been restricted to investigations on the effects of **pile driving** and **explosions**.

Hastings & Popper (2005) provide a detailed overview of results from five recent experimental studies looking at the effects of **pile driving** on fish. Four of them took place off the US west coast and one was undertaken in the UK. Species investigated included the shiner surfperch (*Cymatogaster aggregate*), Sacramento blackfish (*Orthodon microlepidotus*), brown trout (*Salmo trutta*), steelhead (*Oncorhynchus mykiss*), Chinook salmon (*Oncorhynchus tshawytscha*) and northern anchovy (*Engraulis mordax*). **Behavioural observations** were undertaken in one of the studies on caged fish held at different distances from piling (methods reviewed in Hastings and Popper 2005). However, as mentioned by Hastings & Popper (2005), experimental conditions were in most cases difficult to control and conclusions drawn from some of the studies might be viewed as being rather limited.[8] In the course of the San Francisco-Oakland Bay Bridge Project, a variety of external and internal **injuries,** including reddening of the liver, rupture of the swim bladder, or internal bleedings were observed in one of the investigations (Caltrans 2001). In another study, no physical injuries were observed in caged sea trout within a radius of 400 m from the piling (estimated SL = 194 dB re 1 µPa peak to peak) in the harbour of Southampton during construction of the Red Funnel Terminal (Nedwell *et al.* 2003b). There is also evidence from reports in the grey literature that pile driving can kill several different species of fish if they are sufficiently close to the source (review by Hastings & Popper 2005). For example, mortalities were observed after pile driving in the course of the San Francisco-Oakland Bay Bridge Demonstration Project, USA. Sound levels at a distance of 100 and 200 m from the pile were between 160 and 196 dB re 1 µPa rms (Caltrans 2001). Fish were found dead primarily within a range of 50 m (n=13). The external and internal injuries, which were observed, gave reason to assume that there might have been further mortalities, especially of species with swim bladders. The zone of direct mortality was about 10 - 12 m from piling, the zone of delayed mortality was assumed to extend out at least to 150 m to approximately 1000 m from piling. Tests on caged fish

[8] Nedwell et al. (2003b) placed farmed brown trout (Salmo trutta) in cages positioned at different distances from vibro and impact pile-driving operations in Southampton harbour and filmed them using close-circuit television monitoring. 'Startle-reactions' and 'Fish activity level' were investigated prior and during pile driving (but not after the event), with activity levels measured by counting the number of times a fish entered the camera's field of view within a two-minute observation period. The observations revealed no evidence that trout reacted to impact piling at 400 m (average received sound pressure level = 134 dB re 1µPa), nor to vibration piling at close ranges (<50 m; average received sound pressure level not given). However, received sound pressure levels were relatively low and it should also be noted that using farmed fish is likely to result in higher reaction thresholds to noise disturbances (Gill pers. communication; for a detailed critical review of the study see Hastings & Popper, 2005). ,

revealed greater effects when using a larger hammer (1700 kJ, as compared with 500 kJ). The greatest effects were observed in a range of 30 m from piling. Preliminary results indicated increasing damage rates to the fish together with extended exposure times (Caltrans 2001). However, reviewing these and other studies, Hastings and Popper (2005) point out that the results provided are yet highly equivocal[9]. To summarise, some reports in the grey literature indicate that severe damage due to pile driving noise is possible. However, in light of the possible pitfalls of some of the analyses, it is clear that more research is required to investigate the extent and scale of physical effects on marine fish due to pile driving. There have been attempts to define exposure criteria for fish (see Popper *et al.* 2006; see also Woodbury and Stadler 2007), but none have yet been published in the peer reviewed literature.

Mueller-Blenkle *et al.* (2008) played back tones (130-140 dB re 1 µPa) to cod (*Gadus morhua*) through an underwater speaker and found avoidance reactions to the playback stimuli. However, no conclusion on habituation could be drawn.

It is well established that underwater **explosions** can kill fish (*e.g.* Aplin 1947), in fact so called 'blast-fisheries' has been used in many areas of the world and on a considerable number of fish species (review by Saila *et al.* 1993). Blasts occurring during the decommissioning of oil platforms are also able to kill fish (Gitschlag & Herczeg 1994). Reviewing a variety of studies looking at the effects of explosive blasts on fish, Hastings & Popper (2005) conclude that mortality has been proven in some cases, but that there is considerable variability in the effects of explosive blasts, with variables including received sound energy, presence or absence of swim bladders, mass of fish and perhaps body shape. For general issues with non auditory effects on fish, please refer to module 7.

Model estimates of the effects of pile driving (through enhanced mortality of fish larvae) predict a decrease in the number of plaice, sole and herring larvae reaching the Dutch coast and the Wadden Sea. Using the most likely pile driving scenario, these predictions indicate transport success at 3 - 10% in the years when highest numbers of larvae reached these areas. The model predicts that a reduction in larval supply of this magnitude during many years is expected to be of ecological significance, despite the large inter-annual variability in larval supply (Prins *et al* 2008). According to this particular model, physical injury due to a series of strokes is likely to occur within a range of approximately 14 km from a monopole driving location (based on cumulative SEL for driving one monopole NOAA/NMFS dose-response levels, and assuming hearing does not recover (Prins *et al* 2008)

4.4 Effects on other species

The use of underwater explosives in structure-removals can injure and even kill sea turtles (Klima *et al.*, 1988; Gitschlag & Herczeg 1994 (see section 4.2.1). There are, to our knowledge no investigations on the effects of marine construction and industrial activities on marine invertebrates and diving birds.

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[9] Hastings and Popper (2005) note, for example, that no clear correlation between the level of sound exposure and the degree of damage could be determined. They also criticise that in most of the studies, 'pathology was done on fish that did not receive appropriate pathological or histological preparation or analysis'. Finally, in some studies it could not be ensured that exposed and control animals were treated identically (for a more detailed review on the pathological methods used see Hastings and Popper 2005; pp 40).

4.5 Mitigation

In the OSPAR region, most offshore construction activities are licensed and regulated on the basis of international law (e.g. United Nations Convention on the Law of the Sea) and nationally through various instruments (e.g. by DEFRA and MFA through FEPA in the UK; by BSH through the 'Seeanlagenverordnung' in Germany). An overview of the legislation that is in place in the OSPAR member states is provided in various OSPAR JAMP assessments. (see www.ospar.org).

Furthermore, in European waters, for most offshore construction activities, an environmental impact assessment must be carried out. The EIA Directive on Environmental Impact Assessment of the effects of projects on the environment (Directive 85/377/EEC 1985; amended 1997/2003) sets out rules on what information an EIA must provide to comply. For example, German 'Standards for the investigation of the impacts of offshore wind turbines on the marine environment' (StUK 3; Bsh 2007), and the UK guidance notes for EIAs for offshore wind farms (Defra 2004).

There are several options currently available to mitigate the impacts of **pile driving** at source (see also Nehls *et al.* 2007):

- extending the duration of the impact during pile-driving (decrease of 10-15 dB in SL; mostly at higher frequencies > 2 kHz),
- enclosing the ramming pile with acoustically-isolated material (plastic etc.; decrease of 5 – 25 dB in SL; higher frequencies better than lower ones). Whether this will have a reduction in the far field too is unknown at this stage.
- installing an air-bubble curtain around the pile (decrease of up to 20 dB, depending on frequency; Würsig *et al* .2000), and
- applying a soft-start / ramp-up procedure (slowly increasing the energy of the emitted sound; Richardson *et al.* 1995).

The methods mentioned above have benefits and costs. For example, extending the duration of the impact reduces the source level and has biological implications since signals of longer duration would mask communication signals to a greater extent than shorter signals. The method is also limited technically, since shorter pulses are more effective in driving the pile into the bottom than longer ones. Mantling seems to be very promising but has so far only been tested in a relatively short pile. Air bubble curtains are very expensive and might only be effective in relatively shallow water. Soft-start procedures are theoretically promising but their effect has not been tested to a large degree. Ramping-up might also make it more difficult for cetaceans and seals localising the sound source (Richardson *et al.* 1995). Further limitations and implications are discussed in detail by Thomsen *et al.* (2006) and Nehls *et al.* (2007). Mitigation could also occur by using acoustic harassment devices[10]. These have been used for seals and harbour porpoises and are effective in scaring the animals away from the source (Yurk *et al.* 2000; Culik *et al.* 2001). Culik *et al.* (2001) reported a mean avoidance zone of 500 m around a ‘pinger’ for porpoises. Cox *et al.* (2001) reported a smaller avoidance response of approximately 208 m. Therefore both systems seem to work at relatively short ranges, below the potential TTS zones (see above). It might therefore be necessary to deploy several pingers at different distances from the construction site. Precautionary mitigation

[10] Pingers and seal scarers are dealt with in more detail in module 8.

measures would include not carrying out pile-driving in confined areas in close proximity to migrating fish and turtles, feeding or breeding peaks of marine mammals could be avoided.

Alternative methods: Hydraulic pile driving may provide an alternative method *e.g.* for the installation of foundations of offshore wind turbines by impulsive pile driving. This method results in noise emissions at low levels, i.e. close to the background noise level at sea (well below 100 dB re 1µPa). It was developed for pile driving on land or in near shore areas and can easily be adapted for marine applications (*e.g.* GIKEN Europe).

In more general terms, delaying the start of or ceasing piling if turtles or marine mammals are detected (visually or acoustically) close to the source may also be effective in mitigating close-range effects such as PTS or TTS (see also Evans & Hammond 2004).

4.6 Conclusions

Marine construction and industrial activities include pile driving, dredging, drilling, the operation of offshore wind farms, and the use of explosives (construction and decommissioning). Frequencies emitted vary, yet most energy is emitted below 1 kHz. Source sound pressure levels vary widely with drilling operations being relatively low and pile driving and the use of explosives comprising very high source levels. Documented effects on marine mammals are variable and include a range of behavioural reactions and indications of physiological effects (see section 4.2.2) but relevant studies often do not describe the noise, even at relatively high received sound pressure levels. Investigations on injuries to marine life due to marine construction are too limited to draw any conclusions yet. The results are therefore very difficult to fully assess. Investigations on the effects on fish are even patchier and few generalisations can be drawn from the studies undertaken to date. Injuries and death due to pile driving and explosions are documented in a variety of species. Behavioural effects in fish due to construction activity have not been investigated fully. With regards to other species, underwater blasts can kill sea turtles. Most construction activities are regulated within the OSPAR region, often through the environmental impact assessment process. A variety of measures have been used to mitigate the effects of construction, including alterations of the equipment used, acoustic harassment devices, timing restrictions and delayed start/shutdown in the presence of marine mammals.

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Module 5: Shipping

5.1 Noise profiles of marine vessels

Among the many human-induced sources of low frequency sound in the marine environment, marine vessels (and particularly large commercial ships) represent numerous, widespread, and relatively loud individual sources of underwater noise. The exact characteristics of these depend on the ship type, size, mode of propulsion, operational characteristics, speed, and other factors. Much of the incidental noise results from propeller cavitation, though onboard machinery and turbulence around the hull can also result in underwater noise being transmitted underwater via direct or secondary paths. There is considerable variability in the radiated sound fields (in terms of source level or “amplitude”, frequency bandwidth or “pitch”, directionality[11], and other features) from individual vessels of various sizes and types (see below). Various vessel elements produce different frequencies, generally lower in frequency with increasing size. Low frequency sounds generally travel farther due to the physical properties of sound in water; the predominant sound frequencies associated with large vessels are below several hundred Hz, although higher frequencies are present relatively nearby with spectrum levels decreasing with increasing frequency. Consequently, the (predominant) low-frequency noise from many vessels has the potential to affect the “ambient” (background) noise over a much greater area than higher frequencies that may exist near individual sources.

5.1.1 Small leisure craft and boats:

(length up to 50m; *e.g.*, recreational craft, jet skis, speed boats, operational work boats, hover craft)

Small craft and boats produce relatively broadband[12] acoustic signatures with free-field source levels approximately 160-175 dB (re: 1µPa), although the output characteristics are highly dependant on speed and other operational characteristics (see Richardson *et al.*, 1995; Erbe, 2002a; Amoser *et al.*, 2004; Kipple and Gabriel, 2004; Blackwell and Greene, 2005; Miksis-Olds *et al.*, 2007). While there is significant variability in radiated sound fields among the diversity of craft (arbitrarily categorized here as “small”), many of these sources have greater sound energy in higher frequency bands (*i.e.*, above 1 kHz) than large ships. Source spectra for small craft and boats can, as for many vessels, include tonal harmonics at the resonant vibrational frequencies of propeller blades, engines, or gearboxes below about 1 kHz, as well as significant energy resulting from propeller cavitation extending up to and above 10 kHz. Due to the generally higher acoustic frequency and near-shore operation of many smaller boats and ships, sounds from these sources are generally regarded as having more geographically-limited environmental impacts. Small craft and boats may thus be less of a concern with regard to overall increases in low-frequency marine ambient noise from so-called ‘distant shipping’, although they can dominate some coastal acoustic environments,

[11] This would include whether the source projects sound as a ‘monopole’ or as a ‘dipole’. The source levels given below are intended as generalizations for each category and may not explicitly take these differences into account as some reports do not include complete information regarding directional properties of radiated sound fields.

[12] Various sampling bandwidths are commonly used in reporting sound signatures, including spectrum levels (1-Hz bandwidth), 1/3rd-octave, and 1-octave bandwidths. Where the term “broadband” is used here, it is intended to mean levels averaged across the entire frequency band sampled.

particularly partially-enclosed bays, harbours and/or estuaries (e.g., Kipple and Gabriel, 2003b). In areas, and during time periods, in which small vessel traffic (particularly when abundant and concentrated) overlaps marine animal distributions (particularly during sensitive life history stages such as breeding and nursery activities, migratory corridors and feeding), acoustic impacts from small craft may be a significant localized concern.

5.1.2 Medium-size ships

(length 50 - 100m; *e.g.*, support and supply ships, many research vessels)

Tugboats, crewboats, supply ships, and many research vessels in the 50 - 100m size class typically have larger and more complex propulsion systems, often including bow-thrusters. Typical broadband source levels for these small to mid-size vessels are generally in the 165 - 180 dB (re: 1µPa) range (Richardson *et al*., 1995; Kipple and Gabriel, 2003a; 2004; Heitmeyer *et al*., 2004). As for each of these size classes, there is considerable variability in the associated frequency spectra, although medium-sized ships tend to be more similar to large vessels in that the vast majority of sound energy is in the low-frequency band (below 1 kHz). While broadband source levels may be somewhat lower for mid-size ships on average than for the largest ocean-going vessels, there are certainly exceptions to this (e.g., as a function of age or maintenance of the ship), and mid-size ships can produce sounds of sufficient level and frequency to contribute to marine ambient noise in some areas. Mid-sized vessels spend the majority of their operational time in coastal or continental shelf waters, and thus overlap in time and in space with marine animals, many of which also prefer these waters for mating and feeding activities (see 5.3 for more details).

5.1.3 Large vessels

(length greater than 100m; *e.g.*, container/cargo ships, super-tankers, cruise liners)

A significant human contribution to the overall ambient underwater noise at low frequencies is thought to be generated by the growing use of the ocean for international shipping. Commercial ships, which are increasing in number, propulsion power and size, are producing ever-greater amounts of underwater noise as an incidental by-product of operation (Southall, 2005; U.S.G., 2008). Large commercial vessels produce relatively loud and predominately low frequency sounds. Although the exact characteristics of these depend on vessel type, size, operational mode and implemented noise-reduction measures, the strongest energy tends to be concentrated below several hundred Hz with broadband source levels generally in the 180 - 190 dB (re: 1µPa) range (Richardson *et al.*, 1995; Arvenson and Vendittis, 2000; Kipple, 2002; Heitmeyer *et al*., 2004; Kipple and Gabriel, 2004). Most of the acoustic field surrounding large vessels is the result of propeller cavitation (when vacuum bubbles created by the motion of propellers collapse), causing ships at their service speed to emit low-frequency tonal sounds at multiples of propeller blade rate (shaft speed in revs/second x number of propeller blades) and (high-frequency) noise spectra up to tens of kHz quite close to vessels. Smaller, but potentially significant, amounts of radiated noise can arise from on-board machinery (engine room and auxiliary equipment) (Richardson *et al.*, 1995). Heitmeyer *et al*. (2004) obtained recent measurements on individual commercial vessels, indicating that acoustic source levels are not necessarily a function of speed for modern diesel vessels, and that there are significant depth and aspect-dependences of radiated vessel sound fields as a function of shadowing and the Lloyd mirror effect near the surface of the water. Source (propeller) depth is also important in terms of long-range propagation, which is a potentially significant historical factor in terms of ambient noise trends due to shipping, as propeller depths have increased with increasing vessel size. Large vessels are loud near-field sources in both pelagic waters (where they are often concentrated in shipping routes and corridors) and coastal waters (where they are often concentrated in traffic lanes, waterways/canals or ports). Due to their loud and low frequency signatures, large vessels also dominate low frequency background noise in many marine environments worldwide (see 5.2 and Wenz 1962, 1969, Gray and Greeley 1980, Ross 1993, Greene and Moore 1995). Concerns regarding acoustic impacts associated with large vessel noise have focused mainly on animals that predominately use low frequencies to hear and to communicate (see 5.3). However, in addition to their predominant low-frequency radiated noise, modern cargo ships can radiate high frequency noise with 1/3-octave band source levels over 150 dB re 1µPa @ 1m around 30 kHz (Arveson and Vendettis, 2000), or broadband (0.354 - 44.8 kHz in this case) maximum RMS levels of 136 dB re: 1µPa at >700 m distance (Aguilar Soto *et al.*, 2006). Noise in these frequency bands has the potential to interfere (over relatively short ranges) with the communication signals of many marine mammals, including toothed whale species not commonly thought of in terms of shipping noise masking.

5.2 Trends in marine ambient noise

Many studies estimate that there has been an approximate doubling (3 dB increase) of background noise per decade in some ocean areas where sufficient longitudinal measurements support such analyses, particularly off the west coast of North America (Andrew *et al.,* 2002; Cato and McCauley, 2002; McDonald *et al.,* 2006; Andrew *et al.,* in press; see Figure 5.1 below). Over this period, commercial shipping density has increased considerably and is the most probable source of the increase, given that natural sound sources would be unlikely to change so dramatically over such a relatively short time.

Additionally, many other studies have characterized the relative contributions of shipping to low-frequency ambient noise in highly-trafficked and less-trafficked coastal and open ocean areas. These studies indicate that ships are the dominant source of low frequency noise in many, if not most, highly-trafficked coastal zones in the northern hemisphere. Unfortunately, empirical measurements prior to the advent of human noise contributions to the ocean are generally lacking and sufficient longitudinal measurements in all but a few areas are absent. This limits our understanding of precisely whether or not, and how, ocean ambient noise is increasing as a function of shipping or other human activities (*e.g.*, NRC 2000; 2003).

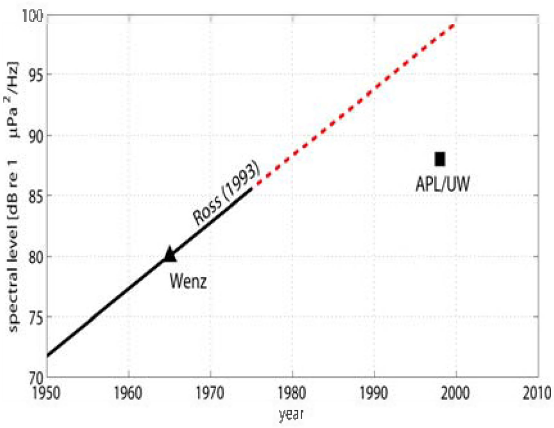

Figure 5.1: *Ambient noise measurements in the 100-200 Hz band measured off California in the 1950's (Ross, 1976) and APL/UW noise measurements in the late 1990's (Andrew et al., 2002) showing a projected and realized increase, presumably as a function of increased large vessel activity in the area over this period.*

5.3 Effects of shipping noise on marine mammals

The production, perception, and processing of sound is critical for various life functions (including communication, foraging, navigation, and predator-avoidance) of most, if not all, marine mammals. Marine mammals use sound as a primary means for underwater communication and sensing (Wartzok and Ketten, 1999). Specifically, the toothed whales have developed sophisticated bio-sonar capabilities to feed and navigate (Au, 1993); the large baleen whales have developed long-range communication systems using sound in reproductive and social interaction (Clark, 1990; Edds-Walton, 1997); and the pinnipeds make and listen to sounds for critical communicative functions (Schusterman, 1981; Schusterman *et al.*, 2000).

Some of the areas where marine ambient noise is known to be significantly increased at low frequencies as a function of vessel activities, are also heavily used by marine animals that depend on sound, many of which use the same low frequency bands (Cato, 1976; Ross, 1976; Worley and Walker, 1982; Zakarauskas, 1986; Bachman *et al.*, 1996; Zakarauskas *et al.*, 1990; Curtis *et al.,* 1999; Andrew *et al.*, 2002,Cato and McCauley, 2002; Heitmeyer *et al.*, 2004; McDonald *et al.* 2006; Hatch *et al.*, 2008). It is also evident that noise may interfere with critical biological functions in various ways, inducing alteration of behaviour, reduction of communication ranges for social interactions, foraging, and predator avoidance, temporary or permanent compromise of the auditory or other physiological systems, and/or, in extreme cases, habitat avoidance or even death (*e.g.*, Richardson *et al.*, 1995; NRC 2003, 2005; Clark and Ellison, 2004; Nowacek *et al.*, 2007; Southall *et al.*, 2007). Noise may also affect behaviour of animals and can also affect physiological functions and cause more generalized stress (Wright *et al.*, 2008). Additionally, the impacts of noise may be additive or synergistic to those of other human stressors (*e.g.*, Evans 2002). With regard to the incidental noise generated by vessels (*i.e.*, commercial shipping), the general low frequency band overlaps the frequencies generally produced by some marine animals, primarily large whales, seals and sea lions, and fish (Figure 5.2).

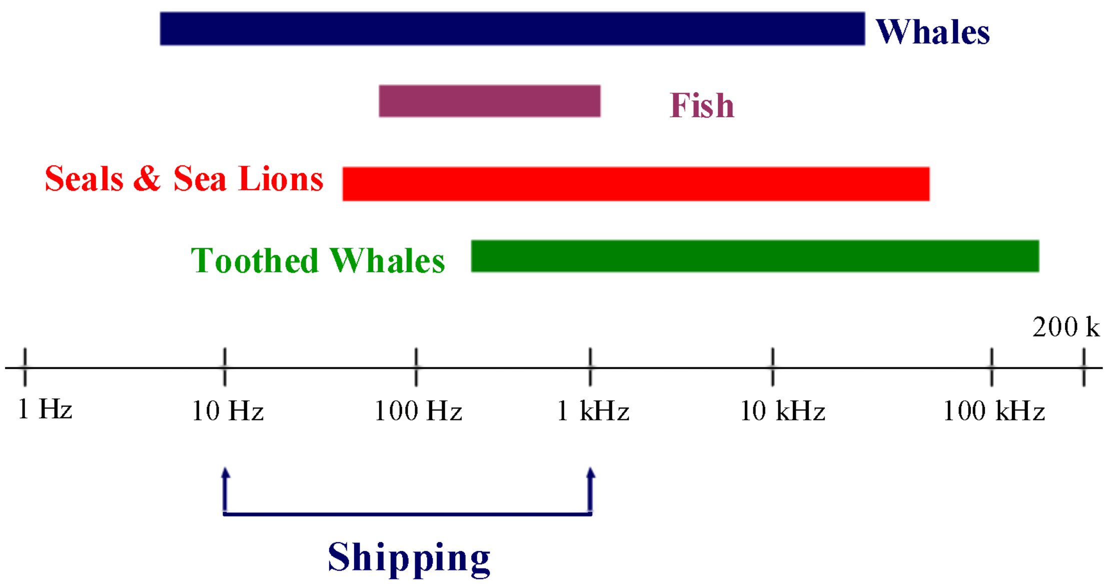

Figure 5.2: *Typical frequency bands of sounds produced by marine mammals and fish compared with the nominal low-frequency sounds associated with commercial shipping.*

5.3.1 Behaviour

A considerable limitation in considering the effects of anthropogenic noise on marine mammal behavior is that most studies are observational rather than experimental. Thus, in many conditions, particularly with regards to the effects of noise from large vessels on marine mammal behavior, available data lacks appropriate controls. Given that limitation, much of the recent data on the effects of vessel activities on marine animals indicate that various dolphin and whale species exposed to close physical approaches as well as noise from different vessels may alter motor behaviors (Janik and Thompson, 1996; Nowacek *et al*., 2001; Williams *et al*., 2002; Hastie *et al*., 2003) as well as vocalization characteristics (Lesage *et al*., 1999; Au and Green, 2000; Van Parijs and Corkeron, 2001; Buckstaff, 2004; Foote *et al*., 2004). These studies generally involve craft considerably smaller than tankers, container and dry bulk ships, and cruise liners, although some of these observations are presumably relevant to these larger sources as well (see Southall, 2005; Wright, 2008). Recently, studies have been conducted involving controlled sound exposure of animals fitted with specialized tags for monitoring movements, received sound fields, and, increasingly, physiological parameters. Using such techniques, manatees have been shown to respond to approaching vessels by changing fluke rate, heading, and dive depth (Nowacek *et al*., 2004). Perhaps the most important experiment to date concerning the effects of shipping noise on marine mammal behavior involved the use of acoustic tags and controlled exposure experiments with North Atlantic right whales. Five of six individual whales responded strongly (interrupted dive pattern and swam rapidly to the surface) to the presence of an artificial alarm stimulus (series of constant frequency and frequency modulated tones and sweeps) but ignored playbacks of vessel noise (Nowacek *et al*., 2004b). Finally, measurements using a sophisticated underwater listening array demonstrated that a Cuvier's beaked whale (*Ziphius carvirostris*) reduced the production of sounds associated with foraging in coincidence with a passing cargo ship (Aguilar Soto *et al.*, 2006).

5.3.2 Injury

In terms of direct physical injuries to hearing structures in marine mammals, it appears from the available data that quite loud and/or sustained exposures are required to cause even temporary changes in hearing sensitivity (see Southall *et al.*, 2007). Consequently, the likelihood that a single exposure to shipping noise would be sufficient to permanently damage the hearing of marine mammals appears to be remote. However, there are some important considerations and caveats to this conclusion, including the fact that the available information on how noise can damage hearing structures is very limited in terms of the species that have been tested (all in captive settings), and the fact that long-term (chronic) noise exposure has not been sufficiently investigated with regards to cumulative damage. Thus, there is the potential for permanent damage (injury) to hearing in marine mammals from sustained and/or repeated exposure to shipping noise over long periods, even if the available laboratory data on single exposures suggest that the ears of some species are particularly resilient to noise exposure.

There are also a range of physiological effects of noise exposure on marine mammals, which may exist even if an animal has learned to tolerate sound exposure and continues to feed or interact socially. Long-term noise exposure may induce stress responses in marine mammals, which are thought to be consistent across various species (Wright *et al.*, 2008), in a manner similar to humans who live near busy highways or airports (Evans, 2001; see references in Wright *et al.*, 2008).

5.3.3 Masking

The primary concern regarding potential adverse impacts of incidental shipping noise is not related to acute exposures, but rather to the general increase in continuous background ambient noise that may result from concentrations of vessel operations, and the potential masking (*i.e.*, simultaneous interference) of marine animals' communication systems echolocation signals, and passive listening capabilities (*e.g.*, for predator avoidance or general orientation). For example, masking can result in the disruption of breeding in animals that use sound during mating and reproduction, and of foraging in animals that use sound to detect prey. In addition, noise can mask important acoustic environmental cues that animals use to navigate and/or sense their surroundings, including sounds that are used to detect predators. The fact that noise masks hearing is well established for human beings (*e.g.*, Fletcher, 1940) and other animals, and it appears to be quite similar as a general phenomenon across many mammalian species (Fay 1986; Ward 1997). Numerous studies have examined the impacts that masking has on a variety of species, and have considered and/or modeled the extent to which low frequency noise from shipping can dramatically reduce communication ranges for marine animals (Payne and Webb, 1971; Erbe and Farmer, 1998, 2000; Southall *et al.*, 2000, 2007; Erbe 2002; Morisaka *et al.*, 2005, Nowacek *et al.*, 2007). Recent data on blue whales (*Balaenoptera musculus*) and North Atlantic right whales (*Eubalaena glacialis*) indicate that these species may be adjusting their vocalization (frequency and loudness) on both short and long timescales to compensate for masking associated with vessel noise (McDonald *et al.*, 2006; Parks *et al.*, 2007).

The greatest potential for masking exists for groups of marine mammals that produce and perceive sounds primarily within the lower frequencies contained in shipping noise; this includes the baleen whales, seals, sea lions, and fish, as well as the lowest social sounds of some of the toothed whales (Figure 5.2, above). The potential for masking at higher frequencies (1 – 25 kHz) exists when the vessel is in close proximity to the animal. In these close proximity circumstances, other marine mammals, including many toothed whales (e.g., beaked whales, sperm whales, dolphins and porpoises) may also experience masking from vessel noise. Because of the logarithmic nature of sound and what is known about hearing systems in mammals, seemingly small changes in background noise levels may result in large reductions of marine animals' communication ranges (Figure 5.3).

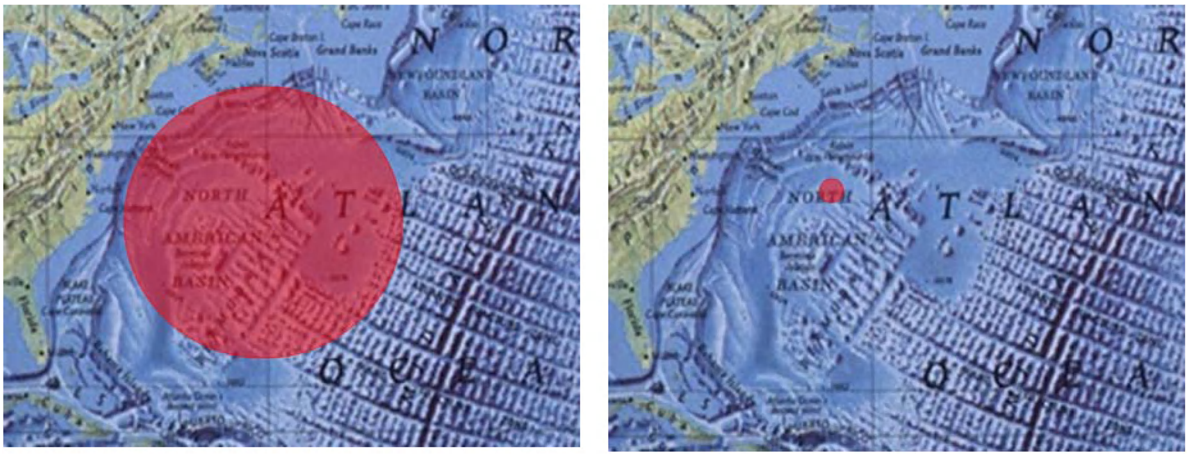

Figure 5.3: *Generalized examples of expected blue whale communication ranges from the many hundreds of square miles possible prior to the advent of commercial shipping and other industrialized sounds (left) compared to the greatly reduced possible ranges for those same voices today (right). While it is uncertain whether whales require or exploit the large ranges evident in the left panel, this example (based on historical and recent low frequency ambient noise and whale call measurements from a classified military recording system) gives an order-of-magnitude example of how changes in ambient noise may alter communication ranges for marine animals [Figure courtesy of Christopher Clark, Cornell University].*

5.4 Effects of shipping noise on fish

Vessel noise, in addition to potentially impacting marine mammals, also overlaps frequencies within the hearing and sound production ranges utilized by many fish species (Amoser *et al.*, 2004). Masking of fish sounds by shipping noise is potentially of greatest concern for species that produce low frequency spawning sounds central to reproductive success. Over 800 species of fish from 109 families worldwide are known to be soniferous (Kaatz, 2002), although this is likely to be a great underestimate. Soniferous fish include some of the most important commercial fish species, including many codfish, drum fish, grunts, groupers, snappers, jacks, and catfish. However, little is known of hearing capacities in most fish species, and fish that are not acoustically active may rely heavily on their acoustic awareness for predator/prey detection or general orientations. Continuous exposure (30 minutes) to recorded noise from small vessels has been shown to increase cortisol levels (stress response) in fish (Wysocki *et al.*, 2006). Additionally, hearing impairment (*i.e.*, temporary threshold shifts [TTS]), associated with long-term, continuous exposure (2 hours), and masked hearing thresholds have also been recorded for fish exposed to noise from small boats and ferries (Scholik and Yan 2001; Vasconcelos *et al.*, 2007). Furthermore, vessels (*i.e.*, trawlers, ferries, small boats) can also alter behavior in fish (*e.g.*, induce avoidance, alter swimming speed and direction, and alter schooling behaviour), similar to marine mammals (Engås *et al.*, 1995, 1998; Sarà *et al.*, 2007). However, it is often difficult in natural conditions to understand whether behavioral responses relate to vessels' presence, operating conditions and/or their noise.

Laboratory experiments have used vessel engine noise to examine noise impacts to fish in controlled settings (Scholik and Yan, 2001, 2002; Kastelein *et al*. 2008). Fathead minnows experienced TTS after the playback of noise from a 55-horsepower outboard engine (142 dB re: 1 μPa across frequency band from 19Hz-15 kHz; peak frequency was 1.3 kHz) for 2 hours, whereas goldfish exhibited a threshold shift after 10 minutes exposure to 160 – 170 dB re: 1 μPa broadband (0.1 to 10 kHz) white noise (Smith *et al.*, 2004). In these studies, the hearing returned to normal over time, but it appears that recovery time varies with the frequency of the sound and the duration of exposure. The amount of hearing loss appears to relate to how loud the noise is compared to the threshold of hearing at that frequency. At frequencies where a fish was more sensitive (*i.e.*, had a lower threshold), TTS produced by constant, broadband white noise was greater.

5.5 Effects of shipping noise on other species

Very few studies have addressed noise impacts to marine animals other than mammals and fish. However, some marine invertebrates produce sounds, including mussels, sea urchins, white shrimp, spiny and American lobster, and perhaps squid (*e.g.*, Iversen *et al*. 1963). In addition, a broader range of marine invertebrates, including those that do not use sound to communicate with conspecifics, may be impacted by reduced auditory awareness in conditions where shipping noise dominates bandwidths with important abiotic or biotic cues.

5.6 Mitigation

As noted above, the scientific understanding of exactly whether or not, when, and how shipping noise causes adverse effects on marine life (particularly regarding behavioural impacts) is currently quite limited. Thus, our appreciation of whether or not and how to mitigate potential impacts is similarly constrained. However, as noted above, sufficient data exist to conclude that acoustic communication is vitally important for many marine species. These varied functions may be negatively impacted by noise exposure (depending on conditions), and ambient noise levels in some biologically important areas appear to be increasing over time as a function of shipping noise (*e.g.*, NRC 2000; 2003; McDonald *et al.*, 2006; Hatch *et al.*, 2008; U.S.G., 2008; Wright, 2008). Given these general observations, reducing the overall noise output from marine vessels is likely to have demonstrable positive outcomes for acoustic communication, and may positively affect navigation, foraging efficiency and predator avoidance capabilities, and could reduce the likelihood of noise induced stress. Each of these outcomes could ultimately improve overall fitness for certain marine species in certain areas. Unlike persistent forms of pollution (*e.g.*, heavy metals), noise does not linger in the marine environment after it is introduced. Thus, the application of vessel-quieting technologies and/or operational strategies has the potential to reap immediate environmental benefits for marine life.

There is a reasonably long and successful history of quieting both surface and sub-surface military vessels to reduce their acoustic signature, and thus vulnerability to detection by enemy passive acoustics. Additionally, commercial applications of ship quieting technology are rapidly advancing in such areas as acoustic research vessel design, ferries, and environmentally-sensitive cruise ships. There are some commonalities in both of these quieting contexts, based purely on the physics of sound and constraints of vessel design, and many of the associated technologies focus on aspects of the propeller or other components of the propulsion systems. Reducing the overall noise level on board might also be beneficial to the ship's crew and passengers, while the reduction of structural vibrations might be beneficial to the integrity and lifetime of the vessel. Additionally, there may well be tangible benefits in terms of efficiency and reduced fuel consumption associated with reduced propeller cavitation, which will also reduce the overall radiated noise signature. Efforts at reducing noise are most effective when incorporated into the design of ships, though retro-fitting of vessels may also be successful to varying degrees, though at generally much greater cost. Minimizing propeller cavitation across the range of operating conditions is likely to remain the primary focus in efforts to quiet large vessels, given the fact that other noise sources (*e.g.*, machinery) will likely be overwhelmed by cavitation noise until considerable quieting treatments are applied (Southall *et al.*, 2004; 2007; Wright, 2008). Efforts to reduce structure-borne noise may be facilitated by advances in electrical propulsion systems which, provided that measures have been taken to reduce interference frequencies in the power supply, can enable the main engine room to be positioned away from the propeller shaft to a location where it can be acoustically isolated from transmitting underwater sound more easily. Additionally, operational measures (*e.g.,* routing and speed restrictions) could have positive outcomes in terms of ambient noise reduction in some areas. However, these must be carefully considered in light of potential related impacts arising from modifying traffic schemes (*e.g.*, possibly increasing noise in specific areas and possible impacts on likelihood of vessel strikes). The relative costs and environmental benefits of either technological or operational mitigation measures related to vessel noise output are not well-known. However, the United States has recently submitted a proposal to the Marine Environment Protection Committee of the International Maritime Organization to explicitly consider this international matter and consider a global strategy to address it (U.S.G., 2008).

5.7 Conclusions

While there is clearly missing information regarding the scope and nature of the environmental impacts associated with incidental noise radiated from marine vessels, there are some simple conclusions that may be drawn, that certainly apply to the marine region encompassed by the OSPAR area. Firstly, sound is clearly of vital biological importance to most, if not all, marine vertebrates and interference with acoustic functions as such communication, foraging, navigation and predator-avoidance may have various adverse effects. Secondly, marine ambient noise, as a result of vessel activities, may be increased on both acute and chronic time scales above natural conditions; in some areas there appears to be an increasing trend associated with increases in commercial shipping. Thirdly, while we are uncertain as to the exact biological significance of effects arising from shipping noise and certainly whether we have reached a critical point in terms of impacts to populations of marine animals, there is certain to be some level of adverse effect of noise introduction from the many tens of thousands of vessel noise sources, and minimizing or reducing their incidentally-radiated noise would be generally environmentally beneficial. Fourthly, there are existing technologies appropriate to both new design and retro-fitting of various vessel classes which, as well as carefully-considered operational measures, could minimize radiated underwater noise; the respective costs and benefits of these measures remain somewhat uncertain and should be explored further to mitigate adverse effects.

Although the potential effects of noise associated with major transportation projects on land (*e.g.*, airport and highway construction) are routinely considered in planning and construction, noise impacts associated with marine transport projects (*e.g.*, port or dock construction) are rarely assessed comprehensively, if they are assessed at all. When included in environmental impact analyses, results from models used to assess underwater noise impacts for offshore commercial projects vary widely both in the quality and quantity of the information they provide for resource management. Noise impacts associated with changes in the distribution, density and/or composition of shipping traffic within coastal areas (*i.e.*, new routing measures, consolidations of lanes, local changes in operational conditions etc.) should be assessed and taken into consideration by national, regional and/or international bodies, whose jurisdictions include environmental impacts associated with maritime transport, port operations and/or coastal waterways. Environmental impact analyses could incorporate empirical data, when possible, regarding local/regional ambient noise profiles, and use standardized, open-source and/or peer-reviewed modeling approaches to predict changes in these profiles resulting from proposed changes in shipping.

More information is badly needed regarding the near- and far-field impacts of shipping in different marine environments. These data must take temporal and spatial variation into account. To address near-field dynamics, acoustic monitoring designed to capture the before- and after-impacts of changes in the contemporary distribution, density and/or composition of shipping traffic would assist both the maritime transportation industry and resource managers to more accurately assess potential impacts to species of concern in local areas. Due to the long range transmission capabilities of shipping noise, mitigation must also address shipping noise impacts experienced by populations relatively distant from highly trafficked areas. In this case, only internationally-focused initiatives addressing the number, average sound profiles and operating conditions of ships will effectively address the increasing degradation of acoustic habitat.

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Module 6: Sonar

6.1 Sound profiles of sonar devices

The use of acoustic energy for locating and surveying is described as active sonar. Sonar was the first anthropogenic sound to be deliberately introduced into the oceans on a wide scale. There are a variety of types of sonars that are used for both civilian and military purposes. They can use all sound frequencies and are categorised for convenience in this document into low (<1 kHz), mid (1 to 10 kHz) and high frequency (>10 kHz). These definitions may differ from those used by some navies in the OSPAR area. Military sonars use all frequencies (but low frequency sonar, as defined here, is not used in the OSPAR area), while civilian sonar uses some mid but mostly high frequencies. Low frequency sonars are not used in OSPAR waters so are not dealt with further in this module except where lessons from their usage elsewhere may inform assessment of the effects of sonars of other frequencies.

6.1.1 Mid frequency sonar

Military mid frequency sonars are used to survey areas tens of kilometres in radius and are used to find and track underwater targets. A US Navy hull-mounted system (AN/SQS-53C) sonar system uses pulses in the 2 – 10 kHz range (normally 3.5 kHz) and has operated at relatively high source levels of 235 dB re 1 µPa @ 1m with ping lengths of about 2.5 sec. Another system (AN/SQS-56) uses this same frequency band but with lower source levels (223 dB re 1 µPa @ 1m) (Evans & England 2001). Mid-frequency sonars are used by many navies of the world, but source levels used by navies from north-east countries are significantly lower than that of AN/SQS-53C. Within the OSPAR area, usage of these systems has been confined to comparatively well-defined exercise areas, which make up a small proportion of the world's oceans. Even in these areas, activity times are relatively short and episodic, and propagation distances are comparatively small (10s kilometres) because of the frequencies and source levels involved. In addition, only a small proportion of the naval fleets ships using the OSPAR area carry these sonar systems. These systems were formerly used predominantly for antisubmarine work in open water, but more nations are operating submarines and these have become more able to operate closer to the shore, with the consequence that antisubmarine systems including sonar now need to operate in the littoral environment.

Some non-military sonars also operate in this frequency band. Bathymetric sonars use these frequencies for wide-area, low resolution surveys. The Fugro Seafloor survey model SYS09 for instance uses both 9 and 10 kHz transducers operated at 230 dB re 1 µPa @ 1m. Sub-bottom profilers typically use 3.5 kHz transducers operated at source levels of 204 dB re 1 µPa @ 1m. The regional resolution GLORIA survey sonar uses 6-7 kHz band (no source level published).

6.1.2 High frequency sonar

Military high-frequency sonars are either used in attacking (mines or torpedoes) or defending (mine countermeasure, anti-torpedo) systems and are designed to work over hundreds of metres to a few kilometres. These sonars use a wide range of modes, signal types and strengths. As with other military sonars, their usage is generally confined to exercise areas.

Fish finders and most commercial depth sounders operate at high frequencies. In principle, they project a lower power signal and have narrower beam patterns and shorter pulse lengths (a fraction of a second) than military sonars. These systems cannot be used at shallow depths at high powers due to cavitation (Urick 1975). Most of the systems focus sound downwards, though some horizontal fish finders are available. Fish finding sonars operate at frequencies typically between 24 and 200 kHz, which is within the hearing frequencies of some marine mammals, but above that of most fish. Globally there are a great many recreational, fishing and commercial vessels, most of which are fitted with some sort of sonar. These vessels are most heavily used in shallow shelf-seas, with sonars used less by those merchant vessels crossing deep water areas. Usage occurs throughout the year and both by day and night. Some horizontally-acting fish finding sonars work at frequencies at the lower end of the 'high-frequency' range and are relatively powerful. An example is the Furuno FSV-24 sonar that operates at 24 kHz and can detect and track shoals of tuna at 5000m horizontally. Source levels of these sonars are not published.

Some depth finding sonars can also be powerful. Boebel *et al.* (2004) describe the Atlas hydrosweep DS-2 deep sea multi-beam sonar. This has source levels exceeding 220 dB re 1 μPa @ 1m at 15.5 kHz with relatively short (24ms) pulses.

Most sonars operate at one frequency of sound, but generate other unwanted frequencies (e.g. harmonics of the fundamental frequency due to non-linear processes). These extraneous frequencies are rarely described (but are usually of a much lower intensity than the main frequency) and may have wider effects than the main frequency used, especially if the extraneous frequencies are much lower than those used (and could consequently propagate further).

6.2 Effects on Marine Mammals

6.2.1 Injury and physical effects

There is one recent experimental study that has shown that mid-frequency sonar can induce temporary reduction of hearing ability in a bottlenose dolphin (Mooney *et al.* 2009a, b). This study concluded that sonar can induce physiological and behavioural effects in at least one species of odontocete; however, exposures must be of prolonged, high sound exposure levels to generate these effects. Observations of injury are almost impossible to obtain under natural conditions.

6.2.2 Non-auditory tissue damage

Much research effort on the potential for anthropogenic sound to affect marine mammals has focused on auditory effects and behavioural modifications following sound exposure. Non-auditory consequences resulting from exposure to sound (not including blast effects) have historically received less attention (Crum & Mao 1996). Studies on terrestrial mammals suggest that non-auditory tissues require exposure to sounds considerably more intense than those that affect hearing. Biologically, this extrapolation suggests that direct tissue damage can occur only very close to an intense sound source.

The first hypothesis about non-auditory consequences of less intense exposures was proposed in the report of the Greek cetacean stranding event (see Section 6.2.6) and considered the concept of acoustic resonance in air spaces. All structures have a natural frequency at which they vibrate, called their resonant frequency. If such a structure is struck by an incoming sound wave of the same frequency as the resonant frequency the structure vibrates at a greater amplitude than normal; the tissues move more than normal and may tear. Acoustic resonance was suggested as a possible explanation of the Bahamian stranding (see Section 6.2.7) but a detailed scientific review (Evans *et al.* 2002) concluded that the resonant frequencies of marine mammal lungs are too low for resonance to be caused by mid-frequency sonar.

The second hypothesized, non-auditory link between strandings and sonar exposure is acoustically mediated bubble growth (*e.g.* rectified diffusion) within tissues that is proposed to occur if tissues are supersaturated with dissolved nitrogen gas (Crum & Mao 1996). Such bubble growth could result in gas emboli formation, tissue separation and increased, localised pressure in tissues, a similar scenario to decompression sickness (DCS) in human divers. Although the rectified diffusion model of Crum & Mao (1996) suggested that received sound levels of >200dB (re: 1 µPa @ 1m) would be needed to drive significant bubble formation in marine mammal tissues, the model was run under relatively low levels of tissue nitrogen supersaturation (100 – 200%). A later study predicted that beaked whales, due to the typical dive profile characteristics, may accumulate over 300% nitrogen tissue supersaturation at the end of a typical dive sequence (Houser *et al.* 2001). This study, based on empirical observations of nitrogen tissue accumulation in bottlenose dolphins (Ridgway & Howard 1979) and dive data from northern bottlenose whales *Hyperoodon ampullatus* (Hooker & Baird 1999), suggested that beaked whales in particular may be more susceptible to acoustically mediated bubble formation than originally predicted by Crum & Mao (1996).

The first evidence of gas and fat emboli and acute and chronic gas bubble lesions has been reported in a number of cetacean species stranded in Europe. In the UK, ten stranded cetaceans comprising four Risso's dolphins *Grampus griseus*, four common dolphins *Delphinus delphis*, a Blainville's beaked whale *Mesoplodon densirostris* and a harbour porpoise *Phocoena phocoena* had acute and chronic lesions in liver, kidney and lymphoid tissue (lymph nodes and spleen) associated with (predominantly) intravascular gas bubbles (emboli) (Jepson *et al.* 2003, 2005; Fernández *et al.* 2004). These animals stranded singly and the etiology of these lesions (including whether or not they were exposed to any form of acoustic activity) is unknown. However, a suite of widely disseminated microvascular haemorrhages associated with gas and fat emboli, lesions highly consistent with DCS, were found in ten beaked whales that died as part of a mass stranding of 14 beaked whales in the Canary Islands linked to an international naval exercise (Neo Tapon) in September 2002 (see Section 6.2.8). The Canaries findings are important for understanding effects on tissues as they are the first to be based on fresh material. In other similar incidents, either tissues were not examined, or were examined much later. The gas bubble hypothesis has received much theoretical attention and evidence, however there has as yet been little scientific testing of it. Such testing is needed and necessary before a full judgement of the hypothesis can be made.

Although a number of anatomical, physiological, and behavioural adaptations that presumably guard against nitrogen bubble formation in marine mammals have been proposed (Ridgway 1972, 1997; Ridgway & Howard 1979, 1982; Falke *et al.* 1985; Kooyman & Ponganis 1998, Ponganis *et al.* 2003), it is possible that the gas emboli and associated lesions found in cetaceans in the Canary Islands and in the UK (Jepson *et al.* 2003; Fernández *et al.* 2004, 2005, Rommel *et al.* 2006) could be caused by disruption of these evolutionary adaptations to deep diving. Anatomical and physiological adaptations to diving are unlikely to alter in the short course of acoustic exposure, but behavioural changes in response to sonar might (Tyack *et al.* 2006). For example, in experiments northern right whales *Eubalaena glacialis* responded to novel acoustic stimuli by changing behaviour including a combination of accelerated ascent rates and extended surface intervals at received sound levels as low as 133 dB re 1 µPa @ 1m (Nowacek *et al.* 2004). If beaked whales change behaviour to a series of shallower dives with slow ascent rates and shorter stays on the surface they could experience excessive nitrogen tissue supersaturation driving potentially damaging bubble formation in tissues via a similar mechanism to the human diver that incurs DCS due to too rapid an ascent. Alternatively, physical mechanisms (*e.g.* rectified diffusion) exist for acoustically-mediated bubble formation in tissues already supersaturated with nitrogen (Crum & Mao 1996; Houser *et al.* 2001).

It is therefore theoretically possible that sonar transmissions (of low, mid or high frequency) could directly initiate or enhance bubble growth if tissues were sufficiently supersaturated with nitrogen and if the received sound pressure levels were of sufficient intensity. However, there is as yet no scientific evidence for any of the steps in these postulated chains of events. A US Marine Mammal Commission Workshop on beaked whales and anthropogenic noise considered it important to test the “bubble hypothesis”, and prioritised a programme of research that incorporates both acoustically mediated bubble formation and bubble formation via a DCS-like mechanism, and includes the use of controlled exposure experiments (Cox *et al.* 2006).

6.2.3 Masking and changes in vocal behaviour

Cetaceans use sound for a number of purposes including communication, searching for food and detecting predators (see module 3). In all cases, a cetacean needs to hear a sound, either originating from itself (with an echo reflected from a target) or originating somewhere else and may not be very loud. In order to detect the sound, the sound has to be louder than (or be able to be differentiated from) the ambient sound level. The hearing mechanisms or auditory processing of the whale also has to be sensitive enough to detect this difference.

Studies have shown that sonar signals induce changes in the use of biological relevant signals in some species. Long-finned pilot whales *Globicephala melaena* changed the type of vocalisation in the presence of military sonar signals (Rendell & Gordon 1999). Some humpback whales lengthened their song cycles when exposed to the LFA (low frequency) source (Miller *et al.* 2000; Fristrup *et al.* 2003); increasing the redundancy of the song may improve communication in a noisier channel. Vocalisation by beaked whales has been shown to reduce or cease in some cases (Cressey 2008). Note that it is difficult to separate the two possible causes of these changes – masking by the sound and direct behavioural response to the sound.

Thus, sonar signals could affect several vocal characteristics or behaviours of cetaceans. The degree to which these changes significantly affect the animals is not known and will be case dependent. Sonar is a lesser contributor to the overall ocean noise budget than other sources of anthropogenic sound and masking through sonar is therefore unlikely to be an important issue. Furthermore, mid- and high frequency sonar signals are probably too short to interfere with most cetacean sounds to any substantial extent.

6.2.4 Behavioural change

Besides the changes in vocal behaviour outlined above, many possible changes in behaviour could occur in the presence of additional noise. Behavioural responses may range from changes in surfacing rates and breathing patterns to active avoidance or escape from the region of highest sound levels.

Many now hypothesise that the mechanism(s) underpinning the phenomenon of beaked whale mass strandings linked to naval sonar are initially triggered by a behavioural response to acoustic exposure rather than a direct physical effect of acoustic exposure (Jepson *et al.* 2003; Fernández *et al.* 2004, 2005; Cox *et al.* 2006). The first potential pathway entails a simple behavioural response to sound that leads directly to stranding, such as swimming away from a sound into shallow water. An alternative scenario involves a behavioural response leading to tissue damage. Such responses could lead to gas bubble formation, hypoxia, hyperthermia, cardiac arrhythmia, hypertensive haemorrhage, or other forms of trauma (Cox *et al.* 2006).

Another explanation has been suggested based on the observation that strandings appear to be correlated with high sea temperatures (Cole 2005). It is suggested that panicked animals suffer heat stress as a result of over-exertion in water that is too warm to effectively cool animals adapted to spend most of the time in cold water below the thermocline. Both 'gas-bubble lesion' and 'heat stress' hypotheses are consistent with the apparent susceptibility of deep diving species.

6.2.5 Review of literature on effects of sonar on beaked whales

There have been several reviews of the possible effects of sonar on marine life including beaked whales (*e.g.* ICES 2005a). Cuvier´s beaked whales *Ziphius cavirostris* are the most common species involved in these events, probably due to its cosmopolitan distribution. Cuvier's beaked whale is a deep-diving, pelagic cetacean that until recently was believed to rarely mass-strand (Heyning 1989). Only seven strandings of more than four individuals were recorded by Frantzis (1998) from 1963 to 1996 worldwide, but more incidents in this period have come to light recently. On most of these occasions, mass strandings showed atypical characteristics unlike those that occur with other whales. This suggested that the cause had a large synchronous spatial extent and a sudden onset. Such characteristics are also shown by sound in the ocean.

Research on LFAS began by NATO in 1981 (NATO-Saclantcen 1993) and the US Navy's research on SURTASS LFA began in about 1986 and a statement on its environmental impact was formally initiated in July 1996. It is worth noting that the first reported and recorded atypical mass stranding of Cuvier's beaked whale was in 1963 (Tortonese 1963), shortly after the time that a new generation of powerful mid frequency tactical sonars became widely deployed (Balcomb & Claridge 2001).

Hildebrand (2004) published a list (compiled by James Mead) of strandings of two or more Cuvier's beaked whales based on records at the Smithsonian Institution and recent literature (Table 6.1). This list is unlikely to be complete, but it represents all cases presently known. In only five of the cases, Greece 1996, Bahamas 2000, Madeira 2000, Canary Islands 2002 and Canary Islands 2004, is it documented that navy vessels were in the area, operating sonar at the time and place of the stranding, and partial or complete necropsies were undertaken. No necropsy results are available for any of the other events. It should be noted that, due to a lack of information, it has often proven very difficult to determine whether or not military sonar was in use sufficiently near the stranding sites to be considered as a possible cause of the stranding; in the above five cases the source vessels were up to 80 nautical miles from the site of the stranding. It is also worth noting that some other strandings, not categorised as mass strandings, could be caused by the same mechanism as behind the mass stranding. These records have not been reviewed for possible correlation with presence of naval vessels. There have also been other recent mass strandings of deep-diving cetaceans other than Cuvier's beaked whale that were contemporaneous with naval exercises (*e.g.* Wang & Yang 2006, Yang *et al.* 2008).

Since the stranding in the Kyparissiakos Gulf, Greece in 1996 (see Section 6.2.6), there has been increasing attention paid to the effects of sonar. Sections 6.2.7 and 6.2.8 illustrate some of the findings based on two further case studies of incidents.

***Table 6.1**: Strandings involving at least 2 Cuvier's beaked whale Ziphius cavirostris (Zc) (after Hildebrand 2004; Brownell et al. 2004; Martín et al. 2004; Litardi et al. 2004; Dolman et al. 2008). "Strandings" refers to individuals that became stranded on beaches and does not imply death (some were redirected out to sea and their fate is unknown). Items listed 'U.S. Fleet?' and 'Naval manoeuvres' represent mostly the word of locals that military ships might have been in the general area and cannot be taken as necessarily linked. These records also represent the only known multiple stranding events for Gervais' beaked whale Mesoplodon europaeus (Me), Blainville's beaked whale Mesoplodon densirostris (Md) and Sowerby's beaked whale Mesoplodon bidens (Mb).*

Year	Location	Species (numbers)	Concurrent activity, when available
1914	New York, United States	Zc (2)	
1960	Sagami Bay, Japan	Zc (2)	?US Fleet
1963	Gulf of Genoa, Italy	Zc (15+)	Naval manoeuvres
1963	Gulf of Genoa, Italy	Zc (15+)	Naval manoeuvres
1963	Sagami Bay, Japan	Zc (8-10)	?US Fleet
1964	Sagami Bay, Japan	Zc (2)	?US Fleet
1965	Puerto Rico	Zc (5)	
1966	Ligurian Sea, Italy	Zc (3)	Naval manoeuvres
1967	Sagami Bay	Zc (2)	?US Fleet
1968	Bahamas	Zc (4)	
1974	Corsica	Zc (3), Striped dolphin (1)	Naval patrol (?not sonar)
1974	Lesser Antilles	Zc (4)	Naval explosion
1975	Lesser Antilles	Zc (3)	
1978	Sagami Bay, Japan	Zc (9)	?US Fleet
1978	Sagami Bay, Japan	Zc (4)	?US Fleet
1979	Sagami Bay, Japan	Zc (13)	?US Fleet
1980	Bahamas	Zc (3)	
1981	Bermuda	Zc (4)	
1981	Alaska, United States	Zc (2)	
1983	Galapagos	Zc (6)	
1985	Canary Islands	Zc (12+), Me (1)	Naval manoeuvres
1986	Canary Islands	Zc (5), Me (1), *Ziphiid* sp. (1)	
1987	Canary Islands	Me (3)	
1987	Sagami Bay, Japan	Zc (2)	?US Fleet
1987	Italy	Zc (2)	
1987	Canary Islands	Zc (2)	
1988	Canary Islands	Zc (3), bottlenose whale (1), pygmy sperm whale (2)	Naval manoeuvres
1989	Sagami Bay, Japan	Zc (3)	?US Fleet
1989	Canary Islands	Zc (15+), Me (3), Md (2)	Naval manoeuvres
1990	Sagami Bay, Japan	Zc (6)	?US Fleet
1991	Canary Islands	Zc (2)	Naval manoeuvres
1991	Lesser Antilles	Zc (4)	
1993	Taiwan	Zc (2)	
1994	Taiwan	Zc (2)	
1996	Greece	Zc (12)	LFAS trials (see Section 4.2.2)
1997	Greece	Zc (3)	
1997	Greece	Zc (9+)	Naval manoeuvres

Year	Location	Species (numbers)	Concurrent activity, when available
1998	Puerto Rico	Zc (5)	
2000	Bahamas	Zc (8), Md (3), *Ziphiid* sp. (2), minke whale (1), *Balaenoptera* sp. (2), Atlantic spotted dolphin (1)	Naval mid-frequency sonar (see Section 4.2.3)
2000	Galapagos	Zc (3)	
2000	Madeira	Zc (3)	Naval mid-frequency sonar
2001	Solomon Islands	Zc (2)	
2002	Canary Islands	Zc, Me, Md (15-17 whales)	Naval mid-frequency sonar (see Section 4.2.4)
2002	Mexico	Zc (2)	RV Ewing seismic
2004	Canary Islands	Zc (4)	Naval mid-frequency sonar[13]
2006	Gulf of Almeria, Spain	Zc (4)	Naval manoeuvres
2008	West Scotland & Ireland	Zc (14), Mb (5), unid beaked (4), Long-finned pilot whale (22)	

Møhl (2004) points out that sperm whale clicks bear some resemblance to those of tactical sonars. The main differences are the ping energy – a receiving animal would need to be 30 - 100 times closer to a sperm whale than tactical sonar to receive the same sonic energy. The duty cycle (or proportion of overall time that the noise is made) is also much higher in tactical sonar and the directionality of each is different – sperm whales emitting a very narrow beam of sonic energy compared to the wide radiation pattern of the tactical sonars. Møhl (2004) felt that this similarity in properties between a natural noise source compared with the novel sources indicated that behavioural rather than physiological causes would be more likely to cause the multiple beaked whale strandings. However, physical damage may be caused both by high peak pressure and total energy flux (Finneran *et al.* 2002).

6.2.6 Case study: Greece

During the early hours of the morning of 12 May 1996, Cuvier's beaked whales started to strand alive in several locations along Kyparissiakos Gulf (a long sandy beach alongside the Hellenic Trench in the west coast of the Peloponnese (Frantzis 1998, Figure 6.1). The strandings continued until the afternoon of 13 May 1996. A few more specimens (4 - 5) were reported as stranded and rescued, entangled and rescued, or swimming very close to the coasts during the next 3 days, however, only one of these reports could be confirmed. In total, 12 stranded whales were recorded on 12 and 13 May. They were spread along 38.2 kilometres of coast and were separated by a mean distance of 3.5 km (s.d. = 2.8, n = 11) (Figure 6.1). Another whale stranded on 16 May and was driven back to the open sea. Two weeks later, one more animal was found decomposing on a remote beach of the neighbouring Zakynthos Island, 57 km away from the closest stranding on the mainland. Eleven of the whales were measured and sampled. Nine of them were immature males with no erupted teeth and two were females. The recorded spread of the stranded animals in location and time was atypical, as whales usually mass-strand at the same place and at the same time. The term "atypical mass stranding" has been proposed for the recorded strandings as opposed to typical mass strandings known mainly from pilot whales *Globicephala* sp. and false killer whales (Geraci & Lounsbury 1993).

[13] See Jaber *et al.* (2004) for details on the stranding event.

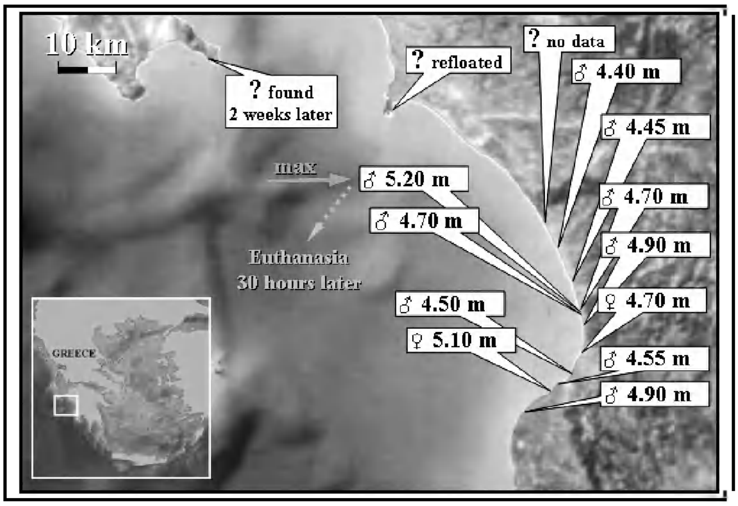

Figure 6.1: *Position, sex and total length of the 14 Cuvier's beaked whales that were recorded during, or shortly after the mass stranding of 12 and 13 May 1996 in Kyparissiakos Gulf, Greece.*

Necropsies of eight stranded animals were carried out, but no apparent abnormalities or wounds were found. These necropsies were limited to basic external examination and sampling of stomach contents, blood and skin. No ears were collected; no entire organs or histological samples were conserved because of many problems related to permits, lack of facilities and means, and lack of relevant knowledge and trained specialists. Stomach contents had variable quantities of squid remains (like beaks and ocular lenses) from three different squid species. Many of them contained cephalopod flesh, indicating that recent feeding had taken place.

All available information regarding the conditions associated with the mass stranding of May 1996 was gathered, and many potential causes were listed and examined. The most important of them were major pollution events, important tectonic activity, unusual geochemical/physical/meteorological events, magnetic anomalies in the area, epizootics and conventional military exercises. However, none of the potential causes listed above coincided in time with the mass stranding or could explain its characteristics (NATO-Saclantcen 1998). Several months after the mass stranding a warning to mariners issued by the Greek Hydrographic Service was found by cetacean researchers that provided significant relevant information. This warning (586 of 1996) stated that 'sound-detecting system trials' were being performed by the NATO research vessel *Alliance* from 24:00 11 May to 24:00 15 May - a period that encompassed the mass stranding. The officially declared area where the sea trials had been carried out enclosed all the co-ordinates of the stranding points. The tests performed were for Low Frequency Active Sonar (LFAS; term used by NATO to describe their dual low- and mid-frequency active sonar), a system that introduces very high levels of low and medium frequency sound into the marine environment to detect quiet diesel and nuclear submarines. Detailed information regarding the time schedule, the runs (Figure 6.2) and the specific sound characteristics of the transmissions became declassified and available through NATO-Saclantcen by the autumn of 1998 (NATO-Saclantcen 1998). The *Alliance* was using high power active sonar, transmitting simultaneously to both low (450-700 Hz) and mid (2.8 - 3.3 kHz) frequencies, at a maximum output of 228 dB re 1 µPa @ 1 m, which enables long detection ranges.

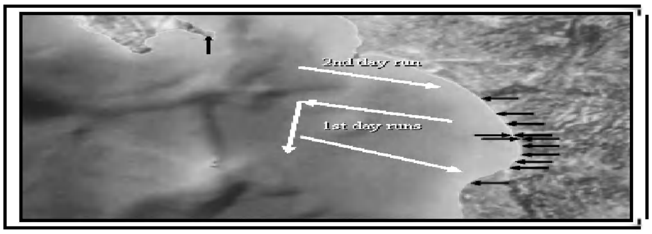

Figure 6.2: Routes of the first and second day of the low and mid-frequency sonar tests (12 and 13 May 1996) according to NATO-Saclantcen (1998). The black arrows indicate the stranding positions of the whales during the same days, with the position of the whale found in Zakynthos Island.

Although the available data in 1996 could not directly prove that the use of active sonars caused the mass stranding in Kyparissiakos Gulf, the evidence clearly pointed to the sonar tests. The main arguments and the supporting evidence are listed below:

- At least 12 of the 14 animals stranded alive in an atypical way.
- The condition of the stranded animals, along with the analyses of their stomach contents was not consistent with pathogenic causes (which in any case are not known to provoke atypical mass strandings).
- No unusual environmental events occurred before or during the stranding (e.g. tectonic activity, magnetic anomalies, geophysical or geochemical events, meteorological events etc.).
- The stranding characteristics suggested a cause with large synchronous spatial extent and sudden onset (*i.e.* those shown by sound in the ocean).
- Most importantly, the probability for the two events (*i.e.* the sonar tests and the mass stranding) to coincide in time and location, while being independent, was extremely low. In other words if the 16.5-year period before the mass stranding is considered (1981 was chosen arbitrarily because this was the year that NATO started to experiment on sonar, and we are sure that no mass stranding, nor other tests of sonar had occurred in the area since that year), the probability of a mass stranding occurring for other reasons during the period of the sonar tests (*i.e.* from 12 to 15 May 1996 instead of any other day) is less than 0.07%.

With the benefit of hindsight, after further mass strandings that followed the Greek case with similar characteristics and always in close association with naval exercises and use of mid-frequency active sonar in the Bahamas (see below), Madeira, and Canary Islands (see below), there is no dispute in the scientific and military communities regarding the cause of the mass stranding in Kyparissiakos Gulf.

The Cuvier's beaked whale stranding history of Kyparissiakos Gulf (Figure 6.3) shows that although no mass strandings had been recorded before the 12 May 1996 the strandings of this species were not rare. The average stranding rate was 0.88 individual/half year (s.d. = 0.99, n = 8). After the mass stranding of May 1996, the stranding rate was reduced to less than one third of what it was before the mass stranding (0.25 individual/half year, s.d. = 0.45, n = 12). This result indicates that the damage could have been significantly higher than the death of the stranded whales. Others may have left the area or may have died in the deep offshore waters.

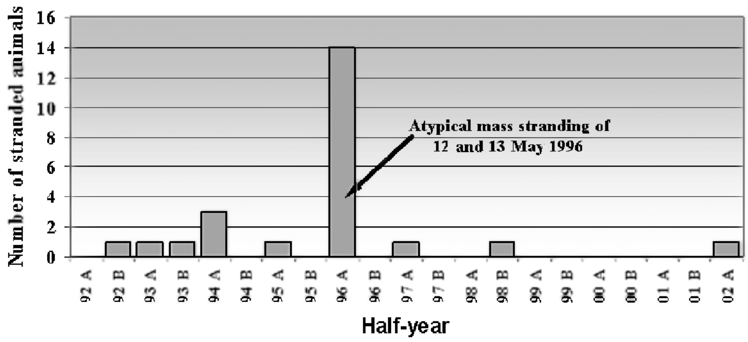

Figure 6.3: *The stranding history of Cuvier's beaked whales in Kyparissiakos Gulf from 1992 to 2002.*

6.2.7 Case study: Bahamas

On 14 and 15 March 2000, five U.S. Naval ships using mid frequency (2-10 kHz) sonar transited the Northeast and Northwest Providence Channels of the Bahamas Islands in an anti submarine warfare exercise lasting 16 hours. The ships were using two types of mid frequency sonar, designated AN/SQS-56 and AN/SQS-53C, that differed somewhat in their operating characteristics. The AN/SQS-56 closely resembles mid-frequency tactical sonars used by many other navies of the world. The ships operated in two loosely coordinated groups that passed through the channel six hours apart.

Beginning on 15 March, only hours after the first group of ships passed, and continuing for the next 36 hours, 17 cetaceans were found stranded dead or alive, or in shallow water, along a 240 km stretch of the Northeast and Northwest Providence Channels on three islands (Figure 6.4). The Bahamas Marine Mammal Survey discovered the stranding, and they, Dr. Alan Bater, veterinarian for Bahamas Department of Fisheries, and members of the public pushed some of the stranded animals back into deeper water, and preserved for post mortem examinations tissues from those that died. The preserved specimens were shipped to the U.S. mainland and distributed among several pathologists for broad-based analysis of the cause of death.

The U.S. Navy was informed about the event and immediately started summarizing ship tracks and times, and modelling acoustic propagation from the sonars. The National Oceanic and Atmospheric Administration (NOAA) sent representatives of its stranding programme to the Bahamas to assist in handling the biological specimens. It later sent specimens to a number of researchers for histological and toxin studies. The Navy and NOAA each prepared verbal reports of their own findings, and in June 2000, the two agencies met for the first time and exchanged information. Subsequently, each agency prepared a written version of its report and submitted them to two editors (Cdr. Paul Stewart for Navy and Dr. Roger Gentry for NOAA) who compiled an interim report on progress to date. The report included the results of NOAA acoustic monitoring of the Bahamas region on the days of the sonar exercise and stranding (Evans & England 2001).

The Navy sonar systems produced a sound approximately every 12 seconds. Except for one four-hour period when one of the ships produced source levels that are classified, the source levels of all ships during the remainder of the exercise did not exceed 235 dB re 1 µPa. Complex propagation modelling showed that because of a surface duct, the sound was largely confined to the top 200 m of the water column, and that in many areas of the channel levels of 160 dB re 1 µPa would have occurred (Evans & England 2001). Reverberation from the walls or floor of submarine canyons is not thought to have added much to these levels because of the surface duct. Whale locations at sea were unknown, so received levels cannot be estimated with confidence.

The animals that stranded included Cuvier's and Blainville's beaked whales, minke whales *Balaenoptera acutorostrata*, and an Atlantic spotted dolphin *Stenella frontalis*. An animation that plotted ship positions by time, and the time and place of each stranding showed a close temporal and spatial correlation for all but the spotted dolphin.

Seven of the stranded animals died, five Cuvier's beaked whale, one Blainville's beaked whale, and the Atlantic spotted dolphin. The latter may have died of causes not associated with acoustic exposure, and in a very different location than the beaked whales. Four of the beaked whales showed some evidence of auditory structural damage, including bloody effusions near and around the ears. The two freshest specimens showed subarachnoid haemorrhage and blood clots in the lateral brain ventricles. It is reasonable to assume the haemorrhages were acoustically induced. The immediate cause of death appeared to be cardiovascular collapse and physiological shock which together commonly result in death after stranding.

NOAA's investigation considered every possible cause of the stranding event, and eliminated all except sonar as the triggering event. Explosions were eliminated by NOAA's acoustic recordings. The evidence that most strongly suggested sonar as the triggering event was the close temporal and spatial match between sonar passage and the stranding events. The underlying mechanism by which sonar had this effect is still not known. It is possible but highly doubtful that direct acoustic exposure of tissues caused the lesions observed. All animals would have had to be very close to the vessels to receive such exposure, which seems unlikely. It is possible that sonar triggered some kind of unfavourable behavioural response which led to stranding and to subsequent tissue injury. It is also possible that some injury occurred before and some after stranding.

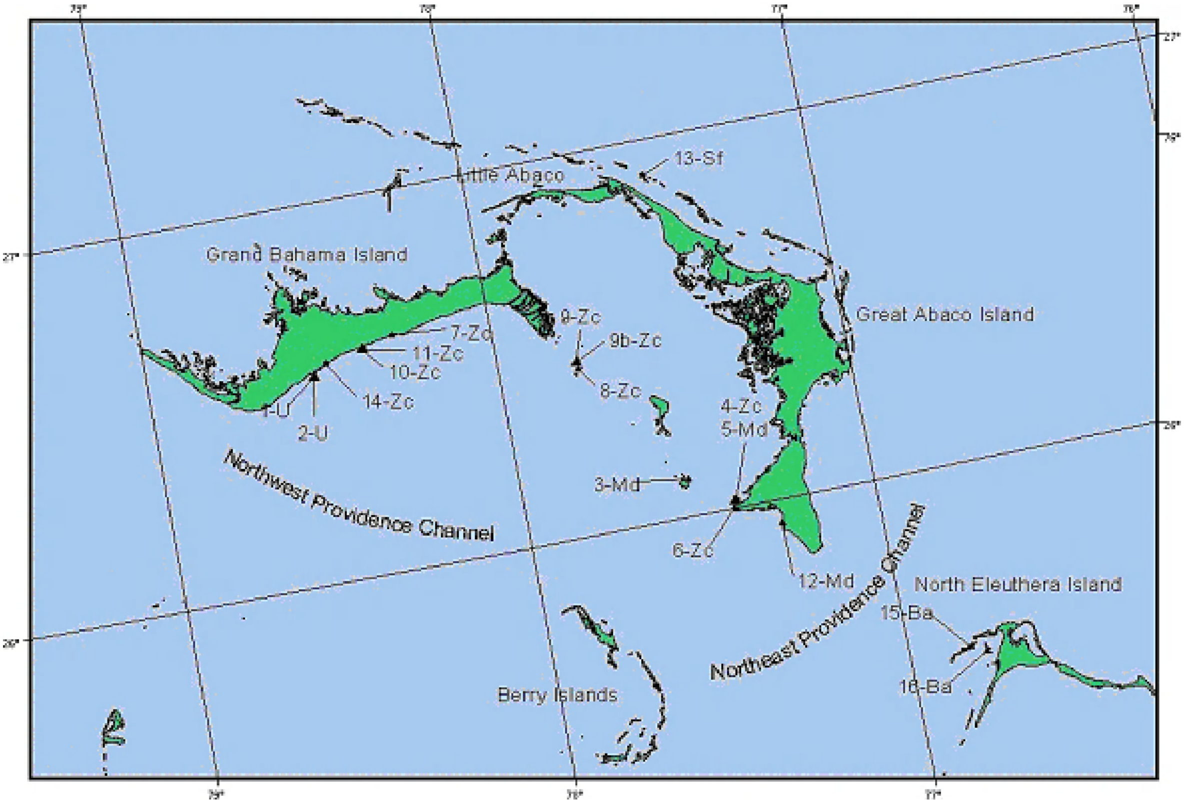

Figure 6.4: *Locations of seventeen marine mammals that were stranded following anti-submarine exercises in the Northeast and Northwest Providence Channels, Bahamas Islands, on 14-16 March 2000. Initials indicate scientific names, numbers show specimen number. Zc = Cuvier's beaked whale Ziphius cavirostris, Md = Blainville's beaked whale Mesoplodon densirostris, Sf = Atlantic spotted dolphin Stenella frontalis, U = unidentified ziiphid, Ba = unidentified baleen whale.*

The association of mid frequency sonar with this, the Madeira, and the Canary Islands strandings suggests that it was not the low frequency component of the NATO sonar that triggered the stranding in Greece in 1996, but rather the mid frequency component.

6.2.8 Case study: Canary Islands

Mass strandings involving beaked whales had repeatedly coincided with the proximity of military manoeuvres from 1988 to 1991 in the Canary Islands (Vonk & Martin 1989; Simmonds & Lopez-Jurado 1991); however no data regarding the nature of the military activity and the possible use of active sonar that was taking place are available.

On 24 September 2002, fourteen beaked whales were stranded on Fuerteventura and Lanzarote Islands in the Canary Islands, close to the site of, and at the same time as, an international naval exercise code-named Neo-Tapon 2002. Strandings began about 4 hours after the onset of the use of mid-frequency sonar activity. Eight Cuvier's beaked whales, one Blainville's beaked whale and one Gervais' beaked whale *Mesoplodon europaeus* were necropsied and studied histopathologically. A study of the lesions of these beaked whales provided evidence of the possible relationship between the sonar activities and the deaths of the whales. Macroscopically, whales had severe, diffuse congestion and haemorrhage especially around the acoustic tissues in the jaw, ears, brain, and kidneys. Fat emboli and lesions consistent with *in vivo* bubble formation were observed in vessels and parenchyma of vital organs (Jepson *et al.* 2003; Fernández *et al.* 2004, 2005). This *in vivo* bubble formation associated with sonar exposure may have been caused by modified diving behaviour (in response to sonar) driving nitrogen super-saturation in excess of a threshold value normally tolerated by the tissues (as occurs in decompression sickness). Alternatively, a physical effect of sonar on *in vivo* bubble precursors (gas nuclei), the activation level of which may be lessened by nitrogen gas super- saturation of the tissues may explain the phenomenon (*e.g.* Crum & Mao 1996). Exclusively or in combination, these mechanisms might initiate, augment and maintain bubble growth or initiate the embolic process. Severely injured whales died or became stranded and died due to a more severe cardiovascular collapse during beaching. Martín *et al.* (2004) describe this incident as well as eight other cases that have occurred in the Canary Islands (see Table 6.1).

6.2.9 Other cetaceans and sonar

As can be seen from Table 6.1, a number of other species have stranded coincident with strandings of beaked whales. These include dolphins (striped *Stenella coeruleoalba*, Atlantic spotted), baleen whales (minke) and two pygmy sperm whales *Kogia breviceps* (these latter are also deep diving species). If these other strandings are linked to those of the beaked whales, the mechanisms are not known.

6.2.10 Research on Mid frequency sonar and cetaceans

An international experiment to measure the behavioural responses of beaked whales and other deep-diving odontocetes took place in the Bahamas during August and September 2007 and 2008.

Data were collected from 16 tag deployments on 4 species, 7 on Blainville's beaked whales, 6 on short-finned pilot whales *Globicephala macrorhynchus*, 2 on false killer whales *Pseudorca crassidens* and 1 on melon-headed whales *Peponocephala electra*. The data collected by the tags included sounds produced by the tagged animal, environmental and anthropogenic sounds received by the animal, details of the animals movements in terms of its diving, swimming speeds, changes in orientation and swimming actions.

Playbacks of sonar signals, killer whale calls and/or pseudorandom noise (PRN) were performed on 2 Blainville's beaked whale and 4 pilot whales, 2 false killer whales and 1 melon-headed whale. Exposures to sounds lasted 4-10 min and were terminated after a programmed time or when changes in behaviour were observed. The sound levels received by the whales varied from ~95 dB (barely audible) to ~155 dB which is well below any level likely to cause temporary physical effects. These playback experiments demonstrated that these methods could safely be used with these species to generate very detailed information on received exposures and behavioural responses. Such experiments are expensive, particularly to gather a large sample size and ideally need an array of bottom-mounted hydrophones to reduce the risk of adversely affecting whales during playbacks.

Data analyses have focused on the playback to the Blainville's beaked whale but some general observation can also be made about the reactions of the other species. When compared with the responses of control animals the playbacks to beaked whales resulted in the following statistically significant effects:

- Reduced duration of foraging during deep dives;
- Reduced rates of ascent to the surface from deep dives;
- Avoidance of the sounds source or the region from which the sound originated;

- Directed swimming away from the region of the playback for several hours following the playback.

Responses to the sonar sound, the killer whale sound and to the PRN were qualitatively similar.

The responses of pilot whales, false killer whales and melon-headed whales varied from nil to mild with some changes of direction of swimming and changing patterns of vocalization that could be attributable to the playbacks.

These results have to be interpreted with care because the analyses stem from a small number of experiments, although they are established against a background of increasing knowledge of the normal, undisturbed, behaviour of these species. Overall, the results suggest that beaked whales are particularly sensitive to any unusual sounds within their vicinity and this sensitivity is apparent at well below levels that would cause physical harm. The results of these studies help to establish a link between sonar sound and beaked whale strandings because they give weight to a mechanism involving panic flight of whales from a sound source. Scenarios where this flight could be exacerbated by bathymetry, coastal topography or movement of the sound source may account for the relationship between strandings and sonar exercises. However, the responses where not specific to the sound signal so this points to a general effect on these species of sound within the 2-4 kHz band.

A scientific consortium including the Norwegian Defence Research Establishment, Norwegian Institute of Marine Research, The Netherlands Organisation for Applied Research (TNO), Sea Mammal Research Unit (UK) and Woods Hole Oceanographic Institution (USA), carried out behavioural response studies in Norwegian waters in 2006 and 2008 to investigate effects of naval sonars on fish and cetaceans. This research was co-funded by Norwegian, Netherlands and US Naval authorities.

Killer whales, long finned pilot whales and sperm whales were instrumented with acoustic and motion sensor tags (dtags, Johnson and Tyack 2003). These recorded the movements of the animals as well as vocal behaviour and exposure levels, before, during and after exposure to signals from an operational towed sonar system on board a research vessel.

The objective was to compare responses to different sonar signal frequencies within the mid frequency band (1-8 kHz), to investigate possible frequency dependency in responses, and relate such dependency to differences in hearing sensitivity of the animals. Studies on other cetacean species than beaked whales allowed examination of the validity of the current understanding that beaked whales are particularly sensitive and, if so, why they were.

During field trials in 2006 and 2008 a total of 26 successful tag deployments resulted in 96 hours of tag recording and 16 exposure experiments on the target species. Preliminary results indicate that killer whales, pilot whales and sperm whales have significantly higher thresholds for avoidance and other behavioural responses such as silencing and abortion of feeding behaviour than beaked whales, but such responses do also seem to occur in these species. Some preliminary results have been reported (Kvadsheim *et al.* 2007). Further studies are planned for 2009.

6.3 Effects of sonar on fish

There have been few studies of the effects of sonar on fish. Ross *et al.* (1995, 1996) reported the use of high frequency sound to deter alewives from entering power station inlets. The alewife *Alosa pseudoharengus* is from the shad family (Alosinae) which have been reported to be capable of detecting sounds at ultrasonic frequencies (Mann *et al.* 2001). Turnpenny *et al.* (1994) reviewed the risks to marine life, including fish, of high intensity, low frequency sonar sources. Their review focused on the effects of pure tones (sine waves) at frequencies between 50-1000 Hz. Johnson (2001) evaluated the potential for environmental impacts of employing the SURTASS LFA sonar system (that is not used in the OSPAR area). While focusing on the potential effects on whales, the study did consider the potential effects on fish, including bony fish and sharks. It considered that the swim bladders of most fish are too small to resonate at low frequencies and that only large pelagic species such as tunas were considered to have swim bladders large enough to resonate in the low frequency range. This conclusion, however, overlooks the findings of Sand & Hawkins (1973) which examined the resonance frequencies of Atlantic cod *Gadus morhua* swim bladders and revealed resonance frequencies from 2kHz down to 100Hz. Many fish species are likely to show resonance frequencies in the range of 100-500Hz (*i.e.* below the frequencies used in the OSPAR area).

Jørgensen *et al.* (2005) carried out experiments examining the effects of mid-frequency (1 to 6.5 kHz) sound at 150 - 190 dB re 1 µPa at 1 m on survival, development and behaviour of fish larvae and juveniles. Experiments were conducted on the larvae and juveniles of Atlantic herring *Clupea harengus*, Atlantic cod, saithe *Pollachius virens* and spotted wolfish *Anarhichas minor*. Swim bladder resonance experiments were attempted on juvenile Atlantic herring, saithe and Atlantic cod.

Sound exposure was performed at 1.5 kHz, 4 kHz and 6.5 kHz with sound simulating naval sonar-signals. These experiments did not cause any significant direct mortality among the fish larvae or juveniles exposed, except in two (of a total of 42) experiments repeated on juvenile herring where significant mortality (20 - 30%) was observed. Among fish kept in tanks one to four weeks after sound exposure, no significant differences in mortality or growth related parameters (length, weight and condition) between exposed groups and control groups were observed. Some incidents of behavioural reactions were observed during or after the sound exposure (‘panic’ swimming or confused and irregular swimming behaviour). The latter seemed to be related to exposure of Atlantic herring juveniles to the lowest frequency (1.5 kHz) or close to the assumed frequency interval of air space resonance in the swim bladder. Histological studies of organs and tissues from selected Atlantic herring experiments did not reveal obvious differences between control and exposed groups, neither did scanning electro-micrograph studies of neuromast organs of young Atlantic herring larvae.

The work of Jørgensen *et al.* (2005) was used in a study by Kvadsheim & Sevaldsen (2005) to examine the possible ‘worse case’ scenario of sonar use over a spawning ground. This found that normal sonar operations will affect less than 0.06% of the total stock of a juvenile fish of a species, which constitutes less than 1% of natural daily mortality. The effect of a sonar exercise on juvenile fish is thus considered negligible. However, these authors did find that the use of continuous-wave transmissions within the frequency band corresponding to swim bladder resonance will escalate this impact by an order of magnitude. The authors therefore suggested that modest restrictions on the use of continuous-wave transmissions at specific frequencies in areas and at time periods when there are high densities of Atlantic herring present would be appropriate.

Doksæter *et al.* 2009 investigated the behavioural reactions of adult overwintering herring *Clupea harengus* to sonar signals in two frequency ranges; 1 – 2 and 6 – 7 kHz and to playback of killer whale feeding sounds. The behaviour of free ranging herring was monitored by upward-looking echosounders as a vessel towing an operational naval sonar source approached and passed over them.

No significant escape reactions, either vertically or horizontally, were detected in response to sonar transmissions. However, killer whale feeding sounds induced vertical and horizontal movements of herring. The results indicate that mid-frequency sonar do not have a significant negative influence on adult herring nor on the herring fishery. The avoidance during playback of killer whale sounds demonstrates the nature of an avoidance reaction and the ability of the experimental design to reveal it.

Similar experiments were also carried out on feeding herring in 2008 in the Norwegian Sea. Preliminary results from this study also confirm the impression that adult herring are not significantly influenced by mid-frequency (1-8 kHz) naval sonar signals.

6.4 Effects of sonar on other species

There have been no studies of the effects of sonar on other groups of animals.

6.5 Mitigation measures for cetaceans

6.5.1 Introduction

As described above, research undertaken so far indicates that one major effect on cetaceans from sonar comes from high intensity mid frequency military sources. This section therefore focuses on this usage, though the principles may be extended more widely to other applications that have not been given as much attention, such as non-military mid frequency sonars etc.

The aim of mitigation is to control and minimise environmental impact, and comprises control of noise at source, mitigation by use of engineering and other methods, and monitoring. The most extreme form of mitigation is to avoid carrying out the activity. In the case of sonar, simulators are already used for training but this cannot remove completely the need for at-sea training. It can be assumed that sonar will continue to be used at sea for defence purposes.

In order for mitigation to be considered, it is necessary to know;

a. the species that might be present,

b. their sensitivity to the noise and hence the area that might be affected;

c. the population density, such that the number of individuals that might be in this affected area can be calculated, and

d. the significance of the effect, or the risk of that effect, on those individuals or their stock.

In order to limit environmental consequences to acceptable levels, use must be made of suitable mitigation measures. However, as military sonar will continue to be used at sea for national defence purposes, in some cases societal/political decisions might overrule the use of suitable mitigation measures to reduce the environmental consequences. Examples where the effects of noise might not be acceptable include:

a. where species are displaced away from a significant proportion of their feeding grounds;

b. where the species are endangered, and management is required to apply particularly risk-averse measures;

c. where the noise is in confined waters, on a migratory route, and is of sufficient duration that a significant proportion of a migratory period would be blocked;

d. where the effect of the noise on marine mammals itself has an economic impact, for instance if whales were displaced from a whale watching area.

Noise can cause effects of no environmental significance. For instance, a behavioural effect in which cetaceans are simply displaced from the area of the sonar operation to another area of similar habitat for a limited period may well be unimportant. On the other hand, for example, persistent or repeated disturbance from an area of biological importance (*e.g.* essential feeding ground, breeding area) may have important consequences.

From first principles, there are three obvious mitigation possibilities, a) limit overall use (total power output), b) limit area of use and c) limit season of use. It is assumed that it is unlikely that design source level could be reduced as these powers are needed for operational reasons. Limits on overall use would reduce risk to cetaceans, while limiting the area of use away from those known or thought to be important to beaked whales may be the most efficient way of reducing risk to the cetacean group that seems most at risk. The difficulty with this is that our knowledge of beaked whale biology and habitat needs is still fairly rudimentary and this species is comparatively difficult to detect in the wild. Acoustic detection, however, might present a way forward: The calls and echolocation clicks of several species of beaked whales have been recorded on several occasions and passive acoustic monitoring devices are used quite successfully in some cases (*e.g.* Manghi *et al.* 1999, Frantzis *et al.* 2002; Aguilar de Soto *et al.* 2004; Johnson *et al.* 2004; Tyack *et al.* 2006; Arias *et al.* 2008).

6.5.2 Control at source

Mitigation can take the form of reducing the total amount of sound produced, possibly by reducing power, duration and/or by reducing the number of times a system transmits sound. Where the species of concern has a well-defined hearing sensitivity, it may be possible to operate at frequencies where the animal's hearing is relatively insensitive. The characteristic(s) of the mid frequency sonar that causes problems for beaked whales are not known – determination of the characteristic(s) and of its precise effect on beaked whales might help in enabling a sonar to be designed that does not affect beaked whales.

6.5.3 Mitigation of death and injury caused by the direct effects of sound

The range at which death or injury due to the direct effect of sound levels (as opposed to behavioural alteration that may lead indirectly to death, discussed in section 6.2.4) can occur is limited. Hence the likelihood of a marine mammal straying into the area prior to the commencement of a sonar transmission is relatively low unless there is a large degree of overlap between important or critical beaked whale habitat and areas of sonar usage. Since the range of the effect is small, there are several mitigation measures that might be effective in preventing injury through the direct effects of sound. A first mitigation measure might therefore be to avoid areas of known beaked whale abundance. Second, it might be possible to regulate the use of sound if marine mammals are detected close to the source. Such detection could occur in two main ways:

Marine Mammal Observers (MMOs)

MMOs are trained observers who aim to visually detect and identify marine mammals, at distances of up to 500 m during daylight hours. Use of MMOs is mandatory on UK, German, and Norwegian naval ships operating active sonar. It may be possible to watch for whales prior to commencing sonar operation and not start transmitting sound if whales are seen or to cease operations if whales enter the area during transmission. However, beaked whales in particular are very difficult to detect and spend a long time under water; in addition the approach does not work in poor visibility, at night or in higher sea states. The efficiency of this mitigation measure is low (for beaked whales) under many conditions likely to be encountered in mid frequency sonar operations (Barlow and Gisiner 2006).

Passive Acoustic Monitoring (PAM) or Active Acoustic Monitoring (AAM)

Both passive and active acoustic monitoring may be used to detect marine mammals. Passive acoustic monitoring is the term used for listening passively to sources of sound, while active acoustic monitoring is the term used for producing sounds and listening for echoes from nearby objects. Both have their advantages and disadvantages. Passive monitoring relies on animals to produce sound (and for those sounds to be recognised) and thus is not reliable for all species at all times. Active acoustic monitoring can detect non-vocalizing animals such as marine mammals or fish, although often only at closer ranges than passive monitoring. Active acoustic monitoring can estimate the range of targets more easily than can passive monitoring. Active acoustic monitoring is relatively undeveloped compared to passive acoustic monitoring for detecting marine mammals and it is an additional source of noise in the marine environment. Both systems might be installed on remotely operated or autonomous vehicles or from buoys to provide a sweep of a wider area or a longer time period than would be possible from one ship at one time.

Passive or active acoustic monitoring offers one way that a wider area might be surveyed for beaked whales. If the lethal effects observed in beaked whales are due to behavioural alteration caused by sound and not to the direct effects of the sound, then such wider area surveys are needed if sonar deployment is to be avoided near beaked whales. This though remains challenging to accomplish despite continuing development of suitable technology.

6.5.4 Other control methods

Two other measures can be taken that would reduce the risk of exposure of marine mammals to loud sound (though as noted earlier, not necessarily risk to behavioural change):

Scheduling

Sonar transmissions may be timed for periods when the species are not in the area, for instance by avoiding migratory periods or periods where local breeding or calf-rearing grounds are used. However, as noted in earlier sections, this information is largely absent for beaked whales, so it is difficult to apply this measure without further research on the use that beaked whales make of certain areas of the sea.

Warning signals

The National Research Council (1994) advocated the development of ‘warning signals’ for marine mammals – sounds that would make marine mammals move away from dangers such as explosions, fast ships, or intense sound sources such as sonars. There has been little development and testing of warning signals, but Nowacek *et al.* (2004) demonstrated that even though right whales do not respond to vessel noise, they do show strong responses to signals designed to alert them. In the absence of information on what sounds cause avoidance reactions, regulators have required some intense sound sources to be increased in level slowly. In principle, such a “soft start” might offer animals a chance to move out of the danger zone, but this seemingly reasonable technique is unproven. Soft start should be viewed as a type of warning signal, one selected because the sound source is already there, not because it is necessarily effective. In most cases, it is more likely that warning signals specially designed to elicit the appropriate avoidance safely would be more successful than soft start. Since it is not known what levels of sonar sounds are safe for beaked whales, warning signals other than sonar sounds would likely pose less risk as well. Nothing is known about behaviours at lower sonar power levels, or in response to sounds other than mid-frequency sonar. In other situations (*e.g.* salmon farms), noise is used to deter marine mammals and it might be that suitable noises exist that could achieve this for beaked whales. There may be value in studying sounds that might elicit avoidance responses in beaked whales that do not pose the risks of sonars. However, the little data available for beaked whales suggests that these are sensitive to a range of low-level sounds (e.g. Aguilar Soto *et al* 2006) and thus alarm signals may pose a risk that would need to be evaluated.

6.5.5 Monitoring

It is plain that much still needs to be learned about the interaction of marine mammals and sonar. Knowledge can be gained and potential mitigation measures identified through good observation and monitoring. Monitoring can include:

Noise monitoring

Anthropogenic noise levels may usefully be recorded in order to be matched against any behavioural reactions by cetaceans. Such recordings also enable the sonar to be ranked against other local sources of noise.

Marine mammal observation

The monitoring of local cetaceans would help to confirm whether there is any obvious effect of the noise. Monitoring the distribution of individuals around the noise source can be by tagging, by using passive acoustic monitoring to detect vocalisation, or by using active acoustic monitoring.

The latter monitoring strategies may serve two purposes, either of demonstrating that there is an effect, or, if an effect is observed, of identifying the level at which it occurs. While it may be argued that the monitoring itself has an effect on the species, this effect may be outweighed by the process providing information which may be used in the longer term to conserve stocks of the species. It should be noted that no monitoring programme can demonstrate that there is no effect, for the range of potential effects is large, and many effects would be too subtle for a generic monitoring programme to detect. A more scientific approach would test for specific hypotheses about effects, with experiments designed with strong statistical power.

6.5.6 Mitigation measures currently in use for military sonars with regard to marine mammals

Guidelines for sonar research testing by NATO and marine mammal risk mitigation research at the NATO Undersea Research Centre

Carron (2004) described the NATO Saclantcen (now called the NATO Undersea Research Centre) marine mammal mitigation programme that was developed following the Greek incident (see Section 6.2.6). The goal of the mitigation programme is to develop a predictive tool for the presence of cetaceans and to develop on-site acoustic risk mitigation rules, procedures and tools. These goals are being met through the collection of cetacean presence data along with relevant hydrographic information. The programme has developed habitat models for Mediterranean Sea species and has developed a prototype Environmental Scoping Tool Kit for use by planners. The programme has, in cooperation with research groups from many nations, performed controlled exposure experiments on sperm whales as a proxy for beaked whales. Data from 5 years of at-sea experiments have been collected by trained visual observers, passive and active sonar and the use of acoustic and non-acoustic sensors attached to the whales (in cooperation with the Woods Hole Oceanographic Institution). Information about these experiments can be found at http://solmar.saclantc.nato.int. The programme has also developed, in cooperation with Mediterranean nations, a Sightings and Strandings database. The programme has played an important role in the analysis of beaked whale vocalisations and has developed a series of passive sensors tuned to listen for these vocalisations (update on the programme can be found under: http://www.nurc.nato.int/research/index.htm).

The risk mitigation policy has several stages. The first requires a scoping study that determines the possible negative effects of sonar operations on the environment. It then establishes an exposure level above which risk mitigation must be applied. This exposure level is likely to vary geographically depending which species are present. One method of reducing risk will derive from the predictive tool referred to above, hopefully enabling planners to choose the times and localities where there will be fewest marine mammals. The second method is the use of observers along with passive acoustic monitoring in the area of any sonar use. If, during the hour prior to tests starting, marine mammals are detected in the area, the test does not commence. If marine mammals are detected during the test, the test is immediately suspended. The amount of noise produced by the sonar is also progressively increased prior to a full-scale test in order to give mammals a chance to move away. It should be noted that these rules apply only to tests conducted by the NATO Undersea Research Centre and do not necessarily apply to use by individual NATO naval vessels outside those tests. NATO has developed rules for use by operational forces acting under the NATO command to minimise risk to marine mammals during noisy operations.

Mitigation on UK naval vessels or in UK sonar tests

The UK Ministry of Defence (MoD) has a policy that any activity which may have a potentially harmful impact on the environment requires mitigating measures to reduce any adverse effects. To this end, detailed command guidance has been issued to ships, submarines, antisubmarine warfare air squadrons, and the survey and mine warfare units. The management of planning underwater acoustic operations is taught during professional training, as are the implementation of mitigation measures and the use of the bespoke Environmental Risk Management System (ERMS) known as S2117 and global environmental impact assessments (EIAs). To supplement formal training ashore, computer-based training via the MOD intranet is accessible to all individuals involved in managing the impact of Royal Naval (RN) acoustic operations from command to Marine Mammal Observers. Formal assessment of the organisation and application of the command guidance is provided during units' operational sea training.

The command guidance applies worldwide outside harbours to all Royal Naval ships, submarines and aircraft operating in-service active sonars. The system covers three phases: planning, monitoring and taking action. During UK sponsored exercises all participants are expected to apply UK mitigation rules as directed by exercise planning staff and further guidance is provided within exercise planning orders.

Planning

When planning active sonar operations, the ERMS (essentially an intelligent database of Sonar parametric, hydrographic, climatological, legislative and biological data) calculates the risk of potential adverse effects on marine mammals within an area where sonar may be operated by the Royal Navy. This includes relevant statutorily protected marine areas and more general materials are provided for other parts of the world. Standard guidance requires that activities and tracks are not planned where there is a likelihood of embayment, across known migratory routes or in breeding areas

Monitoring

The guidelines recognise that military units are well placed to provide situational awareness above and below the water using visual, electro-optic and acoustic sensors. They demand that prior to transmitting sonar, every opportunity should be taken to monitor the local area and observe for cetaceans/marine mammals. This includes both visual monitoring (both from the sonar source ship or aircraft, and from any other nearby ships or aircraft), and acoustic monitoring using standard passive sonars or using bespoke marine mammal acoustic detection equipment. Recording such observations is deemed as essential to continuous improvement of the worldwide data base and forms are provided to note down any marine mammals (or other obvious marine life) seen.

Taking action

The ERMS provides the command with the required information to make decisions on the configuration of sonar assets and the mitigation measures that would be most effective. During the conduct of acoustic operations, standard mitigation action such as the deployment of RN Marine Mammal Observers (MMOs) and a 'soft start' or 'ramp up' procedure is employed. Ramp up permits the gradual increase in source level up to the operational setting to reduce the impact such a source may have if an undetected marine mammal is in the area. RN MMOs are posted for either 1 hour (in water depths greater than 200 m) or 30 minutes (in water depths less than 200 m) in order to monitor and record the presence of marine mammals. The guidance requires that if marine mammals are encountered during the conduct of active sonar operations then further action should be taken, where safe to do so, to minimise any potential disturbance. ERMS and the Command Guidance provide a number of options for mitigation of the calculated risk, and are designed to recommend the most effective option. This recommendation will take account of sonar operating parameters, operational limitations and other real-time inputs to calculate the level and type of environmental risk involved in using active sonar at any given time and location.

Possible recommended courses of action include:

a. cease or modify transmissions - where operationally practical and safe to do so, the operating parameters of sonar or other acoustic device should be adjusted or turned off to avoid disturbance;

b. reduce power - if the option of turning off the sound source is not operationally practical, then the sonar should be operated at a reduced power setting, or different configuration commensurate with the scale of operation;

c. modify the number and or source levels/configurations of associated units temporally or spatially to minimise the potential for adverse effects.

Application of mitigation zones

The UK Royal Navy use a number of key boundary definitions to establish the likely risk to marine mammals from acoustic operations. ERMS displays these to the command both during planning and throughout acoustic operations. A stand off range is calculated using the inputs from a propagation loss model. These key boundary definitions are as follows:

a. Mitigation Action Zone (MAZ). A fixed monitoring range shown on ERMS based upon the observation position, which will require monitoring to a distance of 2000 yards. The 2000 yard range for which the surveillance effort can, under normal conditions, be reasonably expected to detect the presence of marine mammal activity and therefore initiate appropriate mitigation action.

b. Permanent Threshold Shift (PTS). PTS is irreversible physiological damage caused by rupture of the hair cells of the inner ear, resulting in a permanent loss of hearing sensitivity. The consequence of this is a permanent shift in the threshold of hearing of the receptor.

c. Stand Off Range (SOR). The maximum ranges at which a received sound pressure level (SPL) exceeds the relevant threshold appropriate to a given acoustic impact, is referred to as a stand-off range.

d. Temporary Threshold Shift (TTS). TTS is a temporary loss in the efficiency of the mechanical-chemical electrical transfer function in the inner ear, resulting in a temporary loss of hearing sensitivity. The consequence of this is a temporary shift in the threshold of hearing of the receptor.

UK research on passive detection of marine mammals

The UK MoD continues to develop a system that will integrate with in-service sonar to passively detect and classify animal sounds. The system is at sea as a technology demonstrator and information from this trial period will inform the formal procurement process. Further work is continuing to research behavioural responses of marine mammals to underwater noise, to aid the classification of marine mammal vocalisation, and the active detection of marine mammals using current in-service sonars

Mitigation on Norwegian naval vessels

The Royal Norwegian Navy has implemented guidelines for the use of active sonar in Norwegian water since January 2006. These guidelines apply to all military units which employ active sonar transmitting in the mid-frequency band (1 - 10 kHz), with source levels exceeding 200 dB (rms re 1µPa @ 1 m) during sonar exercises in Norwegian waters. The guidelines are up-dated every year as the scientific community gains more knowledge about the potential impact of naval sonars on marine life. The guidelines are accompanied by annual versions of a decision aid tool (SONATE) which supports naval planners and operational commanders in complying with the guidelines (Nordlund and Benders 2008).

SONATE is a decision aid tool for use during planning and execution of sonar exercises in Norwegian waters that aims to minimise conflicts with activities such as fisheries, fish farms, tourism etc and effects on marine life (Nordlund and Benders 2008). SONATE contains information on the distribution of fish and marine mammals, fishing activity and fish farms. It also contains the sonar guidelines that define areas to avoid intensive and routine sonar exercises, information on critical frequency bands and on sonar start-up procedures.

During planning of exercises, planners should aim to avoid areas expected to have a high presence of marine mammals, particularly beaked whales. As part of this consideration the intensity of sonar transmission (duration of exercise and numbers transmitting units) must be considered.

During execution of exercises in areas where marine mammals are expected to be encountered, special procedures for sonar transmission should be used. These procedures aim at mitigating risk of injury to marine mammals by establishing a 200 dB (rms re 1 µPa) isobar safety zone. These procedures include visual and acoustic watch before and during transmission, procedures for start of transmission and special consideration during operations in constricted waters and at high speeds. Procedures for start of transmission include reduction in sailing speed and visual or acoustic examination for marine mammals within safety zone, or initiation of transmission at reduced ping energy (reduced ping duration or source level).

Mitigation and research by the Royal Netherlands Navy (RNLN)

The Netherlands Ministry of Defence (NL MoD) considers active sonar to be an essential capability. According to NL MoD environmental policy, a balance between operational requirements and environmental protection is always required. In line with the precautionary principle, NL MoD have a research programme examining the environmental effects of sonar, and mitigation measures are being used by the Royal Netherlands Navy.

Rules for environmental protection (sonar) are included as a standing part of the Operation Order issued by Commander NL Maritime Forces for RNLN activities. Attention is also given to human divers. Mitigation measures include:

a. Determining (in advance) if sensitive areas can be identified in planned operations area;

b. Monitoring (visual and passive acoustics) prior and during sonar activity;

c. Avoiding embayments or forming barriers between deep water and shallow water;

d. No sonar transmissions are to commence if mammals are observed (visual or audio) within a calculated safety distance;

e. If feasible, commencing transmissions at low power;

f. Reducing power or ceasing transmissions if animals are observed with safety range;

g. Identify if diving activity, or possible diving areas are present in planned operations area;

h. No transmissions inside or within safety range of diving areas;

i. In vicinity of land/diving areas, use of low power if feasible;

j. Logging of information.

The Netherlands research programme on environmental effects of sonars aims to secure long-term environmentally-responsible use of sonar. The research programme includes:

a. Development of a risk assessment tool for use by planners and operators (SAKAMATA);

b. Biological research, both auditory studies on harbour porpoises and behavioural response studies at seas, in cooperation with international partners;

c. Development of equipment for detection, localisation and classification of marine mammals.

In addition to this, an environmental impact assessment will be carried when introducing new active mid-frequency sonar systems. The first operational version of the SAKAMATA tool will be introduced in the RNLN in mid-2009 and includes an updated set of mitigation rules to be used by RNLN.

Mitigation by the Spanish Ministry of Defence

Since 2004, the Spanish Ministry of Defence has maintained a moratorium on the use of sonar at less than 50 nm from the Canary Islands. This followed a series of mass strandings on the islands (see Section 6.2.8); no further mass strandings have occurred since 2004 in the archipelago.

Mitigation on Australian naval vessels

A full description of the Australian system of reducing risk to marine mammals (particularly larger whales) has recently been finalised (Polglaze 2005, J. Polglaze, *pers. comm.*). The elements of planning, monitoring and taking action outlined above are also present in the Australian system. The objectives of the first risk-reduction stages are achieved by the use of Planning Guides and Planning Handbooks. These are augmented by Procedure cards focused at the operational level of activities. Monitoring and action procedures have been codified onto a series of easily accessible procedure cards, each of which covers a ‘family’ of sonar equipment. Families of equipment include:

Hull-mounted anti-submarine sonar

Active towed arrays

Active sonobuoys

Minehunting sonar

Acoustic decoys

Hydrographic survey sonar

Mine and obstacle avoidance sonar

Most of the procedures are visually- based, and ships can augment this with opto-electronic and/or infrared sensors. Radar is also an option but is considered of limited utility in most circumstances. Passive acoustic equipment is considered not reliable enough and cumbersome in use for surface ships, but is used in submarines that are much better equipped in this regard. In addition, submarine sonar operators are better trained in the use and interpretation of passive acoustics.

The core objective of the Australian approach is to detect all whales within 4000 yards; whales detected within this range (based roughly on a likely received levels of 160 dB re

1 μPa) lead to systems not being started up, or to the system being turned off/decreased in power, or the ship altering course to increase the distance to the whales. For those systems where sound can be directed, guidance is given to blank out sectors containing whales at a range of 2000 to 4000 yards, with a requirement to turn the system off under this range. The stand-off range of 4000 yards is based upon conservative modelling using credible ‘worst-case’ scenarios. Lesser stand-off ranges are mandated for systems with lower radiated power output.

Mitigation on Italian naval vessels

The Italian Navy has rules to control sonar use in all seas around Italy and when operating elsewhere (Cerutti 2005). These rules take account of the NURC guidelines (see Section 6.5.6). Several rules apply to the planning phase of any sonar use. Sonar use should i) avoid areas with steep (>5%) slopes and a bottom depth of 1000 m to 2000 m; ii) avoid acoustic trapping situations (*e.g.* canyons, bays, gulfs) and iii) keep 5000 m outside the boundaries of ‘important’ areas (*e.g.* whale sanctuaries) iv) standard conservative stand-off ranges (based on disturbance and damage receiving levels, including some relevant margins) apply for hull mounted and dipping sonars. A database, aimed to help populate NURC’s database, is being developed to help ships know where such areas are.

Prior to the start of sonar use, a search of 30 minutes using both visual and passive acoustic methods is conducted, with the sonar system not being started if marine mammals are detected within 1500 m. If no marine mammals are detected, then the system is brought slowly up (ramp-up) to full-power over 15 - 30 minutes. If the sonar is turned off for whatever reason, these procedures are repeated if the system is not restarted within 30 minutes.

The Italian Navy also has procedures for use with sonobuoys and helicopter-mounted sonars. These are similar to those used with vessel-mounted systems, but with considerably truncated time periods. The search period is reduced to 2 minutes, with a 5-minute ramp-up period to full power.

Mitigation on US naval vessels during training and during exercises

The US Navy has adopted a series of policies similar to those outlined above (Stone 2005). There are variations in these policies depending on the type and location of the proposed activity. For activities where there is a choice of location, pre-planning requires consideration of deepwater areas with no significant bathymetric features that form preferred habitat for marine mammals. Areas with known critical habitat for marine mammals should be avoided (*e.g.* NW Providence Channel and Puget Sound). In areas where sonars will be used, a computer-based system will provide information on what cetacean species may be present in an area is available to help plan operations. This system also provides information similar to that given by the Australian card system. Bridge crew are trained as cetacean look-outs as part of a 200 hour training programme. Passive acoustic monitoring systems are deployed where possible.

Other measures required include:

- Post lookouts with binoculars – lookouts survey for presence of protected species before, during, and following the exercise
- Conduct training during daylight when feasible
- Ensure sea state and weather conditions support visual survey capabilities
- Aerial survey when inherent to the exercise

- Exercise area (range) and ships/submarines conduct passive acoustic surveys before/during exercises
- Report stranded, injured, or dead sea turtles or marine mammals

Codes are also available for aircraft (helicopter) use of sonar.

Future mitigation development

Gentry (2004) considered that most current naval operations were conducted with minimal or no mitigation measures in place and he identifies whale-finding sonar as the mitigation measure of the future. These are high-frequency low-power sonars and therefore have a limited detection range (about 2 km). Their acoustic energy is also low. A difficulty is that ships carrying mid-frequency military sonar operate at relatively high speed and therefore detections may occur too late to take any action. This technique looks very promising though – and it might seem logical to examine the possibility of using the tuna finding sonar described in Section 6.1.2. Since 2004 many navies have applied mitigation measures, pending deployment of additional technologies.

No reference has been found to any attempts or proposals to evaluate or mitigate the environmental effects of any non-military sonars.

6.6 Mitigation measures for fish

The body of data currently available on the response of fish to sounds is thin and only the Royal Norwegian Navy has developed mitigation measures. Most species of fish in OSPAR seas are hearing generalists (*e.g.* cod (Chapman and Hawkins 1973); salmon (Hawkins and Johnstone 1978) and are not able to hear sonar signals above 1 kHz unless they are extremely close. Since naval sonar being used in the OSPAR regions is mostly mid-frequency (operates above 1kHz), the potential problem is limited to two issues; direct physical effects and potential behavioural effects on a few species of fish that are hearing specialists able to hear these higher frequencies. Physical effects (injury) have been demonstrated on juvenile fish but only within a few 10s of metres from a sonar source (Jørgensen *et al.* 2005), and this will not have significant impact at the population level (Kvadsheim and Sevaldsen 2005). It seems highly unlikely that adult fish, which are able to swim away from approaching sonar sources, will ever get close enough to a transmitting source to get injured. Clupeid fishes such as herring and sprat are hearing specialist, which will be able to hear mid frequency sonar signals in the 1-5 kHz band (Enger 1967). Juvenile herring has been demonstrated to react strongly to 1 kHz sonar signals (Jørgensen *et al.* 2005), but adult herring does not seem to respond the same way (Doksæter *et al.* 2009).

The Royal Norwegian Navy has implemented mitigation measures to mitigate impact on fish and fisheries. Some of these measures will be subject to revision upon completion of on-going studies on sonar effects on fish.

During the planning of exercises planners should avoid exercises involving transmissions below 5 kHz in spawning areas, areas with large numbers of herring and brisling (small herring), and areas with intense fishing for herring and brisling. As a general precaution, a safety zone of 200 m from all fish farms and all fishing vessel actively involved in fishing are also implemented. In addition some restrictions on transmission of certain waveforms (CW) and frequencies are implemented. This is based on the findings that these signals at frequencies which matches the swim bladder resonance of juvenile herring, has a much higher impact on the fish than other waveforms and frequencies (Jørgensen *et al.* 2005). The specific frequency being restricted will vary in time and space depending on the size of the juvenile herring in the area.

6.7 General conclusions

The full effects of sonar on cetaceans are not well known, mostly due to the difficulty of studying the interaction, and to a lesser extent because details of sonar equipment and usage are not easy to obtain.

There appear to have been no direct consequences of using low or medium intensity high frequency sonar on cetaceans over the period that such navigation and surveying sonars have been in use. Nevertheless, there have been very few studies of the effects of sonar at these frequencies or low intensities. The propagation properties of these frequencies in water will mean that any sphere of influence of a single source is comparatively small. The use of multiple sources in a wider area, such as in a location where many vessels are using navigational sonar will have a greater effect, and the possibility of this affecting the distribution of some cetaceans in these areas cannot be discounted.

The use of high-intensity mid frequency sonar has led to the deaths of a number of cetaceans in some places. From our very limited knowledge, it appears that beaked whales are the most affected species, in particular Cuvier's beaked whale (possibly as this species appears to be the commonest and most widely dispersed beaked whale). A characteristic of most of the known mortality incidents is that they have been on shores near to the shelf break and deep water habitat favoured by these species. It is unclear therefore if further undetected mortality is occurring where these habitats are further offshore. We do not know the precise mechanism causing the animals to beach themselves – many arrive ashore alive, but obviously distressed. It is unknown whether animals that are affected further out to sea can survive and not strand.

The magnitude of the problem involving beaked whales and sonar presently verifiable by science is as follows. We know of about 40 sonar-related deaths among cetaceans (mostly, if not all, beaked whales) over the last 9 years. Read *et al.* (2003) indicates that worldwide, fisheries kill several hundred thousand cetaceans as bycatch each year. We do not know of the scale of beaked whale bycatches but 35 fishery-related beaked whale mortalities were observed in the pelagic drift gillnet fishery off the east coast of the USA between 1989 and 1995 and between 1991 and 1995 the total average estimated annual fishery-related mortality of beaked whales in the U.S. EEZ was 9.7 (CV = 0.08). It is not known how many beaked whales affected by sonar die uncounted at sea, nevertheless it seems likely that the fishery-related mortality of beaked whales alone is several times higher than that caused by sonar.

Many fishery managers and fishers are attempting to address bycatch of cetaceans through research and use of mitigation measures. Some fisheries have been closed due to the lack of suitable known mitigation measures. It is worth noting that initially strandings have often been the only indicator of by-catch as a potential threat (with relatively small numbers compared to estimated population size of cetacean populations of interest). It was often not until after specifically-targeted studies had quantified numbers caught and killed that it was found that bycatch for some cetacean populations was unsustainable. There are still major uncertainties, difficulties and unknowns in relation to estimating impacts of sonar and noise generally on marine mammals. Those using the environment have a responsibility to minimise environmental impact under many international agreements – this applies equally to those using high intensity low- and mid-frequency sonar. Some mitigation measures are possible already and others need further development. The use and development of these measures should be encouraged.

As outlined above, sonars also contribute to the global ocean noise budget and overall levels of noise in the ocean are increasing, at least in some areas. The potential effects of this increase, if communication vital to the life history and reproduction of some cetaceans is badly affected, could be worse than direct killing. It seems likely that if these effects are occurring, the large baleen whales would be the most affected by increases in low frequency noise; many stocks of these whales are already in a threatened or endangered state due to over-hunting in the past. However a reduction in anthropogenic ocean noise is clearly desirable to minimise human-effects on acoustic mediated biological processes in the oceans.

In evaluating the impact of sound upon fish it is important not just to consider the immediate impact upon individuals, but to look at long term effects upon individuals and upon populations. Sounds may deter individual fish from passing through a noisy location, may cause disruptions in their behaviour and may even damage their hearing, but it is when there is long lasting effect upon the whole population and its ability to sustain itself that there is a need for greatest concern. Deflection of migrating fish, displacement of fish from their feeding grounds or disruption of spawning activities, especially when large numbers of fish are affected, may prove especially damaging.

Sonar will probably never exceed a limited proportion of the overall budget of noise in the oceans because it is driven by electricity which is difficult to produce, unlike air pressure (airguns) or the burning of oil (shipping). On the other hand, shipping's contribution to ocean noise has been projected to increase greatly, especially in coastal areas, in the next 20 years (see Hildebrand 2005, for initial assessments of ocean noise budgets).

The scientific assessment of the biological impacts of sonar indicates that 'sonar is not a major current threat to marine mammal populations generally, nor will it ever be likely to form a major part of ocean noise' (ICES 2005a). ICES also noted that sonar can place individual whales at risk, and has affected the local abundance of beaked whales. This latter point has raised concerns especially from non-governmental organisations such as the National Resource Defense Council (USA) and the Whale and Dolphin Conservation Society (UK); see for example Parsons *et al.* (2008). On the other hand, there is much unknown about the precise effects of sonar on individual sea mammals, on ecosystem level and on fish (ICES 2005a and 2005b). Therefore the military (users of the sonar) have a responsibility to act according to the precautionary principle until such time as sufficient knowledge is available. In practice this means mitigating the effects of active sonar as far as possible and funding research into the effects of the use of active sonar in order to reduce the gaps in biological knowledge. Mitigation measures should be reviewed in light of new knowledge regularly until such time that there is reasonable certainty as to whether active sonar is an ecologically significant problem or not.

As sonar deployment seems likely to increase in the future, the need to understand the effects more thoroughly and to research ways of mitigating the effects of sonar is a priority for future research and development. There are several research programmes currently underway that cover the issue and results of these studies shall help shed light on the effects of sonar on marine life.

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Module 7: Seismic Surveys

7.1 Seismic Surveys

Seismic surveys are used for a number of discrete purposes. First and foremost, marine seismic surveys are central to the oil and gas industry, and have contributed substantially to the discovery and definition of new hydrocarbon reservoirs, as well as playing an integral role in defining the extent and directing the depletion of these reservoirs In addition to application to the oil and gas industry, seismic surveys are also used to gather data for academic and governmental needs. An example of this can be found in the delineation of extensions to Exclusive Economic Zones (EEZs), under the United Nations Law of the Sea Convention.

In outline, a seismic or seabed survey involves directing a high energy sound pulse into the sea floor and measuring the pattern of reflected sound waves that are then processed and interpreted, to reveal geological features often several thousand metres beneath the sea bed. A range of sound sources may be used depending, amongst other things, on the depth of penetration required; these include, airguns, 'sparkers', 'boomers', 'pingers' and 'chirp sonar'. The source most often used in geophysical surveying for oil and gas is the seismic airgun. Airguns are generally used in clusters or arrays, and when fired, the airgun releases a bubble of compressed air. During a survey, guns are fired at regular intervals (*e.g.* every 10 to 15 seconds), as the towing source vessel moves ahead. The sound pulse is directed down into the sea bed and the reflected sound is detected by hydrophones mounted inside a "streamer". Streamers are long cable type devices that are towed behind the source vessel. The number and length of streamers and the distance between groups of hydrophones will vary according to the design of the survey but the cables can be 3 to 12 km in length.

Seismic airguns generate low frequency sound pulses below 250 Hz with the strongest energy in the range 10-120 Hz and peak energy between 30 to 50 Hz. Airguns also release low amplitude high frequency sound, and acoustic energy has been measured up to about 100 kHz (Deruiter *et al.* 2006; Goold & Coates, 2006; Bain & Williams 2006; Sodal 1999; Madsen *et al.* 2006b). While the energy of interest (10 to 120 Hz) is mainly focused vertically downwards, the higher frequency components are also radiated in horizontal directions.

The power of airgun arrays has generally increased during the past decades, as greater depths are explored (Barlow & Gentry 2004). The nominal source level (on axis) of an airgun array can reach up to 260-262 dB (p-p) re 1 µPa @ 1m. It is difficult to measure the actual sound pressure level close to a full source array that is being activated, due to the physical environment surrounding an active seismic array. Therefore, assumptions are made that enable the response of a given source array to be modelled.

The 'far field' assumption suggests that at some distance away from a source array, which is much greater than the dimensions of the source array, the peak energy pulses from the various individual source elements ('near field' signature) arrive at the same time and add together constructively to form the 'far field' response of the source. This response is corrected or back-projected to one metre from the source array to produce the 'near-field' signature of the source at one metre, which is a standard, modelled measure of a source array output. It is well known that the peak energy pulses from individual source elements no longer align at locations close to the seismic source array (in the 'near field') as a seismic source array is a 'distributed', rather than a 'point' source. Therefore, the emitted sound pressure level close to the source array is lower than that calculated using the 'far field' calculation (reference, e.g. Urick 1983).[14]

Sound signals from seismic airgun surveys may be received thousands of kilometres away from the source if spread in a sound channel. For example, the recordings of autonomous acoustic seafloor recording systems of the US-NOAA on the central mid-Atlantic Ridge showed year-round recordings of airgun pulses with a dominance in summer from seismic surveys often taking place usually more than 3000 km away (Nieukirk *et al.* 2004).

Other boundary and channelling effects in certain environments have been shown for the Gulf of Mexico, where received levels from first arrivals of airguns signals in a controlled experiment were identical (162 dB (p-p) re 1 µPa or 127 dB re 1 µPa2s) at both 2 km and 12 km distance from the source due to multipath reflection patterns, but significantly lower in-between (Madsen *et al.* 2006b). In some areas, low-frequency energy from seismic sounds may travel long distances through bottom sediments, re-entering the water far from the source (Richardson *et al.* 1995; McCauley & Hughes 2006).

One alternative to using a short and sharp pulse as the signal from seismic airguns is to generate a long tone with changing frequency. This method, called Vibroseis, is routinely used for terrestrial surveys. Although marine vibroseis has been used (primarily for research) in the past, it is not being used commercially at this time. However, given the increased attention to seismic sound in the marine environment, renewed interest in this technique has emerged.

There is a range of designs for seismic airgun surveys. These include 2-D, 3-D and 4-D (3-D but with time as the fourth dimension); multi-azimuth surveys as well as towed or seabed configurations.

In 2-D operations, a single seismic cable or streamer is towed behind the survey vessel, together with a single sound source. Reflections from the subsurface are assumed to lie directly below the sail line that the survey vessel traverses – hence the name '2-D'. 2-D lines are typically acquired several kilometres apart on a relatively broad grid of lines, and usually over a large area. This method is generally used today in frontier exploration areas, to produce a general understanding of the area's geological structure.

A 3-D survey covers a specific area, generally with known geological targets generated by previous 2-D exploration. In 3-D surveying, groups of sail lines (or swaths) are acquired with the same orientation, unlike 2-D where there is typically a requirement for the lines to be shot orthogonal to the dominant structural grain. Simplistically, 3-D acquisition is the acquisition of many 2-D lines closely spaced over the area. The 3-D sail line separation is normally of the order of 400 to 800 metres, depending on the number of streamers deployed and their cross-

[14] Far from the sound source (in the acoustic far-field), there is a constant ratio between the pressure component and the kinetic component of the sound. Closer to the sound source than approximately 1/6 of the wave length (in the acoustic near-field), this ratio increases dramatically as distance decreases. For 10 Hz, which is at the lower end of the frequency range where air guns provide maximum effect, the wavelength is e.g. approx. 150 m, and the near-field extends to approx. 25 m. It is likely that many of the harmful effects observed on organisms close to the sound source are due to particle acceleration, and not sound pressure. However, it is more difficult to measure particle acceleration than sound pressure, and nearly all of the reports on the effects of seismic signals on marine organisms listed the intensity of the sound as sound pressure. Therefore, it is important to be aware that sound with the same sound pressure may be far more harmful at very short distances compared with longer distances (DNV Report No. 2007-0512)."

line separation. By utilising more than one source and many streamers from the survey vessel, the acquisition of many closely spaced, sub-surface 2-D lines, typically between 25 and 50 metres apart, can be achieved by a single sail line. With the number of sail line kilometres involved, 3-D surveys can take many months to complete.

4-D surveys, or so called “Time Lapse” surveys, are 3-D surveys that are repeated over the same geographical area, but at different times. 4-D surveys are being used regularly on established fields to monitor fluid (oil and gas) movement during the field’s production phase.

Multi-azimuth surveys are 3-D surveys containing sail lines that vary in azimuth (direction). In complex geological areas with steeply dipping salt layers, “shadow-zones” are created, which limit the ability of the geophysicist to see deeper features. By acquiring seismic information in different directions, shadow zones can be eliminated, hence providing a much improved image of the subsurface.

Sparkers and boomers are high frequency devices that are generally used to determine shallow features in sediments. These devices may also be towed behind a survey vessel, with their signals penetrating several hundred (sparker) or tens (boomer) of metres of sediments due to their relatively higher frequency spectrum and lower transmitted power. Typical source levels are around 204 - 210 dB (rms) re 1 µPa @ 1 m (CCC 2002). Larter (2004) gives the following specifications for a deep-tow boomer: source level 220 dB (rms) re 1 µPa @ 1 m, frequency band 0.8 - 10 kHz, pulse length 0.2 ms and beam width 20°.

Chirp sonars also produce sound in the upper frequency range of seismic devices (approx. 0.5 to 12 kHz). The peak source level for these devices is about 210 – 230 dB re 1 µPa @ 1 m. Chirp sonars and sediment echo-sounders can be used in a hull-mounted mode.

7.2 Effects on Marine Mammals

There is a substantial volume of research and observational data on the potential impacts of sound from seismic surveys on marine mammals. In general, this research has been conducted to test the generally accepted hypothesis that intense underwater sounds have the potential to induce a range of effects on marine mammals.

There have been a few cases of strandings of beaked whales and giant squids coinciding with academic seismic surveys (Malakoff 2002; Palacios 2004; Guerra *et al.* 2004). However, there is no conclusive evidence of a link between sounds of seismic surveys and mortality of any marine mammals. Moreover, there have been few studies into non-lethal effects. For example, Finneran *et al.* (2005) conducted studies on impacts on hearing of odontocetes (using beluga whales exposed to single pulses of a watergun), and concluded that for this species, Temporary Threshold Shifts (TTS) might occur if animals were exposed to airgun discharges while within 5 metres of the gun. Finneran *et al.* also concluded that there was no reason to believe that air gun discharges could lead to Permanent Threshold Shift (PTS). In a recent experiment, Lucke *et al.* (2008) exposed a harbour porpoise to sounds from a single airgun. They found that at 4 kHz the TTS criterion at received peak levels of 200 dB re 1 µPa and a sound exposure level of 164 dB re 1µPa^2 * s was exceeded. These levels are lower than those reported for other toothed whales so far and indicate much larger zones of TTS around a seismic airgun than for the beluga whale, at least for this individual (Finneran *et al.* 2004).

By far the greatest amount of information on the responses of marine mammals to sounds from seismic surveys focuses on estimates/observations of behavioural responses. The following paragraphs provide only a cross section of the work carried out, but these do show the range of conclusions that have been drawn with respect to behavioural reactions, and therefore the lack of a consensus in the scientific community on the occurrence, scale and significance of such effects.

Sperm whales in the Gulf of Mexico have been exposed to seismic surveys for many years (Wilson *et al* 2006). Visual surveys (Gordon *et al*, 2006) or satellite tagging (Winsor & Mate, 2006) have not been able to detect changes in the animals' behaviour. Preliminary conclusions from recent controlled exposure experiments (CEEs) with tagged animals have suggested possible responses to sound exposure in terms of foraging activities (Miller *et al*, 2006). The same authors found that during exposure, one animal stayed at the surface for an unusual amount of time, and they interpreted that behaviour as potential vertical avoidance (Miller *et al.* 2006). While most whales in their study continued their normal dive patterns, a behavioural response was inferred from the reduced fluke pitch and vocalisation buzz rates during the dive. The animals showed no avoidance behaviour, however, in a 1 - 13 km range from the sound source at received levels of 152 - 162 dB (peak to peak) re 1 µPa (equivalent to 147 dB (rms) re 1 µPa or 115 - 135 dB re 1 uPa^2s). Miller *et al* (2006) suggested that the 'observed' foraging might have been related to behavioural reactions of the sperm whale prey to airgun sound, or may simply be an artefact of the habituation of these animals to seismic surveys.

Several species of baleen whales (grey, bowhead, blue, sei minke and fin whales) showed avoidance behaviour to sound from seismic surveys (Malme *et al.* 1983; 1984; 1985; 1986; 1988; Richardson 1998; Richardson & Malme 1993; Richardson *et al.* 1986; 1995; Brownell 2004; Gordon *et al.* 2004). For male humpback whales (*Megaptera novaeangliae*), some (partly anecdotal) records of either toleration or attraction to seismic sources have been reported. Female humpback whales, by contrast, have shown avoidance behaviour in response to seismic sound (McCauley *et al.* 1998; 1999; 2000).

Clark and Gagnon (2006) reported cessation of vocalisation of a group of 250 fin whales across an area of 10 000 square nautical miles coincident with a seismic survey on the basis of recordings made using an array of hydrophones located at the continental slope off western Europe. Vocalisation recommenced once the survey had been completed.

Stone and Tasker (2006) analysed 1625 sightings of marine mammals occurring during 201 seismic surveys in UK waters between 1998 and 2000. They found sighting rates of white-sided dolphins, white-beaked dolphins, a grouping of "all small odontocetes" and a grouping of "all cetaceans" were significantly lower during periods of shooting compared with non-shooting periods on surveys with large airgun arrays. However, throughout the course of surveys, sighting rates were not found to differ significantly, indicating that any behavioural responses were short-term in nature.

Controlled exposure experiments with small airguns were carried out (source level: 215 - 224 dB re 1 µPa (p-p) over 1 hr to individual harbour seals (*Phoca vitulina*) and grey seals (*Halichoerus grypus*), and in seven out of eight trials with harbour seals, the animals exhibited strong avoidance reactions. Two harbour seals equipped with heart rate tags showed immediate, but short-term, startle responses to the initial airgun pulses. The behaviour of all harbour seals seemed to return to normal soon after the end of each trial, even in areas where disturbance occurred on several consecutive days. Only one harbour seal showed no detectable response to the airguns and approached to within 300 m of them (Thomsen 2000). An avoidance response was seen during all trials with grey seals. They changed from making foraging dives to making v-shaped transiting dives, and moved away from the source, increased swim speed and/or dive duration. Excluding two seals which hauled out after testing, those that remained in the water seemed to have returned to pre-trial behaviour within two hours of the end of the experiment (Thompson 2000). The authors commented that “both seal species reacted at the maximum ranges tested”, “they therefore reacted to relatively low signal strengths” and that responses to more powerful commercial arrays might be expected to be more extreme, longer lasting and to occur at greater ranges (Gordon *et al.* 1998 after Thompson 2000).

In a recent study of multi-species exposures to an academic seismic survey (Bain & Williams 2006), harbour seals, California sea lions (*Zalophus californianus*) and steller (northern) sea lions (*Eumetopias jubata*) exhibited general avoidance behaviour at exposure levels above 170 dB (p-p) re 1 μPa. Harbour seals generally stayed at the surface with their heads outside the water and looking towards the airguns; individuals were sometimes observed to stay closer together in the water than is typically observed. Although many seals appeared to be “responding” to the airguns, some seemed to be at least equally concerned with the acoustic monitoring vessel, and those observed at low exposure levels did not show a detectable response to the airguns (Bain & Williams 2006).

7.3 Effects on Fish

It is important to note that the principles of hearing in fish differ from those of marine mammals, and these differences have great influence on the way an assessment of sound induced impacts has to be carried out. Studies investigating sound-induced effects on fish are relatively scarce compared with those on marine mammals, and the results are variable (Hastings & Popper 2005). It is often not possible to extrapolate the results gained in specific investigations or in fundamental research to different conditions, owing to variation in hearing systems and differences in the physical properties of the sound stimulus. For more detailed reviews on fish audition, sound production and impacts of sound, see Popper *et al.* (2004); Hastings & Popper (2005); ICES-AGISC (2005) and Thomsen *et al.* (2006).

Mortality: Studies on the effects of impulsive sound found measurable and statistically significant decreases in the survival rate of both eggs and larvae in the northern anchovy (*Engraulis mordax*) (Holliday *et al.* 1987). Other studies have shown similar results, with species exposed to impulsive sound exhibiting decreased egg viability, increased embryonic mortality, or decreased larval growth when exposed to sound levels of 120 dB re 1 μPa (Banner 1973; Kostyuchenko 1973; Booman*et al.* 1996). Swim bladder damage occurred in adult anchovy at peak pressures of 217 - 220 dB (p-p) re 1 μPa as well as 50% mortalities of 2 day and 4 day old larvae at this level (Tsui 1998). A modelling study on the consequences of seismic-exploration-created mortality may have on the population level was performed by Sætre & Ona in 1996. The model was based on the observed mortality figures for larvae and fry at given distances in Holliday *et al.* (1987) and Booman *et al.* (1996). Typical versions of airgun arrays and course line densities used in 3D surveys were used as a basis, together with observed depth distributions for larvae and fry (Bjørke *et al.* 1991; Holmstrøm 1993). As a “worst case” situation, the model predicted that the number of larvae killed during a typical seismic survey could be 0.45% of the total larvae population (0.03% when more realistic expectation numbers were applied). If the same larval population were exposed to multiple seismic surveys, the effect would add up for each survey. Yet, in another investigation, the extent of seismic-induced mortality for commercial species in Norwegian waters was estimated to be so low that it was considered not to have a significant negative impact on recruitment to the populations (Dalen *et al.* 1996).

Several studies have been performed to prove any potential effects of seismic surveys on marine organisms, and the results show that harm to individual fish and increased mortality from firing airguns can occur at distances up to 5 m, with most frequent and serious damages up to 1.5 m. Fish in the early stages of life are most vulnerable (DNV Report No. 2007 – 0512).

Physiological effects: A series of studies in Australia showed that pink snapper (*Pagrus auratus*) sustained extensive damage to the hair cells located at the sensory epithelia of the inner ear after they were experimentally exposed to impulsive airgun sound. The damage was regionally severe, with no evidence of repair or replacement of damaged sensory cells up to 58 days after air-gun exposure (McCauley *et al*. 2003). Other studies have shown that underwater sound can temporarily deafen goldfish, tilapia, and sunfish (Smith *et al*. 2003; Scholik *et al*. 2001).

In contrast to the study by McCauley *et al*., Popper *et al*. published a study in which three freshwater fish species were exposed to 5 or 20 airgun impulses (Popper *et al*. 2005). Temporary hearing threshold shifts (TTS) (temporary elevation of the level at which sound can be detected) has been found in 2 of the 3 species with recovery within 18 hours of exposure at sound levels more than 10 dB higher than those used by McCauley *et al.* (2003)

Behavioural changes: A number of studies noted that various species of fish display “alarm” responses to airguns (Wardle 2001; Hassel *et al*. 2004).

When fish receive a strong sound stimulus, an alarm reaction, or an escape reaction, can be triggered (Dalen 1973; Blaxter *et al*. 1981; Blaxter & Hoss 1981; Popper & Carlson 1998; Karlsen *et al.* 2004). The reaction is often characterised by a typical so-called “C-start” response, as the body of the fish forms a C, and the body points away from the sound source. Chapman & Hawkins (1969) observed that the depth distribution of whiting changed during shooting with an airgun. The fish avoided loud levels of noise by immediately moving deeper into the water.

Studies on caged sand eels (*Ammodytes marinus*) in the North Sea revealed distinct but rather weak reactions to seismic shootings (Hassel *et al*. 2004; Skaar 2004). No increased mortality was found during this experiment.

Dalen & Knutsen (1987) observed in 1984 on the Gullfaks oil field in the North Sea that the distribution of fish at 100 – 300 metres depth changed along the course lines of a seismic vessel towing an airgun array of 40 guns with a total chamber volume of 78 litres (4750 cu.in.) during a 3D seismic survey. The average measured echo abundance, which represented the quantity of fish, was reduced by 36% after the shooting, compared with the measured values prior to shooting. Slotte *et al*. (2004) also observed that fish (herring and blue whiting) in an area where 3D seismic was shot, moved to deeper waters (approximately 10 m deeper).

Large scale changes in behaviour among fish populations exposed to seismic surveys have also been observed, as measured by catch rates from long-line and bottom-trawling fisheries. Norwegian studies have indicated probable declines in the catch rates for both cod and haddock (between 45 and 70%) in the vicinity of an airgun array, affecting fish catches at distances of nearly 25 nautical miles (Engås *et al*. 1996; Løkkeborg et al. 1993). Catch rates did not recover within five days after operations ended. A similar study showed a 52% decline in catches in a rockfish fishery exposed to a single airgun array (Skalski 1992). The reason for this decline in catch rates is unknown, but it has been suggested that it is due to changes in the swimming depth of fish or of shoaling behaviour in response to the airgun sound (Wardle 2001).

Changes in behaviour have been observed during special studies in fishery areas (Pearson *et al*. 1992) on the part of redfish species exposed to airgun shooting. Fish of these species that were held in net cages exhibited changes in swimming patterns and depth distribution during the course of 10 minutes sound exposure. These observations showed that relatively minor behavioural changes were observed even at low sound levels, and that alarm responses became more and more obvious as the sound level increased.

7.4 Effects on other species

Avoidance responses of sea turtles to low frequency sounds have been demonstrated (Lendhart 1994). Behavioural responses among turtles, such as rising to the surface and altered swimming patterns, may be elicited with exposure to seismic signals from a Bolt PAR 600B air-gun with a 0.3 litre (20 cu.in.) chamber with firing rates of 10 s and 15 s at received levels of 166 dB (rms) re 1 µPa (McCauley *et al*. 1999).

Evidence of strong behavioural reactions from squid (*Sepioteuthis australis*) to airgun sounds has been demonstrated through controlled exposure experiments in which the squid showed an increase in alarm responses above 156 dB (rms) re 1 µPa. The squid quickly changed direction away from the airgun and, in many cases, fired their ink sacs. Firing of ink sacs was not evident if the array was ramped up rather than starting at full volume (McCauley *et al*. 2000).

There have been two recorded incidents of multiple shore strandings of giant squid (*Architheutis dux*) on the coast of Asturia, Spain. Necropsies of some of the squid showed no external injuries. However, all squid had badly damaged ears. In addition, two of the squid sustained extensive damage to their internal organs. Stomachs and hearts were ripped open and muscles disintegrated with some organs being unrecognisable (Mackenzie 2004a, b). The occurrence of these strandings during two research cruises conducting seismic exploration suggests that acoustic factors could have caused or contributed to the organ and tissue lesions, and probably caused the deaths of the squid (Guerra *et al*. 2004, 2005).

Virtually no knowledge exists on the underwater hearing abilities of diving birds (such as: cormorants, black- and red-throated divers, guillemots, razorbills, puffins, albatrosses and petrels) and the sensitivity of these birds to intense anthropogenic underwater sound.

7.5 Mitigation

All mitigation measures have a common purpose, namely to prevent exposure of target animals to sound that might harm the animal. This is achieved either by avoiding exposure, or by ensuring that exposure is at levels that do not pose a threat to the animals. In the latter case this is achieved by preventing/deterring animals from approaching a sound source, thus reducing the risk of the animals being harmed by exposure to the sound, for example an operating air gun array during a seismic survey.

There is a range of mitigation measures that has been applied, either singly or in combination to mitigate the potential impacts of marine seismic surveys. The methods employed include: geographical and/or seasonal restrictions, source reduction or optimisation, buffer zones, surveillance of buffer zones by visual, acoustic or other means, “ramp-up” or “soft-start” techniques and reporting requirements. Methods are applied individually or in combination, often as required by operational guidelines.

Geographical and Seasonal Restrictions: The most effective mitigation measures are geographical and seasonal restrictions to avoid ensonification of sensitive species and habitats. Sound producing activities may be designed to avoid areas and/or times where and when sensitive marine mammals and fish species are usually engaged in susceptible activities such as mating, breeding, feeding, or migration.

This approach was taken by Australia (Environment Australia 2001), Brazil (Brazil 2004), the UK, ASCOBANS (ASCOBANS 2003), ACCOBAMS (ACCOBAMS 2004), Norway (Bjørke *et al.* 1991; Dalen *et al.* 1996), and the IWC Scientific Committee (IWC 2004), who call for seismic surveys to be spatial-temporally arranged in a way that eventual acoustic impacts are reduced. The IUCN recommends that member governments work through domestic and international legislation to consider restrictions for sound in their management guidelines for Marine Protected Areas (MPAs) (IUCN 2004). In Norway, seasonal restrictions on seismic surveys may be imposed in specific areas (Bjørke *et al.* 1991), or included in the license conditions (Anon. 1985). Prior to each seismic survey the Norwegian Institute of Marine Research is doing a resource biological evaluation and recommendation, and the Directorate of Fisheries is considering and advising in seasonal fishing periods in the area.

Source Reduction: Two international conservation agreements, ASCOBANS (2003) and ACCOBAMS (2004) and a number of advisory bodies (e.g. the California Coastal Commission, 2003) have suggested limits on source levels used during seismic surveys and have, amongst other things, proposed and/or requested the use of lowest practicable power levels, reduction of unnecessary high intensity sound (JNCC, 2003), array optimisation or avoidance of sources of ‘unnecessarily’ high energy. As an example, within the UK, the UK Joint Nature Conservation Committee calls for operators to reduce unnecessary high-intensity sound produced by airguns or other acoustic energy sources (JNCC 2003), and the JNCC guidelines have been incorporated into relevant permits for oil and gas seismic surveys within the UK.

Buffer/Safety Zones: Buffer or safety zones are frequently defined as a circular area around a sound source (whether this is stationary or moving). Animals outside this zone are presumed not to be exposed to harmful levels of sound. The radius of buffer/safety zones is most often defined by the regulatory agency or promoted by other groups (IUCN 2006), and may for example range from 500 metres (JNCC, 2003) to in excess of 1000 metres (Environment Australia, 2001). The presence of animals within the buffer/safety zone may require shooting down an operating array or delaying its start-up.

Visual Surveillance of Buffer Zones: Managing buffer or safety zones is frequently achieved by specialist marine mammal observers (MMOs). As the name suggests, these observers scan the buffer/safety zone before and during start-up and also through the period of the survey, recording and subsequently reporting sightings of animals within (and beyond) the safety zone. MMOs are often required to have specialist training. The ability to monitor buffer zones will be determined by sea state and practical visibility. Moreover, the ability to monitor certain species is limited even within small radii (Barlow & Gisiner 2006): visual detection probability of beaked whales, for example, is 1 - 2% at most due to their long dives (US-MMC 2004).

Reporting on visual surveillance can provide information that may aid understanding of behavioural reactions. IWC (2004) has recommended:

- Continuous acoustic monitoring of critical habitats on sufficient temporal and spatial scales in relation to pre- and post-seismic activity.
- Independent monitoring of critical habitats (from survey vessel and independent platforms) to evaluate displacement from critical habitat and/or disruption of important cetacean behaviours in the critical habitat.
- Increased effort to monitor strandings that may coincide with the activity (IWC 2004).

Soft Start/Ramp-up techniques: The aim of soft start or ramp-up is the gradual increase in sound from an array (either by starting from a single gun and adding elements sequentially, or by gradually increasing the power output). The soft start is designed to give animals the opportunity to leave the survey area before ‘operational’ sound levels are reached. A soft-start may be employed over 20-30 minutes before full power is reached and a survey line commenced.

Other Surveillance methods: Visual surveillance is frequently supplemented by acoustic and other electronic techniques. These include both passive and active acoustic monitoring, as well as radar and infrared scanning.

7.6 Conclusions

Seismic surveys – for mapping of features beneath the ocean floor – are used for oil and gas exploration, but also for academic research, and for gathering data for the purpose of delineating Economic Exclusive Zone (EEZ) extensions under the United Nations Convention of the Law of the Sea (UNCLOS). In seismic reflection surveys, airguns are the most commonly used sound source. The airguns release compressed air to generate the seismic signals’ regular intervals, typically each 25 metres the vessel moves. The sound waves are scattered from boundaries between the various geological layers in the subsurface. The backscattered signals are registered by several groups of hydrophones mounted in cables towed behind the ship.

It is generally accepted that intense anthropogenic sources have the potential to cause adverse effects on marine mammals and other marine organisms (see section 7.2). There have been a few cases of strandings of beaked whales and giant squids coinciding with academic seismic surveys. However, there is no conclusive evidence of a link between sounds of seismic surveys and the mortality of any marine mammals Furthermore, there is limited information on possible physical injury (permanent or temporary threshold shift).

There is a considerable volume of research concerning behavioural responses to intense sounds generated by seismic airguns. While many of these studies have reported changes of behaviour in a range of species, no universal conclusions can be drawn. Moreover, where responses have been observed in individual animals or small groups of animals, it is not known whether these reactions are significant at the population level for the species investigated.

Studies investigating sound-induced effects on fish are relatively scarce compared with those on marine mammals, and the results are variable. Studies have been performed on the effects of seismic surveys on marine organisms and the results show that harm to individual fish and increased mortality from firing airguns can occur at distances up to 5 m, with most frequent and serious damages up to 1.5 m. Fish in the early life stages are most vulnerable.

The extent of seismic-induced mortality for commercial species is estimated to be so low that it is considered not to have significant negative impacts on recruitment to the populations. Adult fish show behavioural responses to the sound waves from seismic activity. Based on the few existing studies showing a reduction in catch rates during noise exposure behavioural response is indicated within a radius of several kilometres from the sound source. If fish that are on their way to the spawning grounds are exposed to this type of noise, or if they are exposed to the noise during the actual spawning, the effects can have an impact on the fish's spawning success and thereby the recruitment.

Avoidance responses of sea turtles to low frequency sounds have been demonstrated. Evidence of strong behavioural reactions from squid to airgun sounds has also been demonstrated.

A range of measures is currently employed to mitigate exposure of marine mammals during seismic surveys. These are employed singly or in combination, and are often required as part of operating guidelines or as a condition of licence or permit. These include visual and acoustic surveillance, buffer zones, source optimisation and soft start procedures. Seasonal restrictions may also be used to protect animals (mammals and fish) during sensitive parts of their lifecycle such as breeding, calving or spawning, nursing and migration.

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Module 8: Noise profiles of other activities

8.1 Noise profiles of other activities

8.1.1 Acoustic Deterrent Devices

Reeves *et al.* (2001) (48) defined high power devices operating at broadband source levels above 185 dB re 1µPa @1m[15] as Acoustic Harassment Devices (AHDs) (Table 8.1) while those operating at lower source level were termed Acoustic Deterrent Devices (ADDs) (Table 8.2). ADDs or "pingers" are generally used to deter small cetaceans from bottom-set gillnets or other fisheries in order to reduce lethal by-catch. Pingers operate at much lower source levels than AHDs; usually 130 to 150 dB re 1 µPa (Table 8.2). Acoustic characteristics of ADDs differ particularly with respect to randomisation of pulse intervals and pulse duration (Table 8.2). However, the signal structure and source levels of the vast majority of pingers are relatively consistent as they have to comply with guidelines laid down by EU Council regulation (EC) No 812/2004. These devices produce either 10 KHz tones or wide-band sweeps covering a frequency range from 20 to 160 kHz. According to EC 812/2004, pingers that are based on analogue signal generation emit tones (10 kHz) at source levels (broadband) between 130 and 150 dB re 1 µPa while digital devices can either have the same specifications or produce wideband sweeps at broadband source levels of 145 dB 1 µPa (see Table 8.2).

Two further devices listed in Table 8.2 that are designed to reduce dolphin depredation operate at higher source levels , the High Impact Saver by SaveWave and the DDD (Dolphin Dissuasive Device) by STM. The acoustic output of the SaveWave device is however still well below that of acoustic harassment devices used against pinnipeds (Table 8.1).

8.1.2 Acoustic Harassment Devices

Acoustic Harassment Devices (AHDs) were originally developed to prevent pinniped predation on finfish farms, fisheries or salmon runs through production of high source level acoustic signals. Table 8.1 details the wide range of AHD specifications including frequency range, signal types and source levels (SPL @ 1m). For instance, the Lofitech seal scarer emits 11 kHz constant-frequency pure tone pulses while the device produced by Ace-Aquatec emits series (up to 20 seconds) of short pulses (2 - 12ms) that encompass a range of different fundamental frequencies from 5 - 15 kHz. The Terecos seal scarer emits complex and time-variable variable blocks or tones (35). Despite the differences between AHDs, a common feature is that most devices produce substantial energy in the ultrasonic range in addition to the main frequency band [up to 70 kHz (*e.g.* Ace-Aquatec)]. The broadband source level of most AHDs is approximately 195 dB re 1 µPa with the exception of the Terecos device (179 dB re 1 µPa, broadband) and an older model by Ferranati-Thomson (above 200 dB re 1 µPa). Due to their relatively high source level and often broadband characteristics AHDs can potentially be a significant source of noise in areas of dense fish farming (23).

[15] Unless stated otherwise all source levels (SL) in this chapter are given in dB re 1µ Pa @1m distance.

8.1.3 Fish deterrent devices

Fish deterrent devices are primarily used in coastal or riverine habitats for temporarily removing fish from areas of potential harm (*e.g.* guiding fish away from water intakes of power plants). There is considerable variation between fish deterrent devices with respect to the frequency range depending on the fish species that are targeted. For applications where the device is to be effective against a broad range of species, relatively low or infrasonic frequencies are generally used. For example, some devices produce infrasound at frequencies of about 10 Hz (31) and a system tested at a water inlet of a power plant in an estuary consisted of an array of transducers emitting frequencies between 20 and 600 Hz at a broadband sound pressure output of 174 dB re 1 µPa (38). However, other devices produce primarily ultrasonic frequencies and are specifically designed to deter high-frequency hearing specialists (e.g. Sonalyst Inc. or Ultra electronic Oceans Systems). Fish deterrent devices for some clupeid species which have ultrasonic hearing operate at frequencies between 120 kHz and 130 kHz, with source levels up to 190 dB (10, 51, 52). Fish deterrent devices generally produce sequences of short pulses (e.g. 100 - 1000 ms) at intervals of one to several seconds and duty cycles up to 50% (proportion of time the device is switched on).

8.1.4 Acoustic data transmission

Acoustic modems are used as an interface for subsurface data transmission, and are currently used in many industrial and research applications. There is considerable variation in the frequency ranges used by commercial modems. For example, the AQUAModem produced by Aquatec produces a relatively low frequency (8-12 kHz) acoustic signal compared to the Hydroacoustic Modem S2C M 48/78 produced by EvoLogics GmbH (48-78 kHz). However, most encompass a range of around 18-40 kHz. The broadband source levels are relatively high, ranging from 185 to 196 dB re 1 µPa. A relatively new integrated communications project is the “Acoustic Communication Network for Monitoring of Underwater Environment in Coastal Areas (ACME)”. This system uses chirps of continuously varying frequencies and frequency-shift keying noise covering a frequency range from 5 kHz and 15 kHz with most energy centred around 12 kHz (29).

***Table 8.1**: Signal characteristics of Acoustic Harassment Devices (AHDs) typically used on fish farms to deter pinnipeds*

Manufacturer	Model	Source level dB re 1 µPa @ 1m [16]	Frequency structure	Temporal pattern	Duty cycle (proportion of time sound is switched on)	Energy in the ultrasonic range	Reference
Ferranti-Thomson	Ferranti-Thomson MK2, Mk3 & 4X Seal scrammer	MK2 model 195dB @ 27 kHz (peak) for 4X model: 200dB @ 25 kHz	Pulses centred at 5 different frequencies arranged in 5 pre-set sequences which are chosen randomly	20 ms pulses repeated every 40 ms in trains of 20s duration[2]	3 % max. 5.5 scrams per hour	Up to at least 40 kHz	[3]
Ace-Aquatec	Ace-Aquatec	193 dB @ 10 kHz (rms)	Pulses at 28 different frequencies arranged in 64 sequences which are randomly chosen	3.3-14 ms long segments in 20 s long trains	Activity-dependant (50% if trigger is released, but max 18 times per hour)	> 165dB at 30 kHz;145 dB at 70 kHz	[1]
Airmar Technology Corporation	Airmar dB Plus II	192 dB @ 10.3 KHz (rms)	More or less sinusoidal: 10.3 kHz (2nd harmonic 43 dB weaker)	1.4 ms long segments at 20 ms intervals in 2.25m long trains; 4 transducers produce these trains in an alternating pattern	40-50 %	145 dB up to 103 kHz	[1]
Terecos Ltd	Terecos type DSMS-4	178 dB @4.9 kHz[1] (rms)	Complex tonal blocks forming up and down sweeps (fundamental from 1.8 kHz-3 kHz), sequences of continuous and time variant multi-component blocks (2.4 kHz-6kHz), continuous tonal blocks forming sweeps	Depending on operation mode: 8ms segments in sequences of eight or 16ms segments in sequences of 5; variation possible due to randomisation software; trains from 200ms to 8 s long	a. 50 %	less than 143 dB above 27 kHz	[1, 2]
Lofitech (older models by SIMRAD)	"Universal scarer" or "seal scarer"	191 dB @15 kHz	15 kHz (tonal, narrow-band)	500ms pulses in 6s trains long trains	20 - 25 %	Single harmonic (depending on battery status)	[2]

1 Lepper, P. A., Turner, V. L. G., Goodson, A. D. & Black, K. D. 2004. Source levels and spectra emitted by three commercial aquaculture anti-predation devices. In: *Seventh European Conference on Underwater Acoustics, ECUA*. Delft, the Netherlands.

2 Reeves, R.R., Read, A.J., Notarbartolo di Sciara, G., 2001. Report of the Workshop on Interactions between Dolphins and Fisheries in the Mediterranean: Evaluation of Mitigation Alternatives. Istituto Centrale per la Ricerca Scientifica e Tecnologica Applicata al Mare, Rome, Italy

3 Yurk, H. & Trites, A.W. 2000. Experimental attempts to reduce predation by harbour seals (Phoca vitulina) on out-migrating juvenile salmonids. Transactions of the American Fisheries Society 129: 1360-1366

[16] source levels are calculated over a broad analysis bandwidth unless values are given at a certain frequency

Table 8.2: *Signal characteristics of Acoustic Deterrent Devices (ADDs) and other devices to reduce cetacean depredation*

Manufacturer	Model	Source level dB re 1 µPa @ 1m [17]	Fundamental frequency	Frequency range	Pulse duration	Inter-pulse interval	Reference
Airmar Technology group	Gillnet pinger	132 dB	10 kHz	Harmonics present	300ms	4 s	[2]
Aquatec Subsea Ltd	Aquamark 100, 200, 210, 300	145 dB (model 100, 200, 300), 150 dB (model 210)	Model 200, 210: 5-60 kHz frequency sweeps Model 100: 20-60 kHz frequency sweeps Model 300: 10 kHz tonal	Harmonics up to 160 kHz	50 - 300 ms (depending on model)	4 - 30 s (pseudo-randomised except for model 300)	www.aquatec.com and [1]
Dukane NetMark	NetMark 1000 & 2000 (stopped manufacturing)	132 dB	10 kHz	Harmonics up to 73 kHz	300ms	4 s	[2]
Fumunda	FMDP 2000	132 dB	10 kHz	-	300ms	4 s	[2]
Marine Technology Marexi	Marexi Acoustic pinger V.2.2	132 ±4 dB	10 kHz	-	300ms	4 s	www.http://www.pinger.es
SaveWave	"High impact saver" "Long line saver" "Endurance saver"	155 dB (HI, LL model), 140 dB (ES model)	HI model: signal has two partials: 5 - 40 kHz and 30 - 160 kHz; wide band sweeps. LL model: 5 - 60 kHz wide band sweeps. ES model: 5-90 kHz	Harmonics up to 180 kHz	200-900 ms randomised (depending on model)	4 - 16 s (randomised)	www.savewave.net, "Guidelines to minimise cetacean-fishery conflicts in the ACCOBAMS Area" (http://www.accobams.org), and [1]
STM	DDD (Dolphin Dissuasive Device	-	1 - 500 kHz	-	-	-	Manufacturer's information

1 Kastelein, R. A., van der Heul, S., van der Veen, J., Verboom, W. C., Jennings, N., de Haan, D. & Reijnders, P. J. H. 2007. Effects of acoustic alarms, designed to reduce small cetacean by-catch in gillnet fisheries, on the behaviour of North Sea fish species in a large tank. *Marine Environmental Research*, 64, 160-180.

2 Reeves, R.R., Read, A.J., Notarbartolo di Sciara, G., 2001. Report of the Workshop on Interactions between Dolphins and Fisheries in the Mediterranean: Evaluation of Mitigation Alternatives. Istituto Centrale per la Ricerca Scientifica e Tecnologica Applicata al Mare, Rome, Italy.

[17] source levels are calculated over a broad analysis bandwidth unless values are given at a certain frequency

8.1.5 Research activities

Ocean science studies use a variety of different sound sources to investigate the physical structure of the ocean. These include explosives, airguns, and underwater sound projectors. Ocean tomography studies measure the physical properties of the ocean using sound sources with frequencies between 50 and 200 Hz with high source levels (165 - 220 dB re 1 µPa, 48). Perhaps the best known of these is the "Heard Island Feasibility Test" where signals with centre frequencies of 57 Hz were projected in the SOFAR channel at 175 m depth at source levels up to 220 re 1 µPa for around 1 hour each day (4). The signals could be detected across ocean basins with received levels up to 160 dB re 1 µPa at 1km distance. Another major research project using sound as a tool was the "Acoustic Thermometry of Ocean Climate" (ATOC) research programme. The ATOC sound source emitted coded signals at 75 Hz at source levels of 195 dB re 1 µPa for up to 20 min (see 50 for a brief review). Geophysical research activities e.g. studies on sediments in shallow water may also use typical mid- or low-frequency sonar systems or echo-sounders.

8.1.6 Marine renewable energy devices

Offshore tidal and wave energy turbines are a relatively recent technological development and available information on the acoustic signatures of such activities is limited. However, tidal turbines appear to emit broadband noise covering frequency range from 10 Hz up to 50 kHz with significant narrow band peaks in the spectrum (45). Depending on size, it is likely that tidal current turbines will produce broadband source levels of between 165 and 175 dB re 1µ Pa.

8.2 Impact on marine mammals

8.2.1 Responsiveness

When considering the impact of acoustic deterrent and harassment devices on the behaviour of marine species, it is important to distinguish between target and non-target species. While it is clear that a deterrent device aims to cause at least some form of moderate avoidance response in the target species, ideally it should not influence non-target-species behaviour. Since acoustic harassment devices operate at much higher source levels than pingers they can be expected to cause stronger avoidance responses. High-power AHDs (*e.g.* Ferranti-Thompson 4x) can be audible to harbour porpoises (*Phocoena phocoena*) up to 10 km away under low ambient noise (55) while harbour seals (*Phoca vitulina*) could potentially hear a device with a source level of 175 dB re 1µPa @ 1m at distances of 1.4 km to 2.9 km in quiet conditions (57). Early AHDs operating at a peak-to-peak (p-p) source level of 187 dB re 1µPa @1 m caused initially strong avoidance responses in seals. However, responses appeared to decline after several years. Although it is possible that hearing loss occurred over this period, habituation or even a conditioned response resulting from the association of the sound with a profitable food source ("dinner bell effect") are more likely explanations. However, habituation due to positive stimuli (food) may increase the risk of the animals being exposed to deleterious effects of noise.

Jacobs & Terhune (2002) (21) tested an Airmar dB Plus ADD that operated at a measured p-p source level of 172 re 1µPa @ 1m with harbour seals around a haul-out. They found no difference between control and sound exposure sessions. However, other studies have found deterrent effects on target species (grey seals) over several consecutive years (15). In contrast to the findings for target-species, several studies showed that AHDs can have dramatic impacts on odontocetes which do not seem to predate on finfish farms in most areas in Northern and Western Europe. Olesiuk *et al.* (44) investigated effects of the Airmar AHD on harbour porpoise distribution in the British Columbia. They showed that porpoises were completely excluded from an area of 400 m radius around the AHD and the number of sightings dropped to 10% of the expected value at ranges between 2500 and 3500 m from the device. Similarly, Johnston (22) showed that porpoises did not approach an emitting AHD closer than 645 m (received level at this distance would be 128 dB re 1µPa). Porpoise numbers were also significantly lower at ranges of up to 1500m from the sound source. Morton & Symmonds (42) reported a reduction in killer whale (*Orcinus orca*) sightings rates in Johnston Strait, Canada after AHDs had been introduced on fish farms, and a sudden recovery after they were removed. Similarly, Morton (41) found that Pacific white-sided dolphin (*Lagenorhynchus obliquidens*) abundance decreased after AHDs were introduced in the area. In summary the previously mentioned studies seem to show that AHDs can cause both avoidance and habitat exclusion in *odontocetes*, but appear to have only moderate effects on pinnipeds (target species). These differences may have to do with the fact that all *odontocetes* for which audiograms are currently known have 30-40 dB lower hearing thresholds at frequencies where AHDs operate (10-40 kHz) compared to pinnipeds (17). This would mean that AHDs sounds are perceived as louder by odontocetes as by pinnipeds. In addition, in many areas in Northern and Western Europe where AHDs are commonly used, odontocetes do not seem to forage commonly on farmed fish and potentially will be less motivated to remain in the area.

Early experimental studies showed that ADDs have the capacity to reduce by-catch of small odontocetes (32). Field observations around simulated net equipped with pingers that operated at source levels of 145 dB re µPa at 1 m showed that harbour porpoises avoided an area of about 130 m around the sound source (9). Studies in other areas have found larger exclusion zones around pingered nets (300 – 500 m), and this has raised concerns that ADDs might exclude porpoises from important habitats (3). However, there is also some evidence for habituation of porpoise avoidance responses to pingers over periods of several weeks (7). A recent study, that tested different acoustic alarms operating a frequencies between 100 and 140 kHz (source level between 128 and 153 dB re 1µPa) on two captive porpoises also demonstrated a degree of habituation (56). In contrast, an earlier captive study on two harbour porpoise did not find clear habituation within and across playback session.

It is also important to note that different cetacean species may respond differently to pingers. For instance, wild bottlenose dolphins did not show strong avoidance responses to a simulated gillnet equipped with Dukane NetMark 1000 pingers (8). A captive study on a single harbour porpoise (*Phocoena phocoena*) and a striped dolphin (*Stenella coeruleoalba*) demonstrated differential responses to pinger-type signals with the dolphin showing much weaker aversive responses than the porpoise (25). There is also some indication from a correlating study covering a period of 17 years, that beaked whales (*Ziphiidae*) may exhibit stronger responses to pingers than other cetaceans (6). By-catch of beaked whales in gillnet fisheries ceased completely after pingers were introduced while a steady level of by-catch remained for all other species (it should be noted that the level of beaked whale by-catch was only a fraction of those for other species previously).

Fish deterrent devices can influence different species depending on whether they emit high or low frequencies. With respect to low-frequency sources mentioned in this report, baleen whales are likely to be affected. However, this may be reduced to some extent as baleen whales rarely inhabit areas where fish deterrents are used. Kastelein *et al.* 2006b (28) showed that behavioural avoidance responses of seals to mid-frequency artificial sounds (fundamentals 8 - 12 kHz) occurred at received levels of 108 dB re 1 µPa. However, the hearing threshold of a harbour seal is 20 - 25 dB less sensitive at the frequencies where fish deterrents operate (*e.g.* 500 Hz). It is therefore possible that seals would only exhibit responses at higher received levels at these frequencies but direct measurements would be needed to draw any meaningful conclusions and predict deterrence ranges caused by fish deterrent devices. Fish deterrent devices using high or ultrasonic frequencies (52) have a clear potential to affect odontocetes on a similar magnitude as AHDs. Behavioural impact zones are likely to be similar to those predicted for AHDs. High-frequency fish deterrents also have some potential to impact seals.

Behavioural effects of acoustic data transmission systems on captive harbour porpoises have been studied with a system emitting broadband signals with most energy centred at 12 kHz (29). Results showed that porpoises exhibited avoidance responses at received levels between 97 and 112 dB re 1 µPa (all SPL in this study were calculated over the main four fundamental 1/3 octave-bands for the respective sounds, see 30). The authors predicted that if the device were to operate at a source level of 170 dB re 1 µPa the “discomfort zone” would be between 1.2 and 6.3 km from the device depending on the sound type (source level calculated for the main fundamental 1/3 octave bands). In a second experiment using similar methodology, avoidance responses in captive harbour seals (*Phoca vitulina*) occurred at received levels between 107 and 108 dB re 1 µPa. This equates to a predicted impact zone (assuming 170 dB source level) of up to 2km (28). Although both studies were conducted on captive animals they show that data transmission devices have a clear potential to influence marine mammal behaviour over relatively large areas.

Behavioural impacts caused by ocean research activities have been a major concern over the last decade (50), but evidence for effects on behaviour of marine mammals remains controversial. In the “Heard Island Feasibility Test”, sighting rates of medium-sized or large whales (*e.g.* pilot, beaked and baleen whales) dropped in response to the projected 57 Hz signal, but dolphins and fur seals did not show any apparent response and even seemed to approach the source vessel (4). In addition to the movement responses, sperm and pilot whales temporarily ceased calling. A follow-up experiment used a sound source that projected phase-modulated signals at 75 Hz. Au et *al*. (1) measured detection threshold of the signal in two captive delphinids and concluded that the signal would not be audible to these animal unless they dived to a depth of 400 m. The authors also concluded that effects on baleen whales are likely to be minimal since these animals use similarly loud calls for communication (170-180 dB re 1 µPa, broadband). However, the latter argument is controversial since behavioural responses to conspecific calls can be expected to be different from responses to unknown artificial sounds. Southall *et al.* (2007) reviewed behavioural responses of baleen whales to low-frequency artificial sounds and concluded that, in most cases, non-migrating whales responded to received levels between 140 and 160 dB re 1 µPa. In the “Heard Island Feasibility Test”, received levels of 160 dB were measured up to distances of one kilometre (4). In conclusion, there is the clear potential for ocean tomography and thermography studies to influence the behaviour of mysticetes and medium sized odontocetes; this is less likely for small odontocetes and pinnipeds.

Marine renewable energy devices are relatively new, and empirical studies (45) attempted to predict behavioural impact zones for environmental impact assessment purposes. They used a sensation level criterion of 75 dB for marine mammals within the frequency band of interest and predicted that depending on sound propagation conditions, mild aversion zones for harbour porpoises were expected to be 108-280 m, while aversion zones for harbour seals would only extend 5-15 m from the source. These conclusions are arguable since the criterion used is only partly validated by empirical data. Therefore, studies on the behaviour of marine mammals around tidal turbines need to be carried out.

8.2.2 Masking

Marine mammals use sound for communication, orientation, and prey detection (see Richardson *et al.* 1995). Masking generally refers to the detection of one tonal signal being influenced by a second sound. Unless the masking sound is very broadband the masking effect is dependent on the bandwidth of the masker up to a critical bandwidth (see 16). Critical bandwidths in marine mammals are generally below 10% of the signal’s centre-frequency. For masking to occur there needs to be an overlap in the frequency range of signal and masker and the received level of the masker needs to exceed that of the signal. Noise produced by some AHDs (*e.g.* Lofitech) clearly overlaps with the frequency range of the communication signals of many delphinid species. In contrast, vocalisations of baleen whales and many pinniped species that occur in areas of intense aquaculture tend to be lower in frequency and are therefore less likely to be masked (50). However, vocalisations of some seal species have energy over 10 kHz (*e.g.* bearded seals) or even extend into the ultrasonic range and would therefore be prone to masking by high duty cycle AHDs (see 50 for a summary of seal vocalisations).

Pingers generally emit sound in short pulses (300 ms) which might reduce the masking potential with respect to longer duration communication like whistles in delphinids. In theory, there is the potential that ADDs which emit broadband sound pulses in the frequency range between 20 and 150 kHz could mask the echolocation clicks of odontocetes (*e.g.* harbour porpoises) but this seems less likely. One needs to consider that masking effects are attenuated if the masker and signal come from different directions. Bottlenose dolphins (*Tursiops truncatus*) can distinguish sound sources that are presented at angles of less than 3° apart (49). Furthermore, bottlenose dolphin hearing is directional (Au & Moore 1984) which increases the capability of detecting signals in noise if the masker noise source and target sound are spatially separated. Therefore, cetaceans may successfully avoid masking effects in some cases. However, direct measurements have to be obtained to confirm this.

With respect to sound produced by fish deterrent devices, ocean tomography or tidal turbines, little is known about the potential effects of masking. Low-frequency, high source level fish deterrent devices have some potential to mask communication signals of baleen whales and seals depending on whether signals are narrow or broadband. Although there may be little spatial overlap of geographical areas in which fish deterrents are used and the habitat of baleen whales, seals commonly breed in coastal or estuarine habitats. For instance, a fish deterrent device producing a sound pressure component of 195 dB re µPa within a frequency range of 200 Hz and 500 Hz may have strong potential to mask the communication calls produced by harbour seals during the breeding season (see 18 for characteristics of harbour seal calls).

Although there are no studies on the masking effects of data transmission systems, they have some masking potential with respect to dolphin communication signals because the sounds are relatively broadband and overlap with the typical frequency range of communication signals like whistles. There is also some potential for masking of the underwater vocalisation of some seal species. With respect to other low-frequency sound sources (*e.g.* ATOC) there may some potential to mask baleen whale communication or vocalisations of certain seal species (*e.g.* harbour seals; 17). Similarly, low-frequency noise by tidal turbines may again primarily affect seals and baleen whales. If noise by tidal turbines is broadband rather than tonal (with a harmonic structure) the masking potential may be higher.

8.2.3 Injury

The risk of injury caused by any of the aforementioned noise sources is most likely to be limited to hearing damage. Hearing damage occurs first as a temporary but fully recoverable shift of the hearing threshold (temporary threshold shift=TTS). As a result of exposure to higher intensity or longer duration acoustic stimuli recovery may not be possible and the threshold shift becomes permanent causing chronic damage (permanent threshold shift=PTS). The risk of hearing damage is considered to be a function of sound pressure level and exposure time (11). Therefore, sound exposure level (SEL) or energy flux density might be a good measure for defining safe exposure levels. Southall *et al.* (2007) reviewed available literature in an attempt to define noise exposure recommendations for different marine mammal taxa (see also module 3). Although there are a number of limitations with the study, it is currently the only integrated approach combining all available data. However, it should be acknowledged that until more experimental data is available on hearing damage in marine mammals, there remains a high degree of uncertainty in the injury risk predictions.

8.3 Impact on fish

Most fish species have low frequency hearing, and are generally not very sensitive to sound pressures at frequencies higher than 1-2 kHz [hearing generalists (14)]. However, “hearing specialists” like Atlantic herring (*Clupea harengus*) possess good sensitivity to sounds up to several kHz with some clupeid species having ultrasonic hearing (39).

8.3.1 Responsiveness

AHDs and ADDs primarily produce high-frequency sound (10 - 150 kHz). Therefore, it is expected that behavioural disturbance will be limited to species with good high-frequency hearing. Kraus *et al*. (1997) found that herring catch rates in gillnets equipped with pingers were lower than expected. In contrast, a more recent study suggested that there were no significant differences in catch rates (Culik *et al*. 2001). Kastelein *et al*. (2007) tested behavioural responses of a range of North Sea fish species to several types of commercially available pingers. Although none of the fish species exhibited a startle response, sea bass (*Dicentrarchus labrax*), thicklip mullet (*Chelon labrosus*), Norway pout (*Trisopterus esmarkii*), and herring changed their swimming behaviour in response to some pingers while cod (*Gadus morhua*) did not respond to any of the devices. The authors concluded that only pingers which produce frequencies lower than 10 kHz and have a source level above 130 dB re 1 µPa are likely to have a significant influence on the behaviour of fish.

Responses of fish to commercially available acoustic harassment devices (AHDs) have not been tested yet. However, in fish with good ultrasonic hearing, strong avoidance responses have been shown in response to frequencies of 110 - 130 kHz at received levels down to approximately 160 dB re µPa (10). This would indicate that AHDs which produce substantial energy in the ultrasonic range (*e.g.* Ace-Aquatec) may cause some behavioural avoidance responses in clupeids in the immediate vicinity of the transducer (*e.g.* closer than 20m). Kastelein *et al.* (2007) (26) investigated startle responses in fish to a variety of different sounds ranging in frequency from 0.1 to 60 kHz at maximum received levels of up to 180 dB re 1 µPa. The only species that showed a response to frequencies higher than 2 kHz was Atlantic herring which responded at 4 kHz to received levels of approximately 170 dB re 1 µPa. Assuming spherical spreading losses, received levels around a commercial AHD would probably be down to 170 dB re 1 µPa at distances not more than 20 m. In conclusion, ADDs and AHDs are probably unlikely to influence the behaviour of fish unless they are close to the device. Given the similarity in acoustic characteristics of data transmission devices to AHDs, the responses are expected to be similar.

Fish deterrent devices projecting infrasound or low-frequency sound (up to 500 Hz) have been shown to be at least partly successful in keeping several species out of water intakes in riverine or estuarine habitats (*e.g.* 38, 53). These responses are considered to be caused primarily by the particle motion component of the sound, and may therefore be short-range and unlikely to influence fish populations in the surrounding habitat. Responses of fish to deterring sounds also vary depending on the frequency band of the projected sounds; experiments with salmon (*Salmo salar*) smolts showed that a 10 Hz signal 114 dB above the hearing threshold (at the respective frequency) caused an avoidance reaction while a 150 Hz signal did not (30). Fish deterrents emitting ultrasound at frequencies between 130 and 140 kHz are based on the finding that clupeid species exhibit strong avoidance responses to these signals (43). However, behavioural responses to these fish deterrents seem to be limited to the small areas around the sound source (43).

Responses of fish to simulated signals of the ATOC sound source showed that fish exhibited moderate attraction rather than avoidance responses. However, Kastelein *et al.* 2008 (27) showed startle responses to 100 Hz pure tones in several fish species [*e.g.* pelagic horse mackerel (*Decapterus maruadsi*)] at received levels of approximately 120 dB re µPa. This would mean that fish over a relatively large area around an ATOC source could be affected behaviourally.

As with marine mammals, Parvin *et al.* (45) recently made predictions on behavioural impacts of tidal turbine noise on fish but some of the underlying assumption may be problematic (use of sensation level criterion of 75 dB for fish). This study concluded that it is unlikely that that hearing generalists will be affected and hearing specialists may show behavioural responses up to a few metres. Kastelein *et al.* (27) showed that startle responses occur in some in North Sea fish species at received levels down to 130 dB at 300-500 Hz. However, significantly more information is needed to predict the responses of fish to anthropogenic noise from tidal turbines.

8.3.2 Masking

Most communication signals in fish fall within a frequency band between 100 Hz and 1 kHz and (see Zelick *et al.* 1999(60)). Hearing abilities (*e.g.* localization and frequency discrimination) in most fish species are less sophisticated than in mammals (14), which may make them more prone to masking effects. Consequently, masking in fish also follows different principles than in mammals. A neuro-physiological study on goldfish (a hearing specialist) showed that responses of nerve fibres to tones between 400 and 800 Hz can be suppressed by maskers of a broad range of frequencies essentially covering most of the hearing range (13). Elevated detection thresholds as a result of masking have been shown in hearing generalists as well as specialists (58, 59) although species clearly differ in their susceptibility to masking (47). This might mean that even signals outside the frequency band used for communication but within the hearing range may cause masking. Masking in fish has been primarily studied with broadband noise making it difficult to say how pure tones or more multi-tone sweeps influence fish communication. Continuously operating low-frequency fish deterrent devices or signals produced by oceanographic studies or noise from tidal turbines might be a source of masking. In contrast the masking potential of AHDs, ADDs, and data transmission devices may be low, at least with respect to the species that do not posses high-frequency hearing. The potential of masking of fish communication signals really needs further investigation.

8.3.3 Injury

Injury in fish can occur on the level of physical trauma, caused by the pressure wave, or by causing temporary or permanent hearing damage (19). The noise sources discussed here are unlikely to inflict direct physical trauma but may still damage hearing. Hair cell damage in fish has been found in cod exposed to broadband sound pressure levels of 180 dB re 1 µPa for several hours (12), in oscars (*Astronotus occelatus*) that were exposed to 300 Hz sine wave sounds of the same source levels (20), and snappers (*Pagrus auratus*) that were repeatedly exposed to airgun emissions (received levels up to: broadband 180 dB re 1 µPa) (40). Smith *et al.* (2004) found a linear correlation between the logarithm of exposure time and the amount of temporary threshold shift caused by experimental exposure to white noise (170 dB re 1 µPa) in goldfish (a hearing specialist). However, such a relationship was not found in tilapia (a hearing generalist). All mentioned studies used signals that contained at least some energy within the most sensitive hearing range of these species making it difficult to draw conclusions about the effect of higher frequency signals as those used in AHDs, ADDs, or data transmission devices. However, even when assuming that fish may be equally susceptible to higher frequencies effects of current high source level AHDs would probably not extend much beyond 100 m from the source where received levels can be expected to be lower than 150 – 170 dB re µPa.

Lower-frequency sound sources like those used in ocean thermography studies do have the potential to affect fish. Popper *et al.* (46) tested effects of low-frequency active sonar (LFA) on rainbow trout and found exposure to narrow-band 193 dB re 1 µPa to cause 20 dB threshold shift as a result of sound exposure. As mentioned earlier Hastings *et al.* (20) showed that 1h of sound exposure to continuous 300 Hz tones at 180 dB re 1 µPa (frequency 300 Hz) caused some hair cell damage in oscars, but exposure to the same signal at duty cycle of 20% did not result in any damage. Assuming that exposure to received levels of 180 dB re 1 µPa would cause permanent hearing damage, the ATOC signal would only affect fish that are very close to the source. Finally, it may be worth noting that in contrast to some of the damage caused exposure to loud sound may be reversible. However, this may only be the case in certain circumstances and even temporary hearing damage may still have a fitness cost to the fish.

8.4 Effects on other species

The effect of any of the aforementioned noise sources on reptiles and invertebrates will depend on the ability of these animals to detect sound. Sea turtles seem to be primarily sensitive to low-frequency sound below 1 kHz (2). As a result, they are unlikely to be influenced by noise from AHDs, ADDs, and data transmission devices. However, although sea turtles may be influenced by low frequency sound like ocean tomography or thermometry studies (*e.g.* ATOC), there are currently no empirical data available. Similarly, there is some potential that other noise sources like tidal turbines may have some influence on behaviour. Invertebrate detection of vibration stimuli is primarily low-frequency and mostly limited to the particle motion component of the signals. For instance, some cephalopods are sensitive to water movement stimuli up to 100 Hz (5), and a species of prawn has been shown to be able to detect the particle motion component of sounds up to 3 kHz (36). This may indicate that lower frequency noise (*e.g.* by fish deterrents or tidal turbines) may have some effect on these species if they are very close to the sound source. It is unknown whether the pressure component of sound that would be detectable in the far field adversely influences sea turtles or crustaceans. There are currently no reliable data available on hearing damage in sea turtles or invertebrates as a result of exposure to anthropogenic noise.

8.5 Mitigation

Given the marked variation in the acoustic parameters of the noise sources, and the different hearing abilities of the species discussed in this chapter, it is unlikely that a generic process of mitigation could be implemented. Furthermore, it is clear information on behavioural responses, masking potential, and injury risk for many species is limited, and further studies would be required to design a robust set of mitigation measures.

In a technical sense, reducing the potential impacts of devices, such as AHDs, on non-target species could be achieved through changing frequencies to those where non-target species are less sensitive, or by using responsive-mode devices that only emit sound when an animal approaches an area of interest. Similarly, changes in frequency of data transmission devices may help eliminate the potential risk to more sensitive species. However, it is clear that there will be implications for data transmission efficiency with changes in frequency. Noise pollution by AHDs could also be reduced by decreasing the duty cycle of the device. This would not only decrease the risk of causing hearing damage in target or non-target species, but may also decrease the likelihood of target species habituating to the sound. Similarly, it may be possible to use pingers that are triggered by echolocation activity of an approaching dolphin or porpoise, or reduce the duty cycle of pingers.

Reducing the potential impacts caused by noise produced by marine renewable devices may also be feasible at the design stage. Although it may not be possible to reduce noise through changes to individual turbines, it is important to reduce the risk of “acoustic barrier effects” or avoidance of important areas when designing the configuration of arrays of turbines. For example, it is important to ensure that narrow channels used as transit routes for animals are not fully occluded by turbines, or critical habitats are not used to site arrays of turbines.

Reducing the effects of ocean tomography or thermometry studies, and data transmission devices, may be possible by ensuring that the immediate vicinity around the sound source is clear of animals. For marine mammals, there are existing guidelines, *e.g.* from the Joint Nature Conservation Committee, for minimising the risk of acoustic disturbance from seismic surveys that may be appropriate for these sound sources. These precautionary guidelines suggest that, in addition to keeping noise levels at lowest practicable levels, practical measures are taken to ensure no marine mammals are within an area of risk. This is clearly more challenging for animals to that do not regularly come to the surface to breathe (*i.e.* fish).

Playing temporarily aversive sounds that causes animals to show a small-scale avoidance response up to a certain distance from the sound source may provide a means of reducing physical injury (hearing damage). This may be feasible for temporary noise activities like ocean tomography studies or acoustic data transmission. With all species, planning activities so that their timing will reduce the likelihood of encounters with breeding areas or juvenile animals, using the lowest practicable power levels throughout the survey, and seeking methods to reduce and/or baffle unnecessary frequencies from the devices will lead to reduced risk of injury, masking, and behavioural responses.

8.6 Conclusions

Avoidance behaviour by marine mammals can be expected to occur in response to several of the ‘other’ noise sources. However, ranges over which animals might be impacted will depend on the species, and the respective noise source. Large-scale habitat exclusion of odontocetes (a non-target species) has been demonstrated in response to commercially available acoustic harassment devices (AHDs) and should be considered a potential concern (if no mitigation measures are implemented). Although there is the potential that some pingers could exclude porpoise from their habitat, it should be highlighted that avoidance responses do not seem to extend over more than a few hundred metres from the device (3, 8). Therefore, the benefits of reducing lethal by-catch may outweigh the impact on behaviour in some populations and habitats. Due to their high frequency content, both AHDs and pingers are less likely to affect fish behaviour, except for individuals within the immediate vicinity of the device. Ocean research studies and low-frequency fish deterrents have a clear potential to influence the behaviour of large whales (*e.g.* baleen whales) and pinnipeds. However, with respect to ocean research activities, monitored responses during experiments did not always seem to be overt (1, 4). Similarly, some studies failed to show a clear effect of ocean research studies on the behaviour of fish. However, captive experiments on fish may indicate that some effects on behaviour (*e.g.* C-starts) could be expected over substantial ranges. Acoustic data transmission devices have been documented to elicit avoidance responses in captive odontocetes and pinnipeds at relatively low received levels. When extrapolating from captive experiments to wild animals, predicted impact zones would be large, possibly extending over several kilometres. However, further research is needed and these devices should not be implemented in areas of important habitat for pinnipeds or cetaceans. High-frequency or ultrasonic fish deterrents have the potential to influence behaviour of odontocetes in a similar way as AHDs. However, no research into this has been carried out to date. The current scarcity of data makes it difficult to predict impact of any of the noise sources on invertebrates or other non-mammalian animals.

With respect to masking, particularly broadband high source level noise sources may constitute a problem (*e.g.* data transmission devices, AHDs). AHDs and data transmission devices (acoustic modems) could potentially mask the communication signals of delphinids. Lower-frequency sound sources (*e.g.* ocean research studies, tidal) may cause masking of communication signals in fish or pinnipeds. Tidal turbines may mask communication signals for a variety of species in the vicinity of the device due to the broad frequency spectrum of the noise emission. This may be particularly evident for fish which do not have sophisticated strategies to counteract masking like some marine mammals.

Hearing damage may only be caused by some of the high intensity noise sources mentioned in this chapter in species with good hearing sensitivity. Depending on the assumptions made, predicted impact zones may vary markedly. Odontocetes exposed to a single emission of an AHD are unlikely to suffer hearing damage even when being close to the device. However, repeated exposure for extended amount of times (*e.g.* as a result of overlapping sound fields from different devices) may pose a substantial risk. Similarly, long-term exposure to some data transmission devices may pose a risk that needs to be taken into account. Impact of the low-frequency sound sources (*e.g.* ocean tomography) on baleen whales and fish may be limited to the vicinity of the device. However, particularly with respect to fish, the possibility of strong inter-species variation needs to be considered and more data is needed. Since no direct measurements of hearing damage or even hearing abilities in baleen whales are available the possibility of underestimating the risk should also be taken into account.

In conclusion several noise sources described in this chapter have the potential to impact aquatic life. Although general conclusions can be drawn from the available data, knowledge gaps in certain areas will inevitably mean that some predictions have to be based on assumptions that might be potentially controversial. Therefore, it will be important to consider impact for many of the aforementioned noise sources on a case by case basis taking species, habitat and density of noise producers into account. However, mitigation measures should be implemented wherever possible.

8.7 References

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