Fisheries and Oceans Canada
Science

Pêches et Océans Canada
Sciences

C S A S

Canadian Science Advisory Secretariat

S C C S

Secrétariat canadien de consultation scientifique

Research Document 2010/025

Document de recherche 2010/025

Pathways of effects of noise associated with aquaculture on natural marine ecosystems in Canada

Séquences des effets du bruit causé par l'aquaculture sur les écosystèmes marins naturels au Canada

Peter F. Olesiuk[1], Jack W. Lawson[2], and Edward A. Trippel[3]

[1]Marine Ecosystems and Aquaculture Division, Fisheries and Oceans Canada, Pacific Biological Station,
Nanaimo, BC, V9T 6N7

[2]Marine Mammal Section, Fisheries and Oceans Canada, Northwest Atlantic Fisheries Centre, St. John's,
NL, Canada A1C 5X1

[3]Fisheries and Oceans Canada, Biological Station, St. Andrews, NB, Canada E5B 2L9

ISSN 1499-3848 (Printed / Imprimé)
ISSN 1919-5044 (Online / En ligne)

TABLE OF CONTENTS

Correct citation for this publication:
La présente publication doit être citée comme suit :

Olesiuk, P.F., J.W. Lawson, and E.A. Trippel. 2012. Pathways of effects of noise associated with aquaculture on natural marine ecosystems in Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2010/025. vi + 64 p.

ABSTRACT

The purpose of this document is to examine the pathways of effects of noise associated with aquaculture on natural marine ecosystems in Canada. Aquaculture, like most maritime activities, generates sound as a by-product of ongoing operations, and sound may be associated with constructing, maintaining, and decommissioning aquaculture facilities. In some cases, intense sounds have been intentionally produced to deter predator (seal and sea lion) attacks. Water is an excellent medium for transmitting sound, which can propagate tens or even hundreds of kilometres from the sound source. Aquatic organisms utilize sound for communication and foraging, and some species have their best hearing sensitivities within the dominant frequency ranges of sounds produced by aquaculture operations. The use of sound and hearing sensitivity is particularly advanced in marine mammals, which probably serve as good indicators of the potential effects of noise. Relatively loud sounds can potentially invoke behavioural responses and avoidance, interfere with or mask communication and echolocation signals, and cause temporary or permanent hearing loss. As a result, noise can displace animals from their habitat or interrupt normal movement or migration patterns, adversely affect foraging and reproductive behaviour, and increase the risk of predation. Acoustic Harassment Devices (AHDs) used to deter seal and sea lion attacks at salmon farms have been shown to have far-ranging effects on non-target cetaceans, such as harbour porpoise and killer whales, which can be displaced large distances from where AHDs have been deployed. In contrast, pinnipeds (seals and sea lions) appear to habituate to these devices and may experience hearing loss through prolonged exposure or very close approach, such that AHDs are largely ineffective as long-term predator deterrents. AHDs could potentially disrupt the behaviour patterns of some fish that have specialized hearing apparatus, particularly clupeids like herring, but these effects have not been documented. Given their ineffectiveness as deterrents and far-ranging impacts on non-target species, including SARA-listed species, it is recommended that AHD use at fish farms be prohibited. Sounds, particularly from vessels and their sonar, are also routinely produced during normal operation of aquaculture facilities, and may have localized or transitory effects on aquatic animals and are contributing to the chronic problem of increasing levels of anthropogenic noise in the oceans. It is recommended that the industry adopt practices to minimize noise and propagation, especially within or near Ecologically and Biologically Sensitive Areas (EBSAs), and near the important habitats of SARA-listed species. Aquaculture might, in a few circumstances, produce more intense sounds if underwater blasting or pile-driving is used during construction and demolition. These intense sounds were beyond the scope of this assessment, but could potentially harm aquatic animals and should be managed the same as other industrial maritime activity. Ocean noise and its effects on aquatic ecosystems are not yet well known, making it difficult to predict the impacts and fully appreciate the ecological consequences of far-ranging behavioural responses to aquaculture noise. We advocate the continued monitoring of sensitive ecosystem components, particularly cetaceans, in the vicinity of aquaculture operations and vessel traffic.

RÉSUMÉ

Le but du présent document consiste à examiner les séquences des effets du bruit causé par l'aquaculture sur les écosystèmes marins naturels au Canada. Comme la plupart des activités menées en milieu marin, l'aquaculture produit des sons, qui constituent un sous-produit de ses activités courantes, notamment les activités de construction, d'entretien et de déclassement des installations aquacoles. Dans certains cas, des sons forts sont produits intentionnellement pour décourager les attaques des prédateurs (phoques et otaries). L'eau constitue un milieu excellent pour la propagation des sons, qui peuvent se faire entendre à des dizaines, voire à des centaines de kilomètres de leur source. Les organismes aquatiques utilisent les sons aux fins de communication et d'alimentation, et certaines espèces ont une acuité auditive optimale dans la gamme de fréquences dominantes des sons produits par les activités aquacoles. L'utilisation des sons et l'acuité auditive sont particulièrement perfectionnées chez les mammifères marins, et ceux-ci constituent probablement de bons indicateurs des effets possibles du bruit. Les sons relativement forts peuvent provoquer des réponses comportementales, dont un comportement d'évitement, nuire à la communication et à l'écholocalisation, et entraîner une perte auditive temporaire ou permanente. Par conséquent, le bruit peut forcer des animaux à quitter leur habitat ou à interrompre leurs déplacements normaux (p. ex. migration), nuire aux comportements d'alimentation et de reproduction et accroître le risque de prédation. Les dispositifs de harcèlement acoustique (DHA) utilisés pour décourager les attaques des phoques et des otaries contre les fermes salmonicoles sont reconnus pour avoir des effets à grande échelle sur les cétacés non ciblés, comme le marsouin commun et l'épaulard, qui peuvent s'éloigner considérablement des lieux d'utilisation de DHA. Par contre, les pinnipèdes (phoques et otaries) semblent s'habituer à ces dispositifs et peuvent subir une perte auditive à la suite d'une exposition prolongée ou d'une étroite proximité avec ces dispositifs, ce qui fait en sorte que les DHA constituent une mesure dissuasive généralement inefficace à long terme. Les DHA pourraient perturber le comportement de certains poissons dotés d'un système auditif spécialisé, en particulier les clupéidés comme le hareng (*Clupea harengus*), mais ces effets n'ont pas été documentés. Étant donné leur inefficacité comme mesure dissuasive et leurs effets à grande échelle sur des espèces non ciblées, y compris des espèces inscrites à la liste de la *Loi sur les espèces en péril* (LEP), il est recommandé d'interdire l'utilisation de DHA dans les fermes piscicoles. Des sons sont aussi produits régulièrement dans le cadre de l'exploitation normale des installations aquacoles, notamment par les bateaux et leur sonar. Ces sons peuvent avoir des effets localisés ou éphémères sur les animaux aquatiques et ils contribuent au problème chronique lié à l'augmentation du bruit d'origine anthropique dans les océans. Il est recommandé que l'industrie adopte des pratiques visant à réduire au minimum le bruit et sa propagation, en particulier dans les zones d'importance écologique et biologique (ZIEB), à proximité de celles-ci et à proximité de l'habitat important des espèces inscrites à la liste de la LEP. L'aquaculture peut produire des sons plus forts dans certaines circonstances, comme dans le cadre de travaux de dynamitage ou de battage de pieux aux fins de construction ou de démolition. Ces sons forts n'ont pas été étudiés dans le cadre de la présente évaluation, mais ils pourraient avoir des effets néfastes sur les animaux aquatiques et ils devraient être gérés de la même façon que d'autres activités industrielles en milieu marin. Les connaissances sur le bruit en milieu océanique et sur ses effets sur les écosystèmes aquatiques sont encore limitées, ce qui rend difficile la prévision et la compréhension des répercussions écologiques de réponses comportementales à grande échelle au bruit causé par l'aquaculture. Nous sommes en faveur de la surveillance continue des composantes vulnérables des écosystèmes, en particulier les cétacés, à proximité des sites aquacoles et des zones de trafic maritime.

1. INTRODUCTION

The impacts of aquaculture and aquaculture-related activities on natural ecosystems in Canada are currently under investigation. A national Framework for Aquaculture Environmental Management (FAEM) is being developed to provide the basis for a coherent national approach to support sustainable aquaculture in Canada. This framework addresses environmental effects of aquaculture and aquaculture-related practices on four components of Canadian aquatic ecosystems: fish health, fish communities, fish habitat, and water quality. For the purposes of this review, fish refers to finfish, marine mammals, shellfish, and other aquatic invertebrates. Stressors associated with aquaculture that may affect aquatic ecosystems include: alteration in light; release of chemicals and litter; release of pathogens; release or removal of fish; release or removal of nutrients, non-cultured organisms and other organic material; physical alteration of habitat structure; and noise. Identifying Pathways of Effects (POEs) of these stressors on aquatic ecosystem components, including development of state-of-knowledge descriptions of stressor-effects, and descriptions of risk, is a key component to developing sustainable aquaculture practices in Canada (DFO 2010a). This document focuses on the potential POEs of noise associated with aquaculture on natural aquatic ecosystems in Canada.

As Urick (1983) noted: "*Of all forms of radiation known, sound travels through the sea the best. In the turbid, saline water of the sea, both light and radio waves are attenuated to a far greater degree than that from the mechanical energy known as sound."* Sound is integral to the lives of most marine vertebrates, and many species have converged on utilizing sound as a particularly effective means of communication and orientation. Fish, marine mammals, sea turtles, and even some invertebrates have evolved functional and, in some cases, quite elaborate sound production and reception mechanisms (Tavolga 1964; Popper 1980; Watkins and Wartzok 1985; Popper and Fay 1999; Wartzok and Ketten 1999; Popper et al. 2003). For many marine animals, acoustic communication is central to social interactions such as mating and tending to offspring. Some species, such as dolphins, porpoises, and sperm whales, use sound to find their prey and sense their environment (Au 1993). Other aquatic animals listen for predators and prey, or employ sound to navigate in the vast, opaque oceans (Norris 1969; Payne and Webb 1971; Tyack 1998).

The ocean is not a quiet place. Sound is produced naturally by waves, precipitation, earthquakes, wind and animals, all of which contribute to the background (ambient) acoustic environment. Anthropogenic noise is generated as a by-product of other activities such as shipping, but increasingly powerful sounds are also being intentionally generated by humans for various purposes, such as seismic surveys for oil exploration. Ambient noise levels have been increasing over time (Andrew et al. 2002; McDonald et al. 2006). Concerns about the potential adverse effects of anthropogenic noise on marine life began to emerge in the 1970s (Payne and Webb 1971), and over the past several decades there has been increasing recognition, concern, and debate over the environmental effects of various anthropogenic sounds (Richardson et al. 1995; Popper 2003; Southall et al. 2007; Tyack 2008). While certain sounds may be inaudible or entirely benign to marine animals, noise can have various adverse effects ranging from subtle behavioural responses, displacement from important habitat, injury, and even mortality (NRC 1994, 2000, 2003, 2005; Richardson et al. 1995; Wartzok and Ketten 1999; McCauley et al. 2000; Cox et al. 2006; Nowacek et al. 2007; Southall et al. 2007).

Aquaculture, like most maritime activities, generates noise that could potentially impact aquatic animals and marine ecosystems (Taylor et al. 1997; Morton and Symonds 2002; Olesiuk et al. 2002; Wursig and Gailey 2002). Sound may be produced incidentally during routine operation of equipment such as aerators, feeders, generators, power washers, and by boat and vessel traffic associated with servicing aquaculture facilities. In some cases, intense sounds have been intentionally produced at fish farms in an attempt to deter depredation activities by pinnipeds. Occasionally, intense sounds may also be produced during construction or decommissioning of sites if pile-driving or explosives are utilized. In this report, we identify and assess the Pathway of Effects of noise associated with aquaculture in Canada on natural aquatic ecosystems.

1.1 OVERVIEW OF AQUACULTURE PRACTICES

The aquaculture industry in Canada dates back to the 1950s, with trout farming in Ontario, British Columbia and Québec, and oyster culture in New Brunswick, British Columbia and Prince Edward Island. The industry expanded in the 1970s with the successful development of salmon farming and mussel aquaculture. Aquaculture currently takes place in all ten provinces and the Yukon Territory. Production of Atlantic and chinook salmon, trout, arctic char, blue mussel, oyster and clam are well established. Several other species including halibut, sturgeon, tilapia, sablefish and scallop are at various stages of development. Aquaculture production in Canada increased more than four-fold from 40,000 tonnes in 1990 to as much as 170,000 tonnes in recent years, while farm gate value increased from about $200 million in 1990 to as high as $900 million in recent years (DFO 2010b).

Salmon farming is the mainstay of the aquaculture industry in Canada, accounting for over 70% of the production and 80% of the value of Canadian aquaculture (DFO 2010b). Atlantic salmon (*Salmo salar*) is the dominant species, but chinook salmon (*Oncorhynchus tshawytscha*) and coho salmon (*O. kisutch*) are also farmed. Culture methods reflect the anadromous history of salmon. In general, juveniles are raised in freshwater, land-based hatcheries. Smolts are transferred to saltwater cages for grow-out to marketable size (4-5 kg), a period lasting 16 to 24 months. Cages, generally clustered in groups of 10-12 and anchored in nearshore waters, consist of rigid frames from which nets are suspended to contain the fish and to protect them from predators. British Columbia accounts for about 70% of salmon production, with salmon farms distributed widely on the Sunshine Coast, west coast of Vancouver Island, the Discovery Islands, the Broughton Archipelago and the central coast around Klemtu. In Atlantic Canada, New Brunswick is the leading producer, with most farms situated off Charlotte County in the Bay of Fundy, and some salmon farming also occurs in Baie D'Espoir off the south coast of Newfoundland, and along the south shore of Nova Scotia.

Shellfish aquaculture accounts for about 20% of the production and 10% of the value of Canadian aquaculture (DFO 2010b). In BC, Pacific oyster and various species of clam are the dominant species, with activity concentrated in Bayne Sound south of Comox. On the east coast, farming of both Blue mussel and oyster is concentrated in Prince Edward Island, with both species also cultured in the other Atlantic Provinces and Québec. Seed (oyster or mussel larvae) may be collected from natural sources using collectors such as rope strung with cultch, or obtained from hatcheries. Mussels are stocked in mesh socks or sleeves and suspended in the water column from floating longlines (or from rafts in some areas), and grow to marketable size in 18-24 months. Oysters may be placed in trays or racks suspended from rafts or longlines or may be placed in bags on the bottom until they reach marketable size (36-48 months). Shellfish have generally been harvested manually, but is becoming increasingly mechanized as producers grow in size (DFO 2010b).

1.2 OVERVIEW OF NOISE PATHWAYS OF EFFECTS

Most maritime activity generates noise, and the aquaculture industry is no exception. Noise may be generated by aquaculture facilities, and by the vessels and aircraft that service them. In some cases, salmon farms have intentionally generated intense sounds to deter predators. Water is an effective medium for transmitting sound, which may propagate tens or hundreds of kilometres from its source. Many aquatic animals, particularly marine mammals, utilize sound for communicating, mating, foraging, or listening for predators. Noise can interfere with these normal life processes by masking animal sounds, causing temporary or permanent auditory damage, or invoking behavioural responses. This can potentially displace animals from their natural habitat, impede migration or movement patterns, disrupt normal behaviour, and ultimately adversely affect the animals' fitness or performance.

1.3 SCOPE OF REVIEW

1.3.1. Activities Causing Noise

A systematic survey or inventory of the types and prevalence of noise associated with aquaculture in Canada has not been conducted. Aquaculture noise can be broadly categorized as 1) noise produced incidentally as a by-product of routine operations and maintenance; 2) sound produced intentionally to deter predators; and 3) sounds that might occasionally be produced during construction or demolition of infrastructure. Operational noise is produced at aquaculture sites by machinery, generators, aerators, feeders, harvesters, pressure washers, and by the vessel and aircraft traffic servicing these sites. Seal bombs, cracker shells, and more recently powerful electronic Acoustic Harassment Devices (AHDs) have been used to deter seal and sea lion attacks. Construction or demolition might occasionally involve more intense sounds, such as pile-driving or underwater explosives. Table 1 provides a general overview of the sources of noise, and the characteristics and prevalence of sounds associated with aquaculture in Canada.

Aquaculture noise can be generated below or above the water and transmitted between the two mediums. The acoustic impedance of water is about 3,500 times that of air (Urick 1983), and the hearing of some aquatic animals in water extends into the infrasonic and ultrasonic frequencies. The focus of this report is focussed on the effects of waterborne noise, although we briefly mention the effects of airborne sounds on pinnipeds, which haul out on land to rest, moult, and reproduce.

1.3.2 Type/source of Information and Literature Used

There are no dose-response data for directly assessing the effects of aquaculture operational noise in Canada or elsewhere. Marine organisms are difficult to observe and monitor *in situ*, and conducting controlled experiments with captive whales, or even seals and fish, is a challenge. Responses to noise are often subtle, and the long-term consequences of these responses are not readily apparent. This necessitated a more generic assessment of potential pathways of effects of aquaculture noise based on the responses to, or effects of, sound in general, as described in the literature.

Inspired by recent efforts to establish noise exposure criteria for marine mammals (Southall et al. 2007), we attempted to adopt a similar approach for assessing the potential pathways of effects of aquaculture noise. Measurements have been made of the characteristics (intensity, duration, frequency) of the general types of noise associated with aquaculture operations (Table 1). Physical models have also been developed to predict how these sounds propagate through marine ecosystems. Combining information on the properties of sounds (Section 2.1) and propagation models (Section 2.2), we produced very generalized estimates of exposure levels for sounds associated with aquaculture. Laboratory and captive studies have been conducted to assess the hearing sensitivity of a relatively limited number of aquatic animals (Section 2.3.1). Laboratory experiments, simulation models, and field studies have also been used to assess the various effects of sounds, such as masking, hearing loss, behaviour responses, and physical injury (Sections 2.3.2 to 2.3.5). Synthesizing information on predicted exposure levels, hearing sensitivity, and expected responses of aquatic animals to sound, we make inferences as to the range and pathways of effects of aquaculture noise on natural aquatic ecosystems in Canada.

The emergence of powerful AHDs at salmon farms in BC. in 1993 prompted concern over their environmental impact on non-target species. In 1994, DFO in Pacific Region placed a moratorium on any further deployment of AHDs, and established an Underwater Sound Impact Steering Committee to investigate the issue, leading to a number of studies. In the first study, *in situ* measurements of the source strength, frequency spectrum and pulse characteristics were obtained for the two models of AHDs that had been deployed in BC (Haller and Lemon 1994). A second study assessed effects on the abundance and distribution of harbour porpoise, a non-target species, in controlled field experiments with an AHD (Olesiuk et al. 1995, 2002)[1]. Long-term cetacean monitoring programs also documented changes in abundance of dolphins and whales in the Broughton Archipelago during the period AHDs were being utilized (Morton 2000; Morton and Symonds 2002). The effects of AHDs have also been investigated on the east coast of Canada (Strong et al. 1995; Johnston and Woodley 1998; Johnston 2002) and other parts of the world (Taylor et al. 1997; Gordon and Northridge 2002; Wursig and Gailey 2002). These studies allowed a more focused assessment of the effects of AHDs than other aquaculture noise.

Construction and demolition of aquaculture facilities may, in rare circumstance, involve the use of pile-drivers or explosives. These generate intense sounds, as well as shock waves, that can have serious and far-ranging effects on aquatic animals (Yelverton et al. 1973; Yelverton 1981; Richardson et al. 1995; Dzwilewski and Fenton 2003; Madsen et al. 2006; Tougaard et al. 2009). However, since these activities are not normally associated with aquaculture, we considered their pathways of effects outside the scope of this assessment.

Vessel sounds associated with aquaculture operations would likely be in similar frequency and intensity ranges as those of commercial fishing and transport operations. For marine mammals at least, the effects of the sounds from these sources are usually transitory, or the animals can habituate to such sounds with regular exposure. However, the range of effects can be large, and cumulative effects cannot be ruled out for the louder vessels.

Literature sources used in the assessment were diverse. Where relevant studies had been conducted, we used peer-reviewed scientific documents identified through various collections and search engines. In many cases relevant information was obtained from reports and websites provided by provincial and federal governments, international agencies, as well as unpublished DFO files and archives. In a few cases, information on sound sources was obtained from health agencies or manufacturers.

---

[1] A third study investigating the use of bubble screens to constrain the spread of the AHD sounds was initiated, but not completed as the aquaculture industry considered the technique unfeasible.

2. LINKAGES BETWEEN ACTIVITY, STRESSOR AND EFFECTS

The potential for, and magnitude of, the effects of aquaculture noise on natural ecosystems in Canada is determined by the sources of sound (stressor), propagation patterns (exposure), and how aquatic animals respond to it (effects). Sound sources are characterized by their amplitude, duration, signal rise time, frequency range, and geographic prevalence (Section 2.1). Various factors affect propagation of sound including frequency, local topography, temperature, and salinity (Section 2.2). Responses of the aquatic animals include avoidance, disruption of normal life processes, temporary or permanent hearing loss, and in extreme cases acute damage to tissues and organs (Section 2.3). Ecosystem components that are affected may be marine mammals, and to a lesser degree finfish (Sections 2.4 and 2.5). The nature and extent of responses determine population-level impacts and ecosystem linkages.

2.1 SOUND SOURCE (STRESSOR)

The potential for noise to affect animals depends on the intensity of sound and other properties, such as whether the sound is continuous or pulsed, the duration and amplitude of pulses, and the rise time, frequency, and energy content.

2.1.1 Sound Intensity

Sound is the result of the propagation of a pressure wave through some medium. Sound intensity is the amount of energy passing through a unit area per unit time. Intensity is proportional to the square of sound pressure integrated over time, divided by the acoustic resistance of the medium, which is a product of density (ρ) and the speed (c) at which speed sound travels through it:

$$[1]\ \textit{Sound Intensity} = \frac{(Pressure)^2}{\rho \cdot c}$$

The relationship between sound pressure and intensity is thus dependent on the nature of the medium, and there are appreciable differences between air and water. Water is some 800 times denser (ρ) than air, and the speed of sound (c) is some 4.5 times greater (1530 ms^{-1} in water and 340ms^{-1} in air), such that the acoustic impedance of water is about 3,500 times as great as that of air. For sounds of equal intensity in the two media, the sound pressure level (SPL) in water will be about 60 ($3{,}600^{0.5}$) times that in air. This confounds direct comparison of sound levels in water and air.

The intensity of sound is conventionally measured in decibels. The decibel scale for intensity is ten times the log(base 10) of the intensity of the sound (intensity) divided by the intensity of some reference level of sound ($\text{intensity}_{\text{ref}}$):

$$[2]\quad \textit{Intensity (dB)} = 10 \log \frac{Intensity}{Intensity_{ref}}$$

Sound sensors (including the human ear) are typically sensitive to fluctuations in pressure (amplitude of the sound) as opposed to intensity. The decibel scale for sound pressure level (SPL), once again measured relative to some reference sound pressure level (SPL_{ref}), is:

[3] $SPL(dB) = 20 \log \frac{Pressure}{Pressure_{ref}}$

A logarithmic scale is used because of the wide range of SPLs and the enormous range in sensitivity of mammalian ears, and logarithms are a convenient scale for managing such large numbers. The use of logarithms also reflects the fact that humans seem to perceive the loudness of sound on an exponential rather than a linear scale. The multiplication factors (10 for intensity or 20 for SPL) are purely for convenience, ensuring an adequate degree of precision without resorting to decimals. Because dB values are measured relative to some reference level, they are only meaningful if the reference value is specified. The conventional reference value for sound pressure levels in air is 20.4 micro Pascals (μP), which is close to the minimum SP*L* detectable by the human ear. For underwater sound, a reference value of 1 μPa is generally used. The difference between the two reference levels is 20·log (20.4/1) = 26 dB, further confounding the direct comparison of waterborne and airborne sounds.

2.1.2 Pulsed Sounds

Two basic types of sound can generally be distinguished: (1) continuous or non-pulsed, and (2) impulsive or pulsed. Many of the most intense anthropogenic sounds (e.g., airguns, AHDs) are sharply pulsed. For non-pulsed sounds, the sinusoidal sound pressure waves are generally averaged and referred to as root mean square (rms). Until recently, this had been the metric conventionally cited in biological studies. However, the sharpness (rise and fall times of the signal) of pulsed sounds, also needs to be considered. Mammalian hearing is most readily damaged by transient sounds with rapid rise-time, high peak pressures, and sustained duration relative to rise-time (Thiery and Meyer-Bisch 1988; Dunn et al. 1991). For dolphins and porpoises (odontocetes) tested thus far, hearing impairment has occurred at lower exposure levels when the sound is pulsed rather than continuous (Finneran et al. 2002, 2005). Popper et al.[2] also noted that the peak pressure exerted by pulsed sounds will determine whether the ear and swim bladder of fishes are subjected to extreme mechanical stress. Madsen (2005) concluded that rms was not a suitable measure of exposure for pulsed sound, and instead advocated peak-to-peak (p-p) measurements or using the total sound energy flux (see Section 2.1.3). Peak-to-peak sound pressure is the algebraic difference between the maximum positive and maximum negative instantaneous peak pressure, and is typically 2-12 dB higher than averaged values (Madsen 2005).

While hearing loss and impairment may be more likely to be caused by sharp pulses, the hearing of animals tends to be less sensitive to very short pulses (<100 milli seconds (ms) for Beluga whales and <50 ms for harbour seals; Johnson 1968, 1991; Terhune 1988, 1989). Indeed, Terhune (1988) noted that for short pulses sensitivity was a function of the number of cycles irrespective of frequency.

2.1.3 Sound Energy Levels

As noted above (Section 2.1.2), the duration of a sound stimulus may also be important in determining the response to it or its effect. The sound exposure level (SEL) is a metric that integrates the squared-instantaneous sound pressure wave over some finite span of time, often

---

[2] Popper, A.N., T.J. Carlson, A.D. Hawkins, B.L. Southall, and R.L Gentry. Interim Criteria for Injury of Fish Exposed to Pile Driving Operations: A White Paper. Unpublished Report. 15 pp.

a 1-s period. It represents the level of continuous sound which, if it were to last for one second, would have the same total acoustic energy as the transient and is expressed as dB re: 1 μPa^2·sec (Popper et al.[2]). The SEL can be an extremely useful metric for assessing cumulative exposure because it enables sounds of differing duration, sometimes involving bursts of multiple sound pulses, to be compared in terms of total energy. The SEL has been used as a metric in dose-response modelling for humans (NIOSH 1998), and the available data suggests it is also an appropriate metric for determining hearing impairment in odontocetes (Finneran et al. 2002, 2005).

Of the various acoustic metrics (rms or peak-to-peak SPL, SEL) that could be considered in relation to noise impacts on animals, Southall et al. (2007) concluded that for marine mammals, because of inter-specific differences and the differing temporal patterns and pressure signatures of various sounds, that it was impossible to predict unequivocally which metric would be most closely correlated with the likelihood of injury or significant behavioural disturbance. Instead, they advocated that such variation and scientific uncertainty should be accounted for by using a dual-criteria approach based on both peak pressure and energy levels. Popper et al.[2] adopted a similar approach when developing exposure criteria for fish exposed to pile-driving.

2.1.4 Frequency

The frequency spectrum of the sound is also important for assessing it effects for two reasons. First, the sensitivity of marine animals varies with frequency (Section 2.3.1). Second, frequency affects how far a sound will travel in water, with higher frequency sounds being absorbed more rapidly than lower frequency sounds (Section 2.2.2). If a sound is a "pure tone" the changes in sound pressure levels will be in the form of a sine wave, and the rate at which it completes these cycles is the frequency. Frequency is typically measured in cycles per second, commonly referred to as Hertz (Hz). In practice, most sounds span a range or spectrum of frequencies, and a plot of sound energy levels by frequency – a spectrogram - is more informative than a single metric. Even sounds characterized by a narrow frequency range often generate harmonics, and in some cases animals may be more sensitive to these than the primary frequency.

2.1.5 Particle Motion

In addition to hydrostatic pressure, which is the component of sound normally measured, there is also a back and forth motion of the particles of the medium, on the order of nanometres, which can be expressed as the particle velocity, particle displacement or particle acceleration. Particle motion is aligned along a particular direction and is therefore a vector quantity. Many species of fish are more sensitive to particle motion than sound pressure (Popper et al. 2003). Although these fish may be responding behaviourally to particle motion, it is unclear if high levels of particle motion may damage the auditory system, or whether sound pressure waves are the main source of injury. Marine mammal sensitivity to particle motion is poorly understood, but appears to be functionally limited (Finneran et al. 2002) and is usually ignored (Southall et al. 2007). Given the paucity of information on particle motion, we use sound pressure and energy levels as a proxy for particle motion.

2.2 SOUND PROPAGATION (EXPOSURE LEVELS)

Although water is an excellent medium for conducting sound, the amplitude of sound to which an animal is exposed (RL = received level) will always be somewhat less than the source level

(*SL*) due to transmission losses (TL) between the source of the sound and the animal being exposed:

[4] $RL(\text{dB}) = SL(\text{dB}) - TL(\text{dB})$

Several mechanisms contribute to the attenuation of sound, including geometric spreading, absorption, scattering, reflection, refraction and ducting.

2.2.1 Spreading

Transmission losses are a function of distance due to the geometric spreading and dilution of the sound energy as it radiates over distance, r. Under idealized conditions, the sound energy will radiate equally in all three dimensions:

[5] $TL(\text{dB}) = 20 \log r$ (metres)

referred to as spherical spreading (Figure 1). In certain circumstances, sound can become trapped within a "duct" (e.g., a space between the surface and the seabed), in which case it propagates in two, rather than three, dimensions. This can occur when the water depth is less than the distance from the sound source (spreading will be spherical when the water depth is greater than the distance from the sound source). In such situations, the shape of spreading resembles a cylinder and the sound front radiates proportional to the circumference of a circle:

[6] $TL(\text{dB}) = 10 \log r$ (metres)

referred to as cylindrical spreading (Figure 1). In practice, spreading in coastal areas is often a hybrid of spherical and cylindrical models, and losses may be approximated by 15 log r (metres) (Figure 1) (Richardson et al. 1995). Regardless of the model, spreading losses are exponential, such that sound pressure levels initially drop abruptly with distance close to the sound source, but drop more gradually with increasing distance from the sound source (Figure 1). For every doubling of the distance from the sound source, the drop is about 6 dB for the spherical model, 4.5 dB for the hybrid model, and 3 dB for the cylindrical model (Figure 1).

These spreading patterns have several important implications for assessing the impact of noise on aquatic animals. First, sound levels drop off rapidly near the source, so harmful levels will be restricted to a small zone. Second, sound initially spreads spherically until it encounters the bottom, so any sounds generated near the surface (as in aquaculture) will attenuate to some degree before they can be reflected and propagated by cylindrical spreading. Third, as distance from the source increases, spreading losses become smaller, and other factors such as absorption (see Section 2.2.2), play a greater role in attenuation.

2.2.2 Absorption

As sound travels through a medium, some energy is absorbed and dissipated as heat. The rate of this type of absorption typically increases with frequency, and varies with salinity (and to a much lesser degree with temperature and acidity). In seawater, the absorption coefficient (α) is approximated by:

[7] α (dB per km) $= 0.036 \cdot \textit{frequency}\ (\text{kHz})^{1.5}$

up to about 50 kHz (D. Ross pers. comm., cited in Richardson et al. 1995; Ainslie and McColm 1998). Absorption in freshwater is negligible below about 30 kHz, and absorption of very high frequencies is largely independent of salinity (Figure 2) (Urick 1983, Pincock and Voegeli 2002). Thus, absorption is insignificant (<0.04 dB per km) at low to medium frequencies (<1 kHz), but becomes more significant with increasing frequency, with losses reaching 3.2 dB per km at 20 kHz and 36 dB per kilometre at 100 kHz. Absorption thus limits the potential for long-range propagation of higher frequencies through seawater.

2.2.3 Scattering

Particles and other objects suspended in water can reflect, refract, and diffract sound passing though it, causing further transmission loss. Such scattering is a function of the composition of the objects and their dimension relative to the wavelength of the sound. Rigid objects scatter sound most effectively if their size is of the order of a wavelength or more, such that higher frequency sound will be most affected. Air bubbles can absorb sound energy, particularly if the frequency wavelength is less than the circumference of the bubble (Clay and Medwin 1977, cited in Gordon and Tyack 2001). Suspended gas bubbles in surf zones or surface waters during storms can increase transmission losses, resulting in an “acoustic fog” (Gordon and Tyack 2001).

2.2.4 Reflection/Transmission

When waterborne sounds reach a boundary with a new medium, such as the surface or seabed, a portion of the energy will pass into the new medium and some will be reflected. The proportion that is reflected depends on the nature of the media on each side of the boundary and the angle at which the sound wave hits (the angle of incidence). The more perpendicular the angle, the greater the proportion of sound energy transmitted through the boundary. The air-water interface is a very good reflector of sound, with most of the sound energy reflected in calm conditions when the angle of incidence is greater than 13º from vertical, but greater transfer can occur when the surface is roughened by wave action (Richardson et al. 1995). Different bottom types vary in the extent to which they reflect sound, with harder bottoms tending to reflect more sound than softer sediments.

2.2.5 Refraction and Ducting

Just as light is bent when it passes between two media with different transmission properties, sound is refracted back towards the medium with the lower transmission speed. The speed of sound in the ocean increases with both increasing temperature and pressure. Under certain circumstances, sound can be trapped between the warmer surface waters and greater pressures at depth, and propagate in a cylindrical fashion without impedance of the surface or bottom. Ducting of sound in the open ocean is known as the deep sound channel. Payne and Webb (1971) calculated that fin whales (*Balaenoptera physalus)* producing loud 20 *Hz* vocalizations could theoretically hear each other over distances of thousands of kilometres within the deep sound channel, and their predictions have been confirmed with recordings of whale vocalizations with military hydrophones (Gagnon and Clark 1993; Clark 1994). This illustrates the potential for long-range impacts of low-frequency sound, but is generally not applicable to the higher-frequency noise associated with aquaculture in coastal areas.

Shallow water does not allow the same type of channelling of sound that can occur in deep water (Rogers and Cox 1988; Richardson et al. 1995). Mode theory predicts that large propagation losses can occur when the effective water depth is less than one quarter of the sound wavelength (λ). This effect is probably not an important factor for marine mammals, which tend to be most sensitive to higher frequencies and inhabit deeper water, but could mitigate the effects of noise on fish, which are more sensitive to lower frequencies, when water depths are less than a few metres. Propagation of sound in shallow water is also influenced by bottom-type, which often consists of water-saturated sediment that does not act as a discrete reflecting boundary.

The foregoing discussion provides some background on the factors affecting propagation of underwater sound (see Urick 1983). Sophisticated models based on mode or ray theory (e.g., RAYMODE; Erbe and Farmer 2000) have been developed for modelling the propagation of sound under specific conditions, but these models are beyond the scope of the general discussion within this paper. In general, the over-simplified spherical (surface and bottom absorbs all sound) and cylindrical spreading models (surface and bottom reflects all sound) (Section 2.2.1), with allowance for absorption (Section 2.2.2), provide reasonably good approximations for estimating potential exposure levels as a function of line-of-sight distance. Effects of scattering, reflection, transmission, and refraction introduce some variability from these simplified models, and in some circumstances create localized shadows or hot-spots.

2.3 POTENTIAL EFFECTS OF SOUND (RESPONSE)

2.3.1 Audibility

In order to be affected by a sound, an animal must be able to perceive it. When a sound is perceptible amidst background noise, it is considered to be audible. Audibility can differ from detectability in that a receiving system (ear or hydrophone) may detect a signal at some level even when it is incapable of meaningful perception.

Behavioural audiograms, which plot the minimum audible sound (50% detection threshold by attentive subjects) as a function of frequency, have been obtained from captive, trained animals using standard psychometric testing procedures. These laboratory measurements are made under abnormally quiet conditions (or very constant background noise levels). Terhune and Turnbull (1995) suggested adding 20 dB to these minimal levels to obtain a more realistic measure of the *in situ* certain detection threshold.

More recently, electrophysiological audiograms have been obtained by measuring small electrical voltages (auditory evoked potentials; AEPs) produced by neural activity when the auditory system is stimulated with sound. Yuen et al. (2005), Finneran et al. (2007), Schlundt et al. (2007) and Szymanski et al. (1999) have demonstrated that, with carefully-calibrated and repeated measurements, the two procedures can produce comparable detection thresholds in cetaceans. However, in many cases AEPs are more useful for assessing the *relative* sensitivity of an organism to different frequencies.

The hearing sensitivities of animals generally reflects the sounds they produce and use for various functions including communication, echolocation, etc. (see NRC 2003; see Figure 7). As a group, marine mammals have well-developed hearing that, depending on species, may extend into the infrasonic and ultrasonic frequencies well below and above the hearing of humans (Ketten 1998) (Figure 3). Some of the small porpoises and dolphins can hear ultrasonic frequencies up to 180 *kHz* (e.g., Ketten 2000). Direct measurements are lacking for the baleen whales (mysticetes), but based on their behavioural responses to low-frequency sound, the low-frequency sounds they make presumably for communication, and their cochlear morphology, it is believed that large whales can detect infrasonic frequencies as low as the tens of Hz, and up to at least 10 kHz, although recent studies suggest they may produce and hear sound up to perhaps 30 kHz. These sounds appear to serve predominantly social functions, including reproduction and maintaining contact, but they may also play some role in spatial orientation. The odontocetes produce sounds across some of the widest frequency bands that have been observed in any taxonomic group. Their social sounds are generally in the range audible to humans, from a few hundreds of Hz to several tens of kHz, but specialized clicks used to navigate, and to detect and perhaps corral or debilitate prey, extend well above 100 kHz. The pinnipeds also hear over a wide frequency, from about 75 Hz up to about 30 kHz in air, and up to about 75 kHz in water. Pinnipeds produce a diversity of sounds, both underwater and on land, though generally over a lower and more restricted bandwidth (from 100 Hz to several tens of kHz), that are used primarily in social and reproductive interactions. Based on their literature review of marine mammal hearing capabilities, Southall et al. (2007) recognized five functional hearing groups of marine mammals: (1) the mysticetes (baleen whales), designated as “low-frequency” cetaceans; (2) some odontocetes (toothed whales), designated as “mid-frequency” cetaceans; (3) odontocetes specialized in using high frequencies (i.e., porpoises, river dolphins, and the genera *Kogia* and *Cephalorhynchus*); (4) pinnipeds (i.e., seals, sea lions, and walruses) in water; and (5) pinnipeds in air (Table 2).

Hearing thresholds among fish are highly variable. Most teleosts lack specialized hearing organs, are only able to detect low frequencies below 500 Hz, including infrasounds below 20-40 Hz, and are commonly referred to as “hearing generalists” (Chapman and Hawkins 1973; Hawkins and Johnstone 1978; Mann et al. 1998; Sand and Karlsen 1986, 2000). However, some fish possess a gas-filled channel that connects the swimbladder to an ear structure containing dense otoliths (ear bones) that exhibit inertia when stimulated, allowing these species to sense sound over wider frequencies (Blaxter et al. 1981; Kenyon et al. 1998; Popper et al. 2003; Webb et al. 2008). Fish with these specialize hearing systems are commonly referred to as “*hearing specialists*,” which include the clupeids such as herring, sardines, anchovies, and American shad (Mann et al. 1997, 1998, 2001). Herring, for example, are able to detect frequencies up to at least 4 kHz (Enger 1967; Mann and Popper 1997; Mann et al. 2001). Relatively little is known about the hearing of sharks, which lack otoliths but possess calcareous structures known as octoconia that may serve an analogous function (Hueter et al. 2004). AEPs indicate sharks are most sensitive to 20-40 Hz infrasound, and may be able to detect sounds up to 800 Hz (Kritzler and Wood 1961; Casper and Mann 2006, 2007a, 2007b). Marine snakes and reptiles have auditory systems similar to their terrestrial counterparts, but little is known about their underwater hearing, although loggerhead and green sea turtles can hear frequencies of 250-750 Hz and 100-800 Hz, respectively (Bartol et al. 1999; Ridgway et al. 1969, Bartol and Ketten 2006).

Comparatively little is known about the hearing by marine invertebrates (NRC 2003), although a number of species have highly sophisticated structures, called statocysts, which resemble the otoliths of fish (Offutt 1970; Budelmann 1988, 1992). It has recently been shown these are used for hearing low-frequency sound in several species (Kaifu et al. 2008). There is also some evidence that a number of crustacean species, such as crabs, have statocysts that are somewhat similar to those found in cephalopods, although they evolved separately. While there are no data for hearing in marine crabs, semi-terrestrial species such as fiddler and ghost crabs can detect sounds and use sound for communication (see Popper et al. 2001).

Because hearing sensitivity among aquatic animals differ and vary over such a wide range (Figure 3), they would be expected to respond differently to sounds depending on their sensitivity to the frequency of the sound, and to respond less to frequencies outside their best hearing range. In humans, substantial improvement in dose-response models has been achieved by filtering broad-spectrum noise levels through equal-loudness functions, such as the 40-phon ("A-weighting") or 100-phon function ("C-weighting") equal loudness functions. These frequency-weightings take into account both the frequency bandwidth of human hearing and the loudness perception at different frequencies. The filter functions are inverted and normalized to 0 dB at 1000 Hz, and provides a method of de-emphasizing or discounting less sensitive frequencies. Nedwell et al. (2007) proposed an analogous dB_{ht} metric in which species-specific weightings are obtained from audiograms. While conceptually appealing, the dB_{ht} system is difficult to implement in practice due to the lack of detailed information on hearing sensitivities and behavioural responses for most aquatic species. Southall et al. (2007) proposed an M-weighting system for the five functional hearing groups of marine mammals. The M-curves were derived based on the lower and upper hearing thresholds for each functional group (Table 2), and provide a weighting system for de-emphasizing frequencies below and above the hearing thresholds (Figure 4). The M-curves were considered to be precautionary in that they tend to be flatter than the corresponding audiograms over the range of sensitive frequencies (i.e., frequencies near the lower and upper end sensitivity may not be de-emphasized sufficiently), but they do provide a method of de-emphasizing frequencies that are outside the hearing range.

2.3.2 Masking

Audiograms represent the minimum sound levels that are detectable 50% of the time under quiet conditions in controlled laboratory conditions. Terhune and Turnbull (1995) suggested adding 20 dB to these minimal levels to obtain a more realistic measure of the *in situ* certain detection threshold. Moreover, low-level sounds may not be discernible from ubiquitous "background" noise. In order to be discernible, a sound must exceed background noise by a level referred to as the critical ratio (CR). Minimum critical ratios are generally on the order of 20 dB near mid-frequencies, but tend to increase geometrically at higher frequencies (Figure 5).

Background noise levels vary with frequency (Figure 6). Shipping noise is predominant below 100 Hz, and wave action predominates at higher frequencies. At very high frequencies (>50-100 kHz) the thermal noise of water is dominant. Background noise tends to be 5-9 dB higher in coastal areas than the open ocean, and soniferous marine life (e.g., snapping shrimp) can add up to 20 dB, and heavy precipitation up to 15-20 dB (Urick 1983; Richardson et al. 1995; Pincock and Voigeli 2002; NRC 2003).

Background noise, regardless of whether it is natural or anthropogenic, can impede an animal's ability to detect sounds. For example, the minimum detection threshold at 10 kHz is about 70 dB (Figure 3), representing an *in situ* certain detection threshold of about 90 dB (Terhune and Turnbull 1995). However, the critical ratio for harbour seals at 10 kHz is about 23 dB (Turnbull and Terhune 1990), so sound levels would need to exceed ambient noise levels by that amount to be discernible. Jacobs and Terhune (2002) reported ambient sound levels at 9-11 kHz in the Bay of Fundy were 70 dB or greater for 5% of the time, which would raise the *in situ* certain detection threshold from 90 to 113 dB. As noted previously, sound pressure levels drop by 3 to 6 dB for every doubling of distance (Section 2.2.1), so the presence of ambient noise can greatly reduce the range of audibility of a sound source.

Anthropogenic noise may partially or entirely occlude animal signals, a process known as auditory masking. In general, signals of a given frequency tend to be masked by noise at adjacent frequencies, and considerable research has been conducted to determine the width of the masking band and its relationship with the CR. Masking bands tend to be narrow – less than

1/6th octave – at the most sensitive frequencies, but widen toward the lowest and highest audible frequencies. Given the biological importance of sounds among marine mammals, masking may adversely affect an animal's ability to function. For example, an animal's ability to navigate, communicate with conspecifics, find prey or detect predators may be compromised. To some extent, animals could overcome masking effects by increasing the frequency, amplitude or duration of their vocalizations in noisy environments, and there is evidence whales do so in noisy environments (Miller et al. 2000; Foote et al. 2004).

Fish also use sound for communication and to perceive their acoustic environment (Fay and Popper 2000), and anthropogenic noise could mask fish from hearing biologically relevant sounds (Popper 2003). Interestingly, herring appear to respond to biologically meaningful sounds (e.g.*,* echolocation clicks, calls and tail slaps of killer whales) more so than artificial sounds (e.g.*,* military sonar, pingers) of similar amplitude and frequencies (Wilson and Dill 2002; Doksaeter et al. 2009).

2.3.3 Behavioural Responses

This aspect of sound exposure effects is often the most problematic, particularly for long-lived and intelligent species such as marine mammals. Even to the same sound, behavioural effects can vary by species, sex, age, context, and experience.

Animals may respond to anthropogenic noise by changing their behaviour or avoiding the sound, especially if it causes discomfort or pain. Thresholds for discomfort and pain levels associated with noise have been determined only for humans; in this case the threshold of discomfort is a function of frequency, ranging from about 120 dB above the threshold of audibility at the most sensitive frequencies, to 68 dB above the threshold of audibility at the least sensitive frequencies, with the threshold of pain being about 20 dB above these levels (Kinsler et al. 1982; Awbrey and Thomas 1987). Given the lack of data on effects of intense sound on marine mammals in water, application of the human values should be regarded as highly speculative (Awbrey and Thomas 1987; Richardson et al. 1995).

Behavioural responses of animals to sound tend to be highly variable and context-specific (see Wartzok et al. 2004). Some sounds that are distinctly audible to animals may elicit no overt behavioural response, especially if it does not greatly exceed the minimum detectable level and is not increasing or fluctuating (Richardson et al. 1995). Animals sometimes can become habituated or tolerate sounds to even loud sounds, especially if they are continuous and do not infer with normal life processes (Bejder et al. 2009). Responses can be diverse, including avoidance, change in swimming and diving patterns, and physiological reactions such as bradycardia (abrupt slowing of heart rate) (Myberg 1990; Thompson et al. 1998; Costa 2003). The inability to observe an overt response does not necessarily indicate a lack of behavioural (or other) effect, as marine animals are exceedingly difficult to monitor underwater. Southall et al. (2007) summarized a range of behavioural effects for marine mammals ranging in severity from mild to severe (Table 3). Mild responses (ranked 0-3) were brief and consisted of minor changes in behaviour that are unlikely to be of consequence unless persistent or widespread; intermediate responses (ranked 4-6) were considered to have higher potential in affecting normal life processes such as foraging, reproduction and survival; and severe responses (ranked 7-9) were considered more likely to affect vital rates.

Knudsen et al. (1992) conducted laboratory studies in which sound frequencies invoked the greatest response in salmon smolt. He found that infrasound with frequencies lower than 20 Hz triggered evasive reactions more effectively than higher-frequency sounds. The experiments were followed by field tests on Atlantic salmon (Knudsen et al. 1994), several species of Pacific salmon (Knudsen et al. 1997), silver eel (Sand et al. 2000), and several species of cyprinids (Sonny et al. 2006). In all cases, intense infrasound played at close range invoked escape reactions. Fish with generalized hearing abilities have been observed to respond to vessel noise (Olsen et al. 1983; Vabø et al. 2002) and seismic airguns (Engås et al. 1996; Engås and Løkkeborg 2002). Herring, a species with specialized hearing structures, exhibited avoidance responses to killer whale playbacks (echolocation clicks in both the laboratory and wild (Wilson and Dill 2002; Doksaeter et al. 2009).

2.3.4 Hearing Loss

Animals exposed to intense sounds may exhibit an increase in hearing threshold (i.e., poorer sensitivity) following exposure, referred to as a threshold shift (TS). Factors influencing the degree of TS include the amplitude, duration, frequency, sharpness, and overall energy content of the noise exposure. Mild TS typically diminishes over time. If TS eventually returns to zero (i.e., hearing sensitivity returns to pre-exposure levels), it is referred to as a temporary threshold shift (TTS). If TS does not return to pre-exposure levels after a long period (on the order of weeks), the residual TS is referred to as a permanent threshold shift (PTS), and is considered an auditory injury. Causes of PTS in mammals include irreparable damage to the sensory hair cells, or exceeding the elastic limits of certain tissues and membranes in the middle and inner ears (Ward 1997; Yost 2000). Chronic exposure to sound, common in industrialized societies, can result in noise-induced PTS in humans with age (Kryter 1994), and presumably this condition also occurs in other animals.

For marine mammals, sound levels that cause modest TTS (generally <20 dB decrease in sensitivity) have been determined for several species of small odontocetes and pinnipeds. No data exist on exposures levels necessary to cause PTS. Southall et al. (2007) reviewed the available information on TTS in marine mammals, as well as the relationship between TTS and PTS in other mammals including humans, and the reader is referred to their paper for details. Based on their review, Southall et al. (2007) proposed the following criteria for hearing injury: a SPL of 230 dB re: 1 μPa p-p (SEL of 198 dB re: 1 $\mu Pa^2 \cdot s$ and 215 dB re: 1 $\mu Pa^2 \cdot s$ for pulsed and non-pulsed sounds respectively) for cetaceans, a SPL of 218 dB re: 1 μPa p-p (SEL of 186 dB re: 1 $\mu Pa^2 \cdot s$ and 203 dB re: 1 $\mu Pa^2 \cdot s$ for pulsed and non-pulsed sounds respectively) for pinnipeds in water, and a SPL of 149 dB re: 20 μPa p-p (SEL of 144 dB re: 20 $\mu Pa^2 \cdot s$ and 145 dB re: 20 $\mu Pa^2 \cdot s$) for pinnipeds in air.

Temporary threshold shifts (TTSs) have also been documented for various species of fish in response to pulsed sounds such as shooting of seismic airguns (Popper et al. 2005), military sonars (Popper et al. 2007), and simulated white noise (Smith et al. 2004). Popper et al. (2005) determined onset of TTS (i.e., physiological fatigue and not damage) for three species of fish[3] exposed to seismic airgun pulses to occur within a SPL range of 205-210 dB re: 1 µPa p-p, representing a SEL range of 180-189 dB re: 1 $\mu Pa^2 \cdot s$ (Popper et al. 2005, unpublished[2]).

2.3.5 Physical Injury and Mortality

Very intense sounds caused by explosives, airguns, and pile-driving can cause physical injury at close range. At close range, explosives may also produce shock waves, which propagate in a different manner than acoustical energy (Richardson et al. 1995). The auditory system appears

---

[3] The three fish species included a hearing specialist, the lake chub (*Couesius plumbeus*), and two generalists, the northern pike (*Esox lucius*) and the broad whitefish (*Coregonus nasus*).

to encompass the organs most susceptible to noise exposure, at least in humans (Ward 1997). The limited data for captive marine mammals exposed to various noises support a similar conclusion, suggesting that onset of TTS occurs at levels below those required for direct non-auditory physiological trauma (Southall et al. 2007). Experiments have been conducted with terrestrial mammals held underwater and exposed to underwater blasts, which resulted in intestinal and lung injuries (Yelverton et al. 1973; Yelverton 1981). The severity of injuries seemed most closely related to the positive acoustic impulse (mechanical impact produced by a short-term pressure pulse) (Richardson et al. 1995). Although it is unknown whether marine mammals would be similarly affected, the experimental data have been used to develop safe distance models for marine mammals (O'Keefe and Young 1984; Goertner 1982). Risk of injury for a given SEL is a function mainly of depth and decreases with body size, so small pinnipeds and cetaceans would be expected to be more vulnerable to blasting injuries than large whales.

Injury to hearing organs has also been documented in fish. Enger (1981) found that some sensory cells in the hearing organs of Atlantic cod (*Gadus morhua*) were damaged when exposed to continuous, high-amplitude pure tones 50-400 Hz at 180 dB re: 1μPa @ 1 m. Similar effects have also been observed in oscars (cichlids) several days after being exposed to 300 Hz at 180 dB re: 1μPa p-p @ 1 m (Hastings et al. 1996), snappers exposed to 20-100 Hz at 170-185 dB re: 1μPa p-p @ 1 m, the equivalent to an SEL 150-165 dB re: 1μPa^2·s (McCauley et al. 2003). In some cases the damage developed over days following exposure, and had not healed more than two months after exposure. Thresholds for injury from single pulses are considerably higher. Caltrans (2004 cited in Popper et al.[2]) found no evidence of injury to shiner perch or steelhead exposed to a pile-driver with SEL as high as 181-182 dB re: 1 μPa^2·s respectively. Based on a review of existing information, Popper et al.[2] proposed injury criteria for fish exposed to single pile-drive pulses of 187 dB re: 1 μPa^2·s or 208 dB re: 1μPa p-p. Based on models of data from experimental exposure to explosives, the threshold for mortality was estimated to range from about 193 dB re: 1 μPa^2·s for very small fish (0.01-0.1 gms) to 198 dB re: 1 μPa^2·s for larger fish (100-1,000 gms) (Hastings and Popper 2005). Interestingly, there is some evidence that fish, unlike mammals, can replace or repair sensory cells that have been fatigued due to sound exposure (Smith et al. 2004).

Experiments have been conducted with airguns at close range on zooplankton (copepods), mussels, amphipods (*Gammarus locusta*), and molluscs (flat periwinkle and edible periwinkle) (Kosheleva 1992). No significant harmful effects were observed at exposure distances of 0.5 m and greater from a single air gun with a chamber volume of three litres. In another experiment with early life stages of Dungeness crabs, the authors observed a reduction in survival of less than 10% for one stage (second ecdysis) of larval development. Christian et al. (2003) conducted similar experiments with snow crabs, which were subsequently monitored over a 12-week incubation period in the laboratory. Egg development stages exhibited distinct differences between the exposed group compared to control group for eggs exposed at a distance of two metres from a single, small air gun of 0.7 litre, but there was no indication of immediate or delayed mortality or other effects. Given the limited data available and complexity of these experiments, additional studies will be required to reliably assess the effects of intense sound on invertebrates.

Noise exposure may affect the vestibular and neurosensory systems. In humans, dizziness and vertigo can result from exposure to high levels of noise, a condition known as nystagmus (Oosterveld et al. 1982; Ward 1997; Halmagyi et al. 2005). Little is known about vestibular functions in marine mammals. However, there appears to be a temporal and spatial correlation between military exercises involving sonar and cetacean strandings, which so far has eluded explanation. Jepson et al. (2003) and Fernández et al. (2004, 2005) have hypothesized that lesions (gas and fat emboli) observed in beaked whales found stranded after military sonar exercises may have been caused by *in vivo* nitrogen bubble formation. *Osteonecrosis* (a chronic pathology of deep diving) has been observed in sperm whales and attributed to chronic nitrogen bubble formation (Moore and Early 2004), perhaps due to acute nitrogen embolism associated with decompression sickness avoidance behaviour. It has been suggested this may be a symptom of the diving cycles normally necessary to avoid decompression sickness not having been completed due to extended surfacing behaviour as part of an avoidance response. The linkage between acoustic exposure to sonar, bubble formation, and stranding events remains equivocal, but does highlight the gaps in our understanding of these issues.

2.4 EFFECTS OF MARINE MAMMAL DETERRENT DEVICES

2.4.1 Seal Bombs and Cracker Shells

Pyrotechnics such as “seal bombs” and “cracker shells” have traditionally been used to protect fishing gear and salmon net pens from seals and seal lion attacks. Surveys of salmon farms in BC during the late 1980s and early 1990s indicated that 30% of respondents had used seal bombs, although 85% of users had found them to be ineffective and had discontinued their use, and 15% found them to be very effective and continued to use them on a regular basis (Rueggeberg and Booth 1989; Tillipaugh et al.[4]).

Seal bombs are small explosive devices that typically detonate 2-3 m below the surface, producing a flash of light and a 30 ms impulse with a SPL of 220 dB re: 1 μPa p-p (Myrick et al. 1990b) and a *SEL* of 190 dB re: 1 μ $Pa^2 \cdot s$ (Awbrey and Thomas 1987). Most of the sound energy of seal bombs lies below 400 Hz, with peak SEL at 20 Hz (Figure 8). The peak frequency of these devices is near the lower end of the sensitivity curve for low-frequency cetaceans, but well below peak sensitivity and should be significantly de-emphasized (-20 to -40 dB) when considering effects on other cetaceans and pinnipeds (see Section 2.3.1 and Figure 4). These measurements fall about 8-10 dB below the criteria that Southall et al. (2007) proposed for auditory injury in the low-frequency cetaceans. Because of the relative insensitivity of other cetaceans and pinnipeds to these lower frequencies (Figures 3 and 7), they are probably most affected by the 200 Hz component of the energy spectrum, even though energy levels at these frequencies are about 15 dB lower (SEL = 175 dB re: 1 $\mu Pa^2 \cdot s$). The SEL at 200 Hz, when de-emphasized by -4 to -6 dB, falls about 27 to 29 dB below the criteria for injury for other odontocetes. However, Myrick et al. (1990a, 1990b), who did not incorporate a frequency-weighting system in their calculations, estimated that such a charge could cause injury when detonated within 1-3 m of a dolphin, and a human was killed by a similar device when it exploded ~0.3 m from his head (Hirsch and Ommaya 1972). The SPL at 200 Hz falls about 10 dB below the criteria for injury proposed by Southall et al. (2007) for pinnipeds in water.

Interestingly, teleost fish, including generalists like salmon and cod, have peak sensitivity near 100-200 Hz where much of the energy of sea lion bombs is centred. Given the measured SEL of 190 dB re: 1 $\mu Pa^2 \cdot s$ (Awbrey and Thomas 1987), it is unlikely the seal bombs could cause mortality of even small fish (Hastings and Popper 2005; Popper et al.[2]). In terms of the *SPL*, the measured level of 220 dB re: 1 μPa p-p (Myrick et al. 1990b) is somewhat higher than the 205-

---

[4] Tillapaugh, D., C. Brenton and B. Harrower. Predation on salmon farms in British Columbia - the impacts of harbour seals (*Phoca vitulina*) (The results on a 1991 survey). Commissioned by BC Ministry of Agriculture, Food and Fisheries. Unpublished Report 49 pp.

210 dB re: 1 μPa p-p that caused TTS (i.e., physical fatigue, but not permanent damage) in fish exposed to airgun pulses, indicating that seal bombs could potentially cause temporary hearing loss in fish within a range of about 3-40 m depending on whether spreading was spherical or cylindrical.

Seal bombs could potentially have far-ranging behavioural effects. Among the marine mammals, the low-frequency cetaceans are probably the most sensitive to the sounds produced by seal bombs. Audiograms are not available for the low-frequency cetaceans, but baseline background noise at 20 Hz is on the order of 70 dB 1 μPa p-p (Figure 6), so assuming a CR of 20 dB (Figure 5), the sounds associated with the explosion could potentially travel great distances (hundreds of kilometres) before blending into background noise. The *in situ* measurements given in Table 2 of Awbrey and Thomas (1987) seem to corroborate this, as the measured drop was only 12 dB from 116 metres (160 dB) to 926 metres (148 dB), indicative of cylindrical spreading (i.e., the expected drop would be 20 dB for spherical spreading and 13 dB for cylindrical spreading). However, the measured SEL and SPL of the seal bomb (190 dB re: 1 μ Pa^2·s and 220 dB re: 1 μPa p-p respectively; Awbrey and Thomas 1987; Myrick et al. 1990b) are just above and below the response criteria for significant behavioural effects proposed by Southall et al. (2007) for cetaceans (SEL of 183 dB re: 1 μPa^2·s and SPL of 224 dB re: 1 μPa p-p). For single-pulse noises like seal bombs, Southall et al. (2007) equated a significant behavioural response with a temporary threshold shift (TTS). It thus appears seal bombs would not cause temporary hearing loss, except perhaps at very close range. However, they could evoke behavioural responses, such as a startle reaction, at a much greater range, though such responses tend to be of less biological consequence where the stimuli are intermittent and infrequent.

Cracker shells (airborne explosives) are shot from a shot- or flare-gun, usually over the heads of pinnipeds in the water, and generally explode just after they hit the water. Awbrey and Thomas (1987) measured the sound levels associated with two types of cracker shells (fired from shotgun and flare pistol) and found them to be highly variable, with source SEL values extrapolated to range from 170-235 dB re: 1 μPa^2·s (mean 195 dB re: 1 μPa^2·s). The authors noted that cracker shells often explode just below (and occasionally just above) the surface, and thus radiate less energy into the water than seal bombs. In contrast to seal bombs, most of the energy of cracker shells is above 200 Hz, with considerable energy as high as 10 kHz (Figure 8), to which all the functional groups of marine mammals are sensitive. It is difficult to predict the effects of cracker shells given the variability of sound measurements. The highest SEL reported by Awbrey and Thomas (1987) (235 dB re: 1 μPa^2·s) is much higher than would be expected for a device containing several grams of explosive (see Figure 6.24 in Richardson et al. 1995). The mean SEL value (195 dB re: 1 μPa^2·s) is about 3 dB below the injury criteria Southall et al. (2007) proposed for cetaceans, but about 9 dB above the injury criteria proposed for pinnipeds. Thus, the cracker shells could potentially cause permanent hearing loss in a seal or sea lion at a range of about 10 metres.

As is the case with seal bombs, the noise associated with cracker shells are broadcast considerable distances and could potentially have far-ranging behavioural effects. The mean SEL measured for cracker shells exceeds the criteria for significant behavioural responses proposed by Southall et al. (2007) for single pulses in all the functional groups of marine mammals, by 12 dB for cetaceans and 24 dB for pinnipeds. These levels would be expected to cause significant disturbance (TTS) up to about 10 metres for cetaceans and 200 metres for pinnipeds. The cracker shells would be audible and could elicit more subtle behavioural responses (e.g., startle response) at distances on the order of tens of kilometres, but such short-term responses are likely to be of little biological consequence when they occur intermittently and infrequently.

The broad-spectrum sounds generated by cracker shells are within the hearing range of both generalist and specialist fish. The mean measured SEL of 195 dB re: 1 $\mu Pa^2 \cdot s$ exceeds the criteria for TTS proposed by Popper et al.[2] for pile-driving by 15 dB, suggesting that cracker shells could potentially injure fish at distances of about 5-40 metres depending on whether spreading was spherical (in deep water) or cylindrical (in shallow water). Again, behavioural responses could occur at greater distances, but these have not been assessed in fish.

2.4.2 Acoustic Harassment Devices

In recent years, pyrotechnic deterrents have largely been replaced with electronic acoustic devices. Electronic devices are more practical and safer to operate than explosives, and they can be run continuously to defend against attacks where the predator is not observed (i.e., attacks that occur at night or beneath the surface).

The earliest attempts to use electronically-generated sounds to deter marine mammals involved the playback of killer whale (predator) sounds. While somewhat effective in the short-term (Fish and Vania 1971; Cummings and Thomson 1971; Anderson and Hawkins 1978), animals quickly learned to ignore the recordings without the positive reinforcement (sightings of killer whales). In the early 1980s, the first Acoustic Harassment Devices (AHDs) emerged (Shaughnessy et al. 1981; Mate and Harvey 1987). These were powerful devices tuned to the most sensitive frequencies of seals and sea lions (12-17 kHz), and incorporated characteristics thought to be unpleasant: high-amplitude, pulsed, irregular sounds (Mate et al. 1987). Although initially somewhat effective, pinnipeds usually became accustomed or desensitized to the devices over time (Mate and Harvey 1987 op cit). In the early 1990s, a second generation of even more powerful AHDs emerged that produced sounds intended to be loud enough to cause discomfort or pain to seals. The first of these powerful AHDs was deployed in the Broughton Archipelago in 1993, and concerns over their effects were quickly voiced by local whale researchers (A. Morton, Echo Bay, BC, and J. Ford, Nanaimo, BC, pers. comm.). In 1994, DFO in Pacific Region put a moratorium on further AHD deployments, by which time about a dozen of the devices were in use (DFO, Pacific Biological Station, Nanaimo, BC, unpubl.). A survey in the Bay of Fundy (in Atlantic Canada) in 1996 by marine mammal researchers using a hydrophone found that 33 of 69 (48%) salmon farms in the Quoddy Region and two of nine (22%) salmon farms in the Grand Manan region were using AHDs (Johnston and Woodley 1998), prompting similar concerns over their impact on non-target species, particularly cetaceans (Strong et al. 1995; Johnston and Woodley 1998).

The most popular model of AHD on both the east and west coasts of Canada was the AirMar *dB Plus II* (AirMar Technology Corp., Milford, New England, USA). It generated a sequence of pulsed sinusoidal tonal bursts with a fundamental frequency around 10 kHz with a SPL of 191-194 dB re: 1uPa p-p[5] (Figure 9). Each pulse was about 1.4 ms duration with 40 ms spacing and generated in sequences of about 60 pulses lasting 2.5 seconds. The sequences were duty

---

[5] Jacobs and Terhune (2002) measured the output source level of an AirMar dB Plus II AHDs used in experiments at 172 dB re: 1μPa p-p , and two AirMar units (model not specified) deployed at two fish farms in the Quoddy region of the Bay of Fundy at 179 and 178 dB re: 1μPa p-p, which was much lower than the 191-194 dB re: 1uPa p-p measured for the same model by Haller and Lemon (1994). Similarly, Terhune et al. (2002) measured the output source level of an "older" Ferranti-Thompson 4X Special model at 166 dB re: 1μPa p-p, which was very much lower than the 235 dB re: 1μPa p-p measured for the Ferranti-Thompson AHD that had been obtained from the BC distributor (PRA Manufacturing) for testing by Haller and Lemon (1994).

cycled with a 2-second quiet period, and typically cycled across four transducers suspended off each corner of a net pen, each with a 2-second quiet period (Haller and Lemon 1994; Terhune et al. 2002; Lepper et al. 2004). A few salmon farms in BC used Ferranti-Thompson AHD (Ferranti-Thompson Sonar Systems, Dorset, UK). This device had a fundamental frequency of 38 kHz with SPL of 235 dB re: 1μPa p-p (Figure 9) (Haller and Lemon 1994). On the east coast, Johnston and Woodley (1996) also detected "Squeaker" AHDs that generated 16 pulses centred at 15 kHz every 75 seconds, and "Multi-tone" AHDs that generated a 22-second multi-frequency burst on a random duty cycle. Detailed measurements of the sound levels and characteristics are not available for the latter two models. Other models of AHDs have been used outside of Canada, including the *Seal Scrammer* by Ace Aquatic which, with a SPL of 193 dB re: 1μPa concentrated at 10 kHz, is comparable to the AirMar unit, and the DSMS-4 by Terecos, which at 185 dB re: 1μPa is the least powerful of the devices but has the most randomized pulse pattern.

Neither the AirMar or Ferranti-Thompson AHDs exceed the multi-pulse or non-pulsed injury criteria proposed by Southall et al. (2007) for cetaceans (230 dB re: 1μPa p-p), so the devices are unlikely to cause permanent hearing loss, except perhaps as the result of prolonged, chronic exposure. However, the Ferranti-Thompson AHD exceeds the multi-pulse safety criteria proposed for pinnipeds (218 dB 1μPa p-p), and may be capable of invoking injury (permanent hearing loss) within a range of 10-50 metres depending on whether spreading was spherical or cylindrical.

Most of the energy produced by the AHDs is well above the most sensitive hearing range of fish, and would not be expected to cause hearing injury.

The main environmental concern surrounding AHDs is the potential for far-ranging and/or chronic effects on non-target animals, particularly cetaceans. Field measurements indicated that the sounds spread in accord with a spherical or hybrid spherical-cylindrical spreading model when allowance is made for absorption and providing that the path is unobstructed by islets and land (Figure 10) (Haller and Lemon 1994; Terhune et al. 2002; Shapiro et al. 2009). Haller and Lemon (1994) made their in situ measurements of the AirMar AHD in the Broughton Archipelago, and Terhune et al. (2002) made their measurements near Grand Manan, and both areas are probably representative of the topography surrounding salmon farms. The measurements by Haller and Lemon (1994) were made at fixed stations spaced several hundred metres apart, and replicated measurements at each site were quite reproducible (Standard Error = 0.82 dB) (Figure 10 – top panel). In contrast, Shapiro et al. (2009) made measurements continuously from a boat as they travelled away from the AHD. Their data also indicates spherical spreading, but also shows a high degree of fine-scale variability (Figure 10 – bottom panel). They attributed the fine-scale variation to indirect, multi-path propagation resulting from surface-reflected and bottom-reflected rays (Wahlberg 2006; Shapiro et al. 2009) (see Sections 2.2.3, 2.2.4 and 2.2.5). Whatever the underlying cause, the fine-scale variability could have important biological implications, as it would make it difficult for an animal to discern the direction of the sound source without travelling some distance, hampering its ability to avoid the AHD (Lawson 2009).

The primary frequencies of the AHDs (10 *kHz* and 38 *kHz*) occur within the most sensitive range for all functional groups of marine mammals (Figures 3 and 4). Odontocetes, such as harbour porpoise and killer whales, are among the most sensitive species at these frequencies, with hearing thresholds on the order of 40 to 50 dB re: 1μPa respectively. Background noise levels in the Broughton Archipelago during field tests at 10 kHz were measured at 34 dB re: 1μPa

(Haller and Lemon 1994), which were typical of levels expected under calm conditions (Figure 6). In longer-term measurements, Jacobs and Terhune (2002) reported that ambient sound levels at 9-11 kHz in the Quoddy region of Bay of Fundy typically ranged (5th and 95th percentiles) from 45 to 70 dB re: 1μPa. Assuming a critical ratio on the order of 20-30 dB (Johnson 1968; Kastelein and Wensveen 2008) (Figure 5), and that the *in situ* certain detection threshold is 20 dB above the minimal detection threshold (Terhune and Turnbull 1995), and that sensitivity to the short-duration pulses (1.4 ms) would be 20 dB greater than longer-duration sounds, the AirMar AHD would be discernible from background noise at distances of 17-39 km under calm conditions depending on whether spreading followed a spherical or a hybrid cylindrical-spherical model, but the range would be reduced to 0.5-21 km under the ambient noise conditions measured in the Bay of Fundy. Indeed, during periods of heavy precipitation, background noise (Figure 6) would reduce the audible range to a few hundred metres. Using similar assumptions, the Ferranti-Thompson's 38 kHz, 235 dB re: 1 μPa signal would be audible at distances at distances of 8-11 km under calm conditions, or 4-8 km with increased ambient noise levels associated with wave noise (Figure 6). Although the source level of the Ferranti-Thompson model is greater, its higher-frequency sounds are absorbed more rapidly with distance than those of the AirMar model. Moreover, the higher-frequency signal from the Ferranti-Thompson model is not as affected by background noise, which tends to be lower at these higher frequencies (Figure 6).

Because the nature of multi-pulsed and pulsed sounds and the behavioural responses to them can vary so widely, Southall et al. (2007) did not provide discrete criteria for these categories, but instead provided summaries of the types and severity of responses that might be expected as a function of the sound amplitude levels. The AHDs generate repeating chains of pulses on a continuous basis, and thus have characteristics of both multi-pulse and non-pulse sounds. Moderate behavioural responses have been observed in low-frequency cetaceans at sound levels as low as 100-120 dB re: 1μPa rms, and for other odontocetes at sound levels as low as 80-100 dB re: 1μPa rms. The time-averaged (as opposed to peak-to-peak) SPL's were estimated at 174-183 dB re: 1μPa rms for the AirMar and 203 dB re: 1μPa rms for the Ferranti-Thompson models (Haller and Lemon 1994). For the AirMar AHD, the levels necessary to invoke moderate behavioural responses would extend to 0.5-17 km for the low-frequency cetaceans and 3.3-31 km for other cetaceans. For the Ferranti-Thompson AHD, the same levels would extend to 2.0-5.5 km for the low-frequency cetaceans, and 3.7-7.7 km for other cetaceans. Southall et al. (2007) noted only one study where low-frequency cetaceans exhibited extreme behavioural responses to multiple pulses, and that was for migrating humpback whales (*Megaptera novaeangliae*) responding to an airgun at 150-160 dB re: 1μPa rms (McCauley et al. 1998). The AirMar AHD would generate this level at a range of 5-160 m, and the Ferranti-Thompson AHD at a range of 120-1,000 m. It should be noted that the effects of the AirMar device, though not as loud, range further because the higher-frequency pulses are not absorbed as rapidly as the Ferranti-Thompson pulses.

Field experiments and long-term monitoring studies indicate AHDs have far-ranging effects on cetaceans. In an experiment in the Broughton Archipelago with an AirMar AHD, the device was activated for a three week period and compared to a similar control period (and the exposure and control periods were replicated three times). Daily scans were made to document changes in the abundance and distribution of harbour porpoise (Olesiuk et al.[6]; 2002). There was a sharp decline in abundance of porpoise during the three periods the AHD was activated (Figure 11a),

---

[6] Olesiuk, P.F., G. Horonowitsch, G.M. Ellis, T.G. Smith, L. Flostrand, and S.C. Warby. An assessment of harbour seal (*Phoca vitulina*) predation on outmigrating chum fry (*Oncorhynchus keta*) and coho smolts (*O. kisutch*) in the lower Puntledge River, British Columbia. PSARC Working Paper S95-10. Unpublished Report. 72 pp. + Appendix.

with porpoise being completely excluded within 500 metres, and sightings reduced by 90% in the most distant strata observed (2.5-3.5 km from the AHD) (Figure 11b). The maximum range of avoidance by porpoise could not be established because of channelized bathymetry of the study area and the limited ability to observe distant porpoises. Assuming that, as in humans (Kinsler et al. 1982; Awbrey and Thomas 1987), the threshold for discomfort in porpoise is about 90 dB above the threshold of audibility, the 500 metre zone of complete exclusion coincides with the predicted range of discomfort. The reason for avoidance at greater distances could not be determined, but the AHD may have masked sounds or otherwise interfered with the ability to function normally. Porpoise quickly returned to the area when the AHD was deactivated during control periods, suggesting the animals were responding to the sound, and not being displaced very far. In another study with harbour porpoise held in pens and exposed to various 12 kHz sounds, Kastelein et al. (2005) found that animals avoided areas with sound levels of 97-111 dB re: 1 μPa rms. Based on a hybrid cylindrical-spherical spreading model (Figure 10), this coincides with a distance of 1.2-19 km for the AirMa*r* device, and is consistent with the observed range of avoidance during the playback experiment. In shorter-term (2-hour) playback experiments in the Bay of Fundy, avoidance responses by harbour porpoise was also observed when an AirMar AHD was activated (Johnston 2002). In that study, most porpoise sightings occurred more than 1,000 m from the AHD when it was activated, but again the maximum range of avoidance could not be ascertained.

Long-term field monitoring studies indicate that other species of cetaceans were also displaced from the Broughton Archipelago when AHDs were in use (Figure 12). Number of killer whale sightings in the region within line-of-site of AHDs declined significantly when the AHDs were deployed in 1993, and remained depressed until a few months after they had been deactivated in 1999, upon which sightings returned to pre-AHD levels (Morton and Symonds 2002). The same authors noted no such decline in killer whale sightings in adjacent areas where AHDs were not being used. Morton (2000) also noted that sightings of white-sided dolphins (*Lagenorhyncus obliquidens*) in the Broughton Archipelago declined when the AHDs began to be deployed. Collectively, these studies indicate that a variety of cetaceans may be displaced from large regions where AHDs have been deployed (Strong et al. 1995; Johnson and Woodley 1998; Morton and Symonds 2002). These long-term monitoring programs indicate a lack of habituation, even over several years. On a more positive note, however, this implies that cetaceans remained sensitive to the AHD frequencies, rather than experiencing permanent hearing loss, and returned to the area within six months of the AHDs being deactivated (Morton and Symonds 2002). However, the sensitivity and response of cetaceans can vary widely among species. For example, playback experiments with lower-amplitude acoustic deterrent devices (pingers), which had previously been observed to elicit strong evasive behaviour in bottlenose dolphins (*Tursiops truncatus*), failed to elicit any similar behaviour in common dolphins (*Delphinus delphis*) (Berrow et al. 2008).

For pinnipeds, moderate behavioural responses have been observed at levels as low as 160-170 dB re: 1μPa rms for multi-pulsed sounds. For the AHDs, these sound levels could occur anywhere from 2 to 425 m from source. Unlike cetaceans (Johnston 2002; Olesiuk et al. 2002), pinnipeds appear to acclimate to AHDs and have been observed swimming and foraging within a few metres of the transducers (Mate and Harvey 1987), including those of fully-functional AirMar devices (Olesiuk et al.[6]; Brown et al. 2003). AHDs have also been reported as being ineffective in other regions (Jacobs and Terhune 2002; Sepulveda and Oliva 2005). Theoretically, the AirMar AHDs do not produce enough acoustic energy to cause acute, permanent hearing loss (Southall et al. 2007). However, field observations suggest seals experience at least temporary threshold shifts in hearing, as occurs in humans, and that chronic exposure may lead to permanent deterioration of hearing over time. Necropsies on dead seals exposed to such sounds in this pattern have not been conducted to assess this.

It is worth noting that AHDs seem to have limited efficacy at deterring nuisance seals. There is evidence of short-term success in blocking movements of seals up rivers (Graham et al. 2009), displacing seals from key foraging areas (Yurk and Trites 2000), or protecting fishing gear (Fjalling et al. 2006). However, longer-term trials indicate such benefits are temporary. In the Puntledge River, attempts to block seals from moving upriver (Olesiuk et al.[6]), or displace them from foraging below a barrier fence (Brown et al. 2003), were unsuccessful, and in both cases seals seen swimming and foraging within a few metres of the transducers of an *AirMar* AHD. Jacobs and Terhune (2002) also observed seals in close proximity to netpens with active AHDs. Licenses have been issued to most salmon farms to lethally remove nuisance seals when other deterrents fail (e.g., Jamieson and Olesiuk 2001). The deployment of AHDs in the Broughton Archipelago during 1993-99 did not appear to reduce the number of seals killed (Figure 13).

The 7-38 kHz energy spectrum of the AHDs is generally well above the hearing thresholds of fish, and would not be expected to be perceptible to salmon in pens or wild fish. As part of their initial feasibility studies of acoustic deterrents, Mate et al. (1987) conducted a number of laboratory tests on the behaviour and gamete viability of salmonids exposed to their prototype AHD (185 dB re: 1μPa p-p at 12 and 17 kHz). In tank tests, they observed no discernible change in the distribution of fish exposed to an AHD when it was activated at close proximity. They also collected eggs and sperm from fish exposed for 90 minutes at close range (1.5 × 1.5 × 1.5 m tank), and found no discernible effect on their viability compared to control groups (and most of the egg mortality in both groups could be attributed to fungal growth). They also concluded from neurological tests that jack coho salmon were most sensitive to 50 Hz and with an upper hearing limit of about 800 Hz, and during field observations they saw no evidence that salmon detected AHD as it pulsed or was cycled on and off.

Concern has been expressed over the potential for behavioural effects of AHDs on fish with specialized hearing apparatus such as juvenile herring which are captured in the Bay of Fundy weir fishery (Grand Manan Fishermens Association, pers. comm). Audiograms for herring and other clupeids indicate they are insensitive to such high frequencies (Enger 1967). In field studies, Doksaeter et al. (2009) saw no evidence that herring responded to 1-2 kHz sonar signals with a SPL of 127-197 dB re: 1μPa rms or 6-7 kHz signals at 139-209 dB re: 1μPa rms. Interestingly, herring responded to broader-frequency (800 Hz to 20 kHz) killer whale vocalizations at 150-160 dB re: 1μ*Pa* rms. In tank studies, Wilson and Dill (2002) observed subtle responses by herring, such as cessation of feeding activity and changes in swim speed and schooling behaviour, with playback of simulated echolocation pulses at 6-7 kHz at 157-169 dB re: 1μPa rms, but saw no discernible response to pinger pulses at 10-11 kHz with SPL of 133-145 dB re: 1μPa rms. It is possible that herring may respond to sound energy in the lower part of the frequency spectrum of AHDs, but any such response would be expected to occur at short distances from the source.

Based on the limited data available on the hearing sensitivities of turtles and invertebrates, it appears the 7-38 kHz energy spectrum of the AHDs is above their hearing thresholds.

The current generation of AHDs have attained the limits of sound pressure levels that can be generated with electronic devices, as transducers tend to cavitate at higher levels (although it would be possible to increase total sound energy levels by increasing the pulse duration and reducing the duty cycle period). However, a prototype Pulse Power Device (PPD) has been developed that generates pulses by discharging an electric arc between two electrodes in the water column. The prototype device is capable of generating pulses with 1 to 3 kJ of energy, which translated to about 240 dB re: 1µPa p-p @ 1m at an intermediate setting of 1.8 kJ. However, authorization to test the PPD at full power or assess the effects on marine mammals has not been authorized due to environmental concerns (NMFS[7]).

Terminology has not been used consistently for acoustic deterrents, and it is important to distinguish the high-power Acoustic Deterrent Devices (Seal Scarers) being assessed here from the low-power Acoustic Deterrent Devices (Pingers) sometimes used in other fisheries to acoustically mark gear in an effort to reduce bycatch (DFO 2006b). ADDs or Pingers are comparatively low-amplitude devices with source levels on the order of 130 – 150 dB re: 1µPa @ 1 m, so their effects tend to be less pronounced and more localized (Laake et al. 1998; Trippel et al. 1999), and we do not intend to discourage the continued use of these devices for mitigating bycatch.

2.5 EFFECTS OF OPERATIONAL NOISE

2.5.1 Small Vessels

With respect to operational noise, the small boats and vessels used to service aquaculture facilities are a likely source of noise-related impact for marine animals. Many aquaculture sites are only accessible by water, and small boats and vessels are used on a regular basis to transfer gear, supplies, personnel and cultured product. The types of vessels used for aquaculture have not been inventoried, but they range from small runabouts with outboards to tugboats, and occasionally perhaps larger vessels, some towing barges (Table 1). Noise is produced by motorized vessels as a result of propeller cavitation, engine noise, water turbulence, and hull bouncing. Vessel noise tends to be wide-spectrum with peaks ranging from <100 Hz to over 6 kHz (Figure 14a), and signal amplitudes usually rise in positive relation with vessel size. The low frequency sounds tend to blend with what is now ubiquitous background shipping noise and the higher-frequencies of 1-100 kHz probably have the most potential for affecting marine mammals (Richardson et al. 1995). Fish are generally insensitive to these higher frequencies, except for the hearing specialists such as herring.

Erbe (2002) modelled the potential effects of whale-watching boats, which are probably somewhat comparable to the small vessels used in aquaculture sites, on killer whales. Source levels of the whale-watching boats ranged from 145 to 169 dB re: 1µPa @ 1m, and tended to be higher with increasing size and speed. In comparison, large outboards can generate up to 175 dB re: 1µPa @ 1 m and tugboats typically generate anywhere from 146 to 170 dB re: 1µPa @ 1m (Richardson et al. 1995). None of these levels would be sufficient to cause injury, but would be audible and according to the criteria in Southall et al. (2007) could potentially invoke moderate behavioural responses in the low-frequency cetaceans at distances up to 10 kilometres, and up to several kilometres for the other functional marine mammal groups. Erbe (2002) predicted that the noise from fast whale-watching boats would be audible to killer whales at distances up to 16 km and could mask killer whale calls at distances up to 14 km. Holt (2008) also estimated that vessels could greatly reduce the range at which killer whales could detect a chinook salmon using echolocation (Figure 14b). Erbe (2002) also estimated that slow-moving boats could potentially cause a temporary threshold shift (TTS) in hearing of 5 dB after 30-50

---

[7] National Marine Fisheries Service (NMFS). Environmental assessment on testing a pulsed power generator to reduce California Sea lion depredation of gear and catch aboard an actively fishing charter boat off Southern California. Unpublished Draft Report.

min of exposure at a distance of 20 m, and fast-moving boats at a distance of 450 m. Assuming a 120 dB re: 1µPa @ 1 m threshold for behavioural effects, (Richardson et al. 1995) estimated that slow-moving boats would elicit a distinct behavioural response within 50 m and fast-moving boats within 200 m. Erbe's (2002) assumptions and predictions regarding behaviour are supported to some extent by observational studies, which indicated that killer whales responded to sound levels of about 116 dB re: 1µPa @ 1 m (Williams et al. 2002), and changes in killer whale behaviour have been observed at distances of up to 400 m from boats (Kruse 1991). Extrapolating data from humans, Erbe (2002) postulated that the superimposed noise levels of multiple boats circling or following whales were close to the critical level assumed to cause a permanent hearing loss over prolonged exposure, although that seems somewhat unlikely given the injury criteria proposed by Southall et al. (2007). The effects of vessel traffic associated with aquaculture may have less impact, as it would tend to be transient and, unlike whale-watching vessels, not focused in close proximity to cetaceans.

The sonar used on vessels also generates significant noise. Commercial sonar is used for fish finding, depth sounding and bottom profiling, and typically operate within a narrow frequency range of 400 Hz to 200 kHz, with sound pressure levels typically ranging from 210 to 235 dB re: 1μPa p-p @1 m, and sometime as high as 250 dB re: 1μPa p-p @ 1m (Richardson et al. 1995; NRC 2003). Even the portable depth sounders (fish-finders) used routinely on small boats can generate pulses in excess of 180 dB re 1 μPa p-p @ 1 m. For example, one popular model (Furuno Model LS-6000) operates at two frequencies, 50 kHz and 200 kHz, with output power on the order of 1 kW (201 dB re: 1µPa p-p @ 1 m), which is sufficient to invoke a moderate behavioural response in all functional groups of marine mammals (Southall et al. 2007). The 50 kHz sonar pulses coincide with the same frequencies of echolocation clicks produced by killer whales (Au et al. 2004), and could thus mask the ability to detect prey (see also Holt 2008). In practice, however, the potential impact of sonar is somewhat tempered by the fact the sound is directed in a narrow beam downwards, the high frequencies are rapidly absorbed in saltwater (α = 10 to 50 dB·km^{-1}), and the duty cycling is only 0.1% (0.2 ms pulses ever 0.2 sec) which greatly reduces the total SEL. As noted in Section (2.1.2), animals tend to be less sensitive to very brief pulses, and animals also tend to have higher Critical Ratios at these high frequencies (Figure 5). The 0.2 ms pulses of the Furuno sounder represents 10 cycles at 50 kHz and 40 cycles at 200 kHz, which would be expected to increase the threshold of sensitivity on the order of 10-20 dB (see Figure 2 in Terhune 1988). This makes it difficult to predict the zone of audibility of these brief, high-frequency pulses, but for the mid- and high-frequency cetaceans it could - if directed laterally - range as far as 5-7 km (50 kHz pulse at 201 dB re: 1μPa p-p, assuming the auditory threshold of 50 dB is reduced by 10 dB due to brevity of the pulses, and by an additional 10 dB due to the elevated CR).

2.5.2 Routine Operations

Routine use of generators, tools, pressure washers, pumps, aerators, and forced-air fish feeders all generate noise in air and through the water. Sound produced by diesel power generators increases with their size, ranging from about 86 dB re: 20μPa @1 m for 125 kW to 99 dB re: 20μPa @ 1 m for 2,000 kW, with most of the energy in the 100 Hz to 2 kHz range (ASHRAE 2002). Small gas engines like chain saws, compressors, pressure washers, and lawn mowers produce sound levels of 90-110 dB re: 20μPa @ 1 m (NIDCD 2009), and this also tends to be broad spectrum noise. Some of this sound can be transmitted directly and quite efficiently though infrastructure (from docks to pilings) or, depending on the angle of incidence and wave action, transferred through the air-water interface. Just as a portion of waterborne sound is reflected back into the water at the air-water interface, a portion of airborne sound is reflected back into the air, and this proportion increases with angle of incidence and decreases with increasing wave action. Under calm conditions, virtually all acoustic energy is reflected if the angle of incidence is less than 13º from vertical, but some energy can be transmitted into water at lower angles when there is wave action (Richardson et al. 1995). This explains why sound travels so well over water, especially on a calm day. Note, that airborne sounds are conventionally measured with a reference of 20 μPa, and are thus 26 dB (20·log[20/1]) higher than waterborne sounds measured in reference to 1 μPa. Measurements of overall noise levels in land-based aquaculture tanks indicate a wide-spectrum of noise, with a low frequency peak at 25-250 Hz attributed to water flow, pumps and machinery vibrations transmitted to the fibreglass tank, and a peak of high-frequency noise at 630-2,000 Hz attributed to electric generators, electric air and water pumps, and oscillating air bubbles (Bart et al. 2001).

None of the noise levels associated with routine aquaculture operations appear to be sufficient to cause injury, but could invoke behavioural responses at close range. Indeed, species like harbour porpoise at an aquaculture site in the Bay of Fundy have been observed to be displaced temporarily by noisy activities such as cage cleaning with pressure hoses, but returned quickly (within 10 minutes) to the area when the disturbance ended (Haar et al. 2009). Wysocki et al. (2007) found no adverse effects of sound on rainbow trout in aquaculture tanks, which are probably noisier than open ocean pens. Because most airborne sound is reflected off water at shallow angles, most of the acoustic energy entering the water from above the surface would be directed downwards, such that any effects would tend to be localized.

Airborne sounds can also disturb pinnipeds at haul-out sites. Steller sea lions are particularly vulnerable to disturbances at rookeries during the summer breeding season when males are actively defending territories and pups are too young to swim (DFO 2009), but sea lion breeding sites are located in exposed, offshore areas where aquaculture is not practiced. Some pinnipeds, like harbour seals, can become habituated to noise and other disturbances, and haul out on a regular basis in urban areas or near industrial sites, such as on the log booms at paper mills and adjacent to airports.

2.5.3 Construction and Demolition

In rare circumstances, explosives or pile-driving may be required during the construction or demolition of infrastructure associated with aquaculture. The percussion drivers used to install large-diameter steel piles can generate broadband underwater sounds (up to 100 kHz with peak energy at 160 Hz) with source levels as high as 235 dB re: 1μPa p-p @ 1 m (Tougaard et al. 2009). These sounds are sufficient to invoke injury in all functional groups of marine mammals at close range (Madsen et al. 2006; Southall et al. 2007) and potentially invoke behavioural responses at considerable distances. Indeed, harbour porpoise have been observed to respond to pile-driving at distances of at least 20 km (Tougaard et al. 2009). Underwater explosives can generate very intense sound pressure waves, as well as shock waves, which complicates an assessment of it effects on organisms (Richardson et al. 1995; Dzwilwski and Fenton 2003). Because these activities are not normally associated with aquaculture, their pathways of effects were considered to be outside the scope of this review, and should be considered in the broader context of maritime construction.

2.6 MITIGATION OF STRESSOR OR EFFECTS

2.6.1. Elimination of Noise Sources

Sound energy disappears when the source is eliminated. For the most part, the biological effects of aquaculture noise are also transient, except perhaps for permanent hearing loss that could result from short-range or extended exposure to the intense sounds produced by some marine mammal deterrents, and explosives used in demolition operations. Field studies indicate that animals displaced by AHDs (including known individual killer whales) return when the devices are turned off (Morton and Symonds 2002; Olesiuk et al. 2002).

Section 7 of the Marine Mammal Regulations prohibits the disturbance of marine mammals except while fishing for them. Disturbance has been broadly interpreted as activities interfering with normal life processes, such as resting, travelling, foraging, avoiding predators, communicating, socializing, mating, and caring for their young (Lien 2001). The notable displacement of harbour porpoise and killer whales from their natural habitat by AHDs constitutes disturbance under this definition.

Recent amendments to the Marine Mammal Regulations allow for the lethal removal of nuisance seals that represent a danger to fishing equipment (including aquaculture sites) despite deterrence efforts. While AHDs appear to be somewhat effective as deterrents in the short-term, they are largely ineffective in the long-term, and there is no evidence they alleviate the need for lethal removal of predators. AHDs should not be considered as acceptable or viable deterrents at fish farms, and should be prohibited.

2.6.2 Mitigation Measures to Minimize Noise Effects

Measures can be taken to reduce the effects of aquaculture noise. Bubble curtains or screens have been used to impede the propagation and contain intense sounds from point sources, such as the noise associated with blasting and pile-driving (Wursig et al. 2000). Preliminary field tests were conducted to assess the feasibility of a bubble screen for constraining AHD signals. While initial tests showed promise for impeding transmission of AHD sounds (D. Farmer, Institute of Ocean Sciences, Sidney, BC, pers. comm.), it was also calculated that the amount of power and size of compressors needed to implement an effective bubble screen for a typical fish farm would be impractical and prohibitively expensive.

Predator deterrents such as AHDs might be more effective and environmentally acceptable if they could be activated only when predator attacks were occurring. It would be expected that seals would be more likely to respond and less likely to habituate if the AHD (negative-reinforcement) was only administered when seals were actively attacking fish farms (Pryor 1987). Moreover, the behavioural responses of non-target animals like cetaceans tend to be of less biological significance when the disturbance occurs intermittently and infrequently (e.g., short-term startle response versus long-term displacement from habitat) (see Southall et al. 2007). During their use in BC, operators were requested to use AHDs judiciously (i.e., activate them only when predator attacks were occurring) until an environmental review could be conducted. However, this proved impractical as most seal attacks occurred underwater or at night, and the AHDs were operated throughout the night and often throughout the day. Surveys indicated a similar pattern of usage in NB (Johnston and Woodley 1998). DFO in Pacific Region requested the manufacturer of the most popular device (AirMar Technology Corp., Milford, New England, USA) develop an automated triggering mechanism, but attempts to develop such a system by detecting sonar images of predators or their underwater vocalizations were unsuccessful. One manufacturer (Ferranti-Thompson Sonar Systems, Dorset, UK) offered a mechanical system in which chimes were attached to nets, which rang and activated the AHD when seals pushed the net, but wind and wave action also rang the chimes and triggered the AHDs. In addition to being difficult to implement, an automated triggering system would abruptly expose animals to intense pulses at close range, (as opposed to a continuous signal whose received strength that would increase slowly as the animal approached), increasing the risk of auditory injury. AHDs could be designed to ramp up slowly to avoid auditory injury, but this might give seals time to remove fish and motivate further depredations, and increase the likelihood of habituation.

Measures can also be taken to minimize operational noise. For example, sonar and depth sounders can be turned off when not required, especially when cetaceans are in the general area. Noisy activities, such as pressure washing, can also be suspended whenever marine mammals are in the immediate vicinity. For airborne noises, transmission directly into the water can be minimized by acoustic decoupling; i.e., mounting equipment such as generators, compressors and pressure-washers on rubber pads or tires.

3. INTERACTION WITH OTHER STRESSORS ASSOCIATED WITH AQUACULTURE

While largely ineffective at deterring nuisance seals that are habituated to feeding at finfish farms, AHDs could have some beneficial effect in deterring naïve animals that may approach finfish farm in search of new foraging opportunities. This could potentially reduce the number of nuisance seals, thereby reducing the likelihood of net pen damage and escape of cultured stock into the wild. Although detailed statistics are unavailable, the persistence of seal attacks and continued need for the lethal removal of nuisance seals during the extended period of AHD use in BC suggests any such benefits are minimal.

In their review of predator control at salmon farms in BC, Jamieson and Olesiuk (2001) noted that one of the major companies that had been deploying AHDs subsequently attained some success in deterring seal attacks with predator nets. The use of Flexgard 11 (Flexabar-Aquatech Corporation, Lakewood, N.J., USA), a particularly heavy water-based latex, copper oxide anti-foulant, stiffened the nets and was considered a key to the success Stolt Sea Farms has had in mitigating predator attacks (Gary Robinson, Stolt Sea Farms, pers. comm., cited in Jamieson and Olesiuk 2001). Thus, eliminating AHDs may contribute to an increase in the release of chemicals by aquaculture into natural aquatic ecosystems.

4. CONCLUSIONS AND RECOMMENDATIONS

4.1 POE DIAGRAM

The pathways of effects of aquaculture noise on natural aquatic ecosystems in Canada are illustrated schematically in Figure 15. Noise associated with aquaculture can be categorized as sounds generated as a by-product of routine operations, sounds produced for deterring predators, and perhaps occasionally sounds made during construction and demolition. Aquaculture noise can be classified as single pulsed or intermittent (acute) sounds, or as multi-pulsed or continuous (chronic) sounds. Noise propagates through the water, exposing aquatic animals to sounds that are audible or, in the case of acoustic deterrents or construction noise, potentially harmful. The effects of audible sounds include short-term behavioural responses, avoidance of noisy areas, masking of animal sounds, and temporary hearing loss. Short-term behavioural responses are generally of little biological consequence, except where animals are particularly vulnerable (e.g., pinniped pups or cetacean calves may become separated from their mothers, or neonates may be trampled or driven into the water during disturbances), but animals often habituate to such disturbances. Animals may avoid noisy areas, or the noise may mask the sounds used by animals to communicate, forage or navigate. Such noise affects an area but not the animal *per se* (which can continue to function normally if they move to a less noisy area or the noise stops), so in essence these effects can be regarded as impacting habitat or water quality. Noise can also result in temporary or, in the case of acoustic deterrents and construction noise, permanent hearing loss. This directly affects an animal, and performance can be adversely affected, at least for some period of time, even if they move out to a less noisy area or after the noise stops, so these effects can be regarded as impacting the health of aquatic animals.

4.2 SUMMARY OF EFFECTS

This assessment indicates that noise associated with aquaculture is unlikely to cause permanent injury to aquatic animals, except for marine mammals exposed to pyrotechnic deterrents or the most powerful AHDs at very close range or over extended periods of time.

The main pathways of effects of aquaculture noise are the result of behavioural responses that disrupt the normal life processes of marine mammals, and possibly some species of fish such as herring. Information on the amplitude and properties of the sound pulses generated by AHDs indicate they would be audible to marine mammals over great distances, sometimes ranging over tens or several hundred kilometres. Some species, such as the seals and sea lions at which the sounds are targeted, appear to habituate, possibly as a result of hearing loss. However, a number of non-target species, such as porpoise, dolphins, and whales, are more sensitive and avoid areas where AHDs are being used. Given the large regions and travel corridors from which cetaceans can be excluded, these effects should be regarded as population-level impacts. Impacted animals include SARA-listed species, such as killer whales (Pacific Northern and Southern Resident populations designated as Endangered and Threatened) and harbour porpoise (Pacific and Atlantic stocks designated as Special Concern), and noise has been specifically identified as threats in recovery strategies and management plans associated with each. The killer whale is the ocean's apex predator (Ford and Ellis 1999) and cetaceans like harbour porpoise are an important dietary component of transient killer whales in fjords (Morton 1990; Ford et al. 1998) where aquaculture sites are often situated. The displacement of non-target cetaceans by AHDs can thus impact sensitive species and disrupt marine ecosystems.

Noise is also produced as a by-product by the vessels and machinery routinely used for aquaculture. For the most part, the effects of this noise are expected to be localized and short-term, but it is contributing to the broader problem of increasing background noise levels in the ocean. Simple measures, such as turning off sounders when not required, acoustic decoupling of machinery, and suspending noisy activities when marine mammals are present, can minimize these effects. In some circumstances, even seemingly benign disturbances (flushing seals off a haul-out site) can have serious consequences (separating mother-pup pairs), so it is important to minimize noise disturbances in sensitive areas and of sensitive species.

In rare cases, explosives or pile-driving may be required for construction or demolition of aquaculture facilities, which could generate intense sounds that could injure and disrupt aquatic animals. However, these activities are not normally associated with aquaculture and were considered to be outside the scope of this review.

4.3 SUMMARY OF KNOWLEDGE GAPS AND RECOMMENDED RESEARCH

While recognized as a growing concern, assessing the impacts of ocean noise on marine mammals and other aquatic organisms is an evolving science. In earlier assessments, noise issues were often dealt with by setting rigid exposure criteria (e.g., NMFS[7] guideline of 180 dB re: 1uPa rms limit for pinnipeds and odontocetes and a 190 dB re: 1uPa rms limit for sperm whales and mysticetes exposed to seismic airguns). While setting such limits might ensure animals are not exposed to intense sounds, rigid limits are unrealistic and impractical to apply in the broader context of other anthropogenic and natural noise sources. Not only the amplitude, but also the frequency, sharpness, duration, and other characteristics of the sound source need to be considered. Even off-the-shelf fish finders can produce pulses in excess of 200 dB re: 1μPa p-p @ 1 m, but their pulses are brief, narrowly focused, rapidly absorbed, and often outside the most sensitive hearing range of aquatic animals. Fish-finder pulses are not equivalent to the similar-amplitude pulses produced by AHDs, which are longer in duration and produced in rapid bursts, omni-directional, propagate to greater distances before being absorbed in seawater, and are tuned to the most sensitive hearing frequencies of many aquatic animals.

Some progress has been made in developing a more meaningful framework for assessing noise impacts (e.g., Southall et al. 2007). There is still some debate as to how the extent of injury or severity of behavioural responses relates to the total sound energy level, sharpness and duration of pulses (NIOSH 1998; Finneran et al. 2005; Madsen 2005; Popper et al.[2]; Kastak et al. 2007). The dB_{ht} and M-weighting systems have been proposed for dealing with the varying sensitivity of animals to different frequencies, but more species-specific data will be needed to implement the dB_{ht} weightings, and the M-weightings are probably too flat over too broad a spectrum, making them overly conservative. Good data is lacking on the relationship between temporary and permanent hearing loss in marine mammals, and the extent to which permanent damage may result from chronic, long-term exposure to low-amplitude sounds. Little is known concerning the danger of short-duration sounds with high pressure levels, even in humans.

In the absence of data for marine mammals, we have in some cases drawn inferences or based assumptions on humans or other terrestrial mammals, a practice that is somewhat questionable. Marine mammals evolved from terrestrial, air-adapted ancestors (Domning et al. 1982; Barnes et al. 1985) and have retained some semblance of the mammalian tripartite peripheral auditory system. Most of the basic mechanisms of mammalian hearing have been conserved, but marine mammal auditory systems also exhibit specialized adaptations related to pressure, hydrodynamics and sound reception in water (Wartzok and Ketten 1999). While it is probably reasonable to apply some of the general principles of mammalian hearing - such as integrating sound energy levels over time and employing frequency-weighting functions - it is less clear to what extent more specific parameters like critical ratios and threshold shifts can be extrapolated from terrestrial to marine mammals. Necropsies of marine animals that have experienced intense or chronic exposure to loud sounds (e.g., pinnipeds habituated to AHDs) would be valuable for assessing permanent hearing injuries.

The fundamental problem with ocean noise is that it propagates well through water, and some marine animals are very sensitive to it. Aquaculture noise typifies this problem. On the one hand, we have concluded that aquaculture noise is generally not intense enough to cause injury to aquatic animals, except for the loudest deterrent devices at close range. On the other hand, we also estimated that aquaculture sounds could be broadcast over great distances under ideal conditions, and may be audible to marine mammals tens of kilometres away. This could result in enormous zones in which animals are unlikely to be physically harmed, but may be behaviourally impacted.

Behavioural responses are hard to predict and vary widely in severity, making it difficult to interpret their consequences on the performance or fitness of affected animals. Some species, like harbour porpoise, seem shy and avoid motorized vessels (Amundin and Amundin 1973; Polacheck and Thorpe 1990; Barnes 1999), whereas other species like Dall's porpoise (*Phocoenoides dalli*) and Pacific white-sided dolphins (*Lagenorhyncus obliquidens*) often approach vessels from great distances to ride and play on bow waves. Sometimes even seemingly benign behavioural reactions can have serious consequences. In one study, Johnson (1977) estimated that occasional disturbances by low-flying aircraft may have been responsible for about 10% mortality in the two thousand pups born at one Alaskan haul-out site, because nursing pups were unable to follow their mothers into the water and became separated. In other areas, however, harbour seals seem to tolerate noise, with haul-out sites often situated in urban areas or near industrial sites, including one haul-out about 350 metres off the main runway at the Vancouver International Airport (Olesiuk 2009; unpublished data).

Hearing loss or masking may affect the ability of predators like killer whales and porpoise to detect prey using echolocation (Holt 2008), but it is also possible that prey may be less capable of detecting their predators. For example, killer whales feeding on fish tend to echolocate more than those feeding on marine mammals, probably because of the differing auditory abilities of fish and mammals to detect these sounds (Barrett-Lennard et al. 1996). Deecke et al. (2002) noted that seals were able to discern between the vocalizations of resident (fish-eating) and transient (marine mammal-eating) killer whales, and responded strongly only to the latter. In some cases, there can be synergistic effects between responses to noise and other behavioural effects. For instance, Bain (2002) noted that the combined cost of a reduced search field due to noise generated by boats, combined with increased swimming costs associated with avoiding the boats, could limit whale population when prey were scarce (see also Ford et al. 2009). Thus, our inability to delineate clear linkages between aquaculture noise and the health of aquatic animals and aquatic communities does not imply such linkages do not exist, but merely reflects our poor understanding of these complex processes.

The effect of sound on fish has received less attention than its effects on marine mammals. Much of the work on fish has not been published in the peer-reviewed literature, and varies in quality. On the other hand, the existing body of peer-reviewed, published work is too limited to make generalizations or extrapolate results (Popper and Hastings 2009). There are a wider variety of species of fish (29,000+ extant species), many go through distinct life stages (e.g., eggs, larvae, fry, smolts and adults), they have varied life histories, are difficult to observe in situ, and they exhibit a variety of structures or organ systems that could potentially be damaged by sound (e.g., ear structure, lateral line, swim bladders and other tissues). Our understanding of sound reception, utilization, mechanisms and potential adverse effects is even less advanced for invertebrates. While the available information does not appear to indicate that aquaculture noise has major impacts on fish or invertebrates in natural ecosystems, such conclusions need to be tempered given the sparsity of pertinent information.

Given the uncertainties associated with the potential long-term, chronic effects of far-ranging aquaculture noise, it is important that we continue to monitor the health of adjacent aquatic ecosystems. Marine mammals, particularly harbour porpoise and killer whales, and fish with specialized hearing such as herring, may serve as good indicator species. Population-level impacts of noise output associated with aquaculture operations occur when there are significant levels of injury or disturbance to important components or large fractions of marine animal populations or large areas of their habitat. This appears to have been the case when AHDs were in widespread use, and their far-ranging sounds displaced porpoises, dolphins and whales from large regions (Strong et al. 1995; Morton 2000; Morton and Symonds 2002; Olesiuk et al. 2002).

The emphasis of this review has been on finfish aquaculture and marine mammals, with little mention of shellfish or freshwater aquaculture. To some extent this reflects the bias in the authors' areas of specialization and local knowledge, but it probably also reflects the nature of the aquaculture noise issue. To our knowledge the sounds generated to deter pinnipeds at salmon farms are far more intense than many other sources of aquaculture noise. Marine mammals are more sensitive to sounds over a broader spectrum than other aquatic animals, their use of acoustics is advanced, so they probably serve as a good barometer of potential noise effects. Nevertheless, there is growing appreciation that anthropogenic noise can affect fish, especially species with specialized hearing (Popper et al. 2004), and evidence exists that some invertebrates like squid use statocysts for hearing low frequency sounds (Kaifu et al. 2008). Detailed data are largely lacking for sharks and turtles, but both can hear at lower frequencies. We are not aware of acoustic deterrents being used at shellfish sites[8] or freshwater systems, but the general issues of operational noise and its effects on freshwater fish and terrestrial mammals that dive (e.g., river otters) are probably similar to those outlined here.

---

[8] There are issues with pinnipeds hauling out on or adjacent to shellfish aquaculture facilities, and concern over them causing elevated faecal coliform counts (Calambokidis et al. 1989), so it is conceivable that acoustic deterrents have been used in some instances.

5. ACKNOWLEDGEMENTS

We very much appreciate the expert reviews of an earlier draft of this document provided by Dr. A.N. Popper and Dr. J.M. Terhune.

6. REFERENCES

Ainslie, M.A. and J.G. McColm. 1998. A simplified formula for viscous and chemical absorption in sea water. J. Acoustic. Soc. America 103(3): 1671-1672.

American Society of Heating, Refrigerating and Air-Conditioning Engineers (ASHRAE). 2002. Generator Noise Control – An Overview. (available at: http://www.enoisecontrol.com/).

Amundin, M. and B. Amundin. 1973. On the behaviour and study of the harbour porpoise (*Phocoena phocoena*) in the wild. pp. 317-328. In G. Pilleri [ed.]. Investigations on Cetacea. Bern: Hirnanatomisches Institute.

Andersen, S. 1970. Auditory sensitivity of the harbour porpoise (*Phocoena phocoena*). Invest. Cetacea 2: 255-259.

Anderson, S.S. and A.D. Hawkins. 1978. Scaring seals by sound. Mammal. Rev. 8: 19-24.

Andrew, R.K., B.M. Howe, and J.A. Mercer. 2002. Ocean ambient sound: comparing the 1960s with the 1990s for a receiver off the California coast. Acoustic Research Letters Online 3:65–70.

Au, W. 1993. The Sonar of Dolphins. Springer-Verlag, New York, 277 pp.

Au. W. W.L., J.K.B. Ford, J.K. Horn, and K.A. Newman Allman. 2004. Echolocation signal of free-ranging killer whales (*Orcinus orca*) and modeling of foraging for chinook salmon (*Onchorhynchus tshawyscha*). J. Acoust. Soc. Amer. 77:726-730.

Awbrey, F.T. and J.A. Thomas. 1987. Measurement of sound propagation from several acoustic harassment devices. pp. 85-104. In B.R. Mate and J.T. Harvey [eds.]. Acoustical Deterrents in Marine Mammal Conflicts with Fisheries. A Workshop held 17-18 February, 1986, Newport, OR. Oregon State Univ. Publ. No. ORESU-W-86-001.

Awbrey, F.T., J.A. Thomas, and R.A. Kastelein. 1988. Low frequency underwater hearing sensitivity in belugas, *Delphinapterus leucas*. J. Acoust. Soc. Amer. 84: 2273-2275.

Bain, D.E. 2002. A model linking energetic effects of whale watching on killer whale (*Orcinus orca*) population dynamics. Unpubl. Rep. sponsored by Orca Relief Citizens Reliance. 23 pp.

Barnes, L.G., D.P. Domning, and C.E. Ray. 1985. Status of studies on fossil marine mammals. Mar. Mamm. Sci. 1: 15-53.

Barnes, L.G. 1999. Porpoises. pp. 74-75. In M. Carwardine [ed.]. Whales, Dolphins and Porpoises. Checkmark Books, New York.

Barrett-Lennard, L.G., J.K.B. Ford, and K. Heise. 1996. The mixed blessing of echolocation: Differences in sonar use by fish-eating and mammal-eating killer whales. Anim. Behav. 51: 553-565.

Bart, A.N., J. Clark, J, Young, and Y. Zohar. 2001. Underwater ambient noise measurements in aquaculture systems: a survey. Aquacult. Eng. 25: 99-110.

Bartol, S.M., J.A. Musick, and M. Lenhardt. 1999. Auditory evoked potentials of the loggerhead sea turtle (*Caretta caretta*). Copeia 99: 836-840.

Bartol, S.M. and D.R. Ketten. 2006. Turtle and tuna hearing. pp. 98–105. In Sea Turtle and Pelagic Fish Sensory Biology: Developing techniques to reduce sea turtle bycatch in longline fisheries. [ed.]. Y. Swimmer, and R. Brill. NOAA.

Bejder, L., A. Samuels, H. Whithead, H. Finn, and S. Allen. 2009. Impact assessment research: use and misuse of habituation, sensitization and tolerance in describing wildlife responses to anthropogenic stimuli. Mar. Ecol. Prog. Ser. 00: 1-9.

Berrow, S., R. Cosgrove, R.H. Leeney, J. O'Brien, D. McGrath, J. Dalgard, and Y. Le Gall. 2008. Effects of acoustic deterrents on the behavior of common dolphins (*Delphinus delphis*). Cetacean Res. Manage. 10: 227–233.

Blaxter, J.H.S., J.A.B. Gray, and E.J. Denton. 1981. Sound and startle response in herring shoals. J. Mar. Biol. Assoc. UK 61: 851-869.

Brown, T.G., B. Munro, C. Beggs, E. Lochbaum, and P. Winchell. 2003. Courtenay River Seal Fence. Can. Tech. Rep. Fish. Aquat. Sci. 2459.

Budelmann, B.U. 1988. Morphological diversity of equilibrium receptor systems in aquatic invertebrates. pp. 757-782 *In:* J. Atema et al. [eds.]. Sensory Biology of Aquatic Animals, Springer-Verlag, New York.

Budelmann, B.U. 1992. Hearing in crustacea. pp. 131-139. In Webster, D.B., et al. [eds.]. Evolutionary Biology of Hearing. Springer-Verlag, New York.

Calambokidis, J., B. McLaughlin, and G. Steiger. 1989. Bacterial contamination related to harbor seals in Puget Sound, Washington. Report to Jefferson County and the Washington Department of Ecology in cooperation with the Washington Department of Social and Health Services. 74 pp.

Casper, B.M. and D.A. Mann. 2006. Evoked potential audiograms of the nurse shark (*Ginglymostoma cirratum*) and the yellow stingray (*Urobatis jamaicensis*). Environ. Biol. Fishes 76:101-108.

Casper, B.M. and D.A. Mann. 2007a. Dipole hearing measurements in elasmobranch fishes. J. Exp. Biol. 210:75-81.

Casper, B.M. and D.A. Mann. 2007b. The directional hearing abilities of two species of bamboo sharks. The Journal of Experimental Biology 210: 505-511.

Chapman, C.J. 1973. Field studies of hearing in teleost fish. Helgoländer wiss. Meeresunters. 24: 371-390.

Chapman, C.J. and A.D. Hawkins. 1973. A field study of hearing in the cod (*Gadus morhua* L.). J. Comp. Physiol. A 85:147-167.

Chapman, C.J. and O. Sand. 1974. Field studies of hearing in two species of flatfish *Pleuronectes platessa* and *Limanda limanda*. Comp. Biol. Physiol. 47A: 371-385.

Christian, J.R., A. Mathieu, D.H. Thomson, D. White, and R.A. Buchanan. 2003. Effects of Seismic Energy on snow Crab (*Chionoecetes opilio*). Report from LGL Ltd. Oceans Ltd. for the National Energy Board, File No.: CAL-1-00364, 11 April 2003. 91 pp.

Clark, C.W. 1994. Blue deep voices: insights from the Navy's Whales '93 program. Whalewatcher 28:6-11.

Clay, A.S. and H. Medwin. 1977. An acoustical view of the sea . pp.1-29 In: McCormack, M.E. and R. Bhattachraryya [eds.]. Acoustical oceanography principles and applications. John Wiley and Sons, New York.

Costa, D.P., D.E. Crocker, J. Gedamke, J.P.M., Webb, D.S. Houser, S.B. Blackwell, D. Waples. S.E. Hayes, and B.J. Le Boeuf. 2003. The effect of a low-frequency sound source (Acoustic Thermometry of Ocean Climate) on the diving behavior of juvenile northern elephant seals, *Mirounga angustirostris*. J. Acoust. Soc. America 113:1155-1165.

Cox, T.M., T.J. Ragen, A.J. Read, E. Vos, R.W. Baird, K. Balcomb, J. Barlow, J. Caldwell, T. Cranford, L. Crum, A. D'Amico, G. D'Spain, A. Fernández, J.R. Finneran, R. Gentry, W. Gerth, F. Gulland, J. Hildebrand, D. Houser, T. Hullar, P.D. Jepson, D. Ketten, C.D. MacLeod, P. Miller, S. Moore, D.C. Mountain, D. Palka, P. Ponganis, S. Rommel, T. Rowles, B. Taylor, P. Tyack, D. Wartzok, R. Gisiner, J. Mead, and L. Benner. 2006. Understanding the impacts of anthropogenic sound on beaked whales. J. Cetacean Res. Manage. 7: 177-187.

Cummings, W.C. and P.O. Thompson. 1971. Gray whales, *Eschrichtius robustus*, avoid the underwater sounds of killer whale, *Orcinus orca*. Fish. Bull. 69: 525-530.

Deecke, V.B., P.J.B. Slater, and J.K.B. Ford. 2002. Selective habituation shapes acoustic predator recognition in harbour seals. Nature 420: 171–173.

DFO 2006a. Canadian Aquaculture Industry Export and Production Figures. Aquaculture Fact Sheets http://www.dfo-mpo.gc.ca/aquaculture/sheet_feuillet/figures-eng.htm. (Accessed 10-Sept-2009).

DFO 2006b. Science and Implementation Considerations of Mitigation Techniques to Reduce Small Cetacean Bycatch in Fisheries. DFO Can. Sci. Advis. Sec. Proceed. Ser. 2006/020.

DFO 2009. Management Plan for the Steller sea lions (*Eumetopias jubatus*) in Canada. Species at Risk Act Management Plan Series. Fisheries and Oceans Canada, Ottawa. vii + 71 pp.

DFO. 2010a. Pathways of Effects for Finfish and Shellfish Aquaculture. DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 2009/071.

DFO. 2010b. Socio-economic Impact of Aquaculture in Canada. DFO/2010-1645 Cat. No. Fs23-551/2010E.

Doksæter, L., O.R. Godø, N.O. Handegard, P.H. Kvadsheim, F.-P. A. Lam, C. Donovan, and P.J.O. Miller. 2009. Behavioral responses of herring to 1–2 and 6–7 kHz sonar signals and killer whale feeding sounds. J. Acoustic. Soc. Amer. 125: 554-564.

Domning, D.P., G.S. Morgan, and C.E. Ray. 1982. North American Eocene sea cows (Mammalia: Sirenia). Smithsonian Contrib. Paleobiology 52: 1-69.

Dunn, D. E., R.R. Davis, C.J. Mercy, and J.R. Franks. 1991. Hearing loss in the chinchilla from impact and continuous noise exposure. J. Acoustic. Soc. Amer. 90: 1979-1985.

Dzwilewski, P.T. and G. Fenton. 2003. Shock Wave/Sound Propagation Modeling Results for Calculating Marine Protected Species Impact Zones During Explosive Removal of Offshore Structures. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study MMS 2003-059. 39 pp.

Engås, A. and S. Løkkeborg. 2002. Effects of seismic shooting and vessel-generated noise on fish behaviour and catch rates. Bioacoust. 12: 313–315.

Engås, A., S. Løkkeborg, E. Ona, and A.V. Soldal. 1996. Effects of seismic shooting on local abundance and catch rates of cod (*Gadus morhua*) and haddock (*Melanogrammus aeglefinus*). Can. J. Fish. Aquat. Sci. 53: 2238-2249.

Enger, P.S. 1981. Frequency discrimination in teleosts—Central or peripheral? pp. 243-255 In W.A. Tavolga, A.N. Popper, and R.R. Fay [eds.]. Hearing and Sound Communication in Fishes. Springer, New York.

Enger, P.S. 1967. Hearing in herring. Comp. Biochem. Physiol. 22: 527–538.

Erbe, C. 2002. Underwater noise of whale-watching boats and potential effects on killer whales (*Orcinus orca*), based on an acoustic impact model. Mar. Mam. Sci. 18: 394-418.

Erbe, C. and D.M. Farmer. 2000. A software model to estimate zones of impact on marine mammals around anthropogenic noise. J. Acoust. Soc. of America 108: 1327-1331.

Fay, R.R. and A.N. Popper. 2000. Evolution of Hearing in Vertebrates: The inner ears and processing. Hearing Res. 149: 1-10.

Fernández, A., M. Arbelo, R. Deaville, I.A.P. Patterson, P. Castro, J.R. Baker, E. Degollada, H.M. Ross, P. Herraez, A.M. Pocknell, E. Rodriguez, F.E. Howie, A. Espinosa, R.J. Reid, J.R. Jaber, V. Martin, A.A. Cunningham, and P.D. Jepson. 2004. Pathology: Whales, sonar and decompression sickness (reply). [Brief Communications]. Nature, 428: 6984, U1-2.

Fernández, A., J.F. Edwards, F. Rodríguez, A. Espinosa de los Monteros, P. Herráez, P. Castro, J.R. Baker, E. Degollada, H.M. Ross, P. Herraez, A.M. Pocknell, F. Rodriguez, F.E. Howie, A. Espinosa, R.J. Reid, J.R. Jaber, V. Martin, and P.D. Jepsen. 2005. Gas and fat embolic syndrome involving a mass stranding of beaked whales (Family Ziphiidae) exposed to anthropogenic sonar signals. Veterinary Pathology 42: 446-457.

Finneran, J.J., D.A. Carder, C.E. Schlundt, and S.H. Ridgway. 2005. Temporary threshold shift (TTS) in bottlenose dolphins (*Tursiops truncatus*) exposed to mid- frequency tones. J. Acoustic. Soc. Amer. 118: 2696-2705.

Finneran, J.J., C.E. Schlundt, B. Branstetter, and R.L. Dear. 2007. Assessing temporary threshold shift in a bottlenose dolphin (*Tursiops truncatus*) using multiple simultaneous auditory evoked potentials. J. Acoust. Soc. America 122: 1249-1264.

Finneran, J.J., C.E. Schlundt, R. Dear, D.A. Carder, and S.H. Ridgway. 2002. Temporary shift in masked hearing thresholds (MTTS) in odontocetes after exposure to single underwater impulses from a seismic watergun. J. Acoustic. Soc. Amer. 111: 2929-2940.

Fish, J.F. and J.S. Vania. 1971. Killer whale, *Orcinus orca*, sounds repel white whales, *Delphinapterus leucas.* Fish. Bull. 69: 531-536.

Fjalling, A., M. Wahlberg, and H. Westerberg. 2006. Acoustic harassment devices reduce seal interaction in the Baltic salmon-trap, net fishery. ICES J. Marine Sci. 63: 1751-1758.

Foote, A.D., R.W. Osborne, and A.R. Hoelzel. 2004. Whale-call response to masking boat noise. Nature 428: 910.

Ford, J.K.B. and G.M. Ellis. 1999. Transients: mammal-hunting killer whales of British Columbia, Washington, and Southeastern Alaska. UBC Press, Vancouver, BC.

Ford, J. K. B., G.M. Ellis, L. Barrett-Lennard, A.B. Morton, R. Palm, and K.C. Balcomb III. 1998. Diet specialization in two sympatric populations of killer whales (*Orcinus orca*) in coastal British Columbia and adjacent waters. Can. J. Zool. 76: 1456–1471.

Ford, J.K.B., G.M. Ellis, P.F. Olesiuk, and K.C. Balcomb. 2009. Linking killer whale survival and prey abundance: food limitation in the oceans' apex predator? Biol. Letters 00: 1-4.

Gagnon, G.J. and C.W. Clark. 1993. The use of the US Navy IUSS passive sonar to monitor the movements of Blue whales. In Abstracts, Tenth Biennial Conference on the Biology of Marine Mammals, Galveston, Texas.

Goertner, J.F. 1982. Prediction of underwater explosion safe ranges for sea mammals. NSWC/ OL TR-82-188. Naval Surface Weapons Center, White Oak Lab., Silver Spring, MD. 25 pp. NTIS AD-A139823.

Gordon, J. and S. Northridge. 2002. Potential impacts of acoustic deterrent devices on Scottish marine wildlife. Scottish Natural Heritage Commissioned Report F01AA404.

Gordon, J. and P.L. Tyack. 2001. Cetaceans and Sound. pp. 139-194 In P.G.H. Evans and J.A. Raga [eds.]. Marine Mammals: Biology and Conservation. New York, Kluwer Academic / Plenum Publishers.

Graham, I.M., R.N. Harris, B. Denny, D. Fowden, and D. Fullan. 2009. Testing the effectiveness of an acoustic deterrent device for excluding seals from Altantic salmon rivers in Scotland. ICES J. Mar. Sci. 66.

Greene, C.R. 1985. Characteristics of waterborne industrial noise, 1980-84. pp. 197-253 In W.J. Richardson, [ed.]. Behavior, Disturbance Responses and Distribution of Bowhead Whales, *Balaena mysticetus,* in the eastern Beaufort Sea, 1980-84. OCS Study MMS 85-0034. 306 pp.

Haarr, M.L., L.D. Charlton, J.M. Terhune, and E.A. Trippel. 2009. Harbour porpoise (*Phocoena phocoena*) presence patterns at an aquaculture cage site in the Bay of Fundy, Canada. Aquatic Mamm. 35: 203-211.

Hall, J.D. and C.S. Johnson. 1972. Auditory thresholds of a killer whale, *Orcinus orca, Linnaeus*. J. Acoustic. Soc. Amer. 51: 515-517.

Haller, D.R. and D.D. Lemon. 1994. Acoustic source levels of AirMar and PRA seal-scarers. Dept. Fisheries and Oceans Contr. Rept. No. XSA FP941-3-8025/00/A. 52 pp.

Halmagyi, G.M., I.S. Curthoys, J.G. Colebatch and S.T. Aw. 2005. Vestibular responses to sound. Clinical and basic oculomotor research: In honor of David S. Zee. Annals of the New York Academy of Sciences. 1039: 54-67.

Hastings, M.C. and A.N. Popper. 2005. Effects of Sound on Fish. Unpublished report prepared for California Department of Transportation. Available at: http://www4.trb.org/trb/crp.nsf/reference/boilerplate/Attachments/$file/EffectsOfSoundOnFish1-28-05(FINAL).pdf

Hastings, M.C., A.N. Popper, J.J. Finneran, and P.J. Lanford. 1996. Effects of low-frequency underwater sound on hair cells of the inner ear and lateral line of the teleost fish, *Astronotus ocellatus*. J. Acoust. Soc. Am. 99: 1759–1762.

Hawkins, A.D. and A.D.F. Johnstone. 1978. The hearing of the Atlantic salmon (*Salmo salar*). J. Fish Biol. 13: 655-673.

Hueter, R.E., D.A. Mann, K.P. Maruska, J.A. Sisneros, and L.S. Demski. 2004. Sensory Biology of Elasmobranchs. In J.C. Carrier, J.A. Musick and M.R. Heithaus [eds.]. Biology of Sharks and their Relatives. CRC Press, USA.

Hirsch, A.E. and A.K. Ommaya. 1972. Head injury caused by underwater explosion of a firecracker. J. Neurosurg 37: 95-99.

Holt, M.M. 2008. Sound exposure and southern resident killer whales (*Orcinus orca*): A review of current knowledge and data gaps. NOAA Technical Memorandum NMFS-NWFSC-89. 59 pp.

Jacobs, D.W. and W.N. Tavolga. 1967. Acoustic intensity limens in the goldfish. Anim. Behav. 15: 324-335.

Jacobs, S.R. and J.R. Terhune. 2002. The effectiveness of acoustic harassment devices in the Bay of Fundy, Canada: seal reactions and a noise exposure model. Aquat. Mamm. 28: 147-158.

Jamieson, G.S. and P.F. Olesiuk. 2001. Salmon farm – pinniped interactions in British Columbia: An analysis of predator control, its justification, and alternate approaches. Can. Sci. Advis. Rep. Res. Doc. 2001/142.

Jepson, P.D., M. Arbelo, R. Deaville, I.A.P. Patterson, P. Castro, J.R. Baker, E. Degollada, H.M. Ross, P. Herraez, A.M. Packnell, F. Rodriguez, F.E. Howell, A. Espinosa, R.J. Reid, J.R. Jaber, V. Martin, A.A. Cunningham, and A. Fernandez. 2003. Gas-bubble lesions in stranded cetaceans. Nature 425: 575-576.

Johnson, B. 1977. The effects of human disturbance on a population of harbour seals. pp.422-431. In: Environ. Assess. Alaskan Cont. Shelf. Annu. Rep. Principle Invest. NOAA Boulder, CO 708 pp.

Johnson, C.S. 1968. Relation between absolute threshold and duration of tone pulses in the bottlenosed porpoise. J. Acoust. Soc. Amer. 43: 757-763.

Johnson, C.S. 1991. Hearing thresholds for periodic 60-kHz tone pulses in the beluga whale. J. Acoust. Soc. Am. 89: 2996-3001.

Johnson, C. S., M. W. McManus, and D.Skaar. 1989. Masked tonal hearing thresholds in the beluga whale. J. Acoust. Soc. Am. 85: 2651–2654.

Johnston, D.W. 2002. The effect of acoustic harassment devices on harbor porpoises (Phocoena *phocoena*) in the Bay of Fundy. Canada. Biol. Conserv. 108: 113-118.

Johnston, D.W. and T. H. Woodley. 1998. A survey of acoustic harassment device (AHD) use in the Bay of Fundy, NB, Canada. Aquat. Mamm. 24: 51-61.

Kaifu, K., T. Akamatsu, and S. Segawa. 2008. Underwater sound detection by cephalopod statocyst. Fisheries Sci. 74: 781–786.

Kastak, D., C. Reichmuth, M.M. Holt, J. Mulsow, B.L. Southall, and R.J. Schusterman. 2007. Onset, growth, and recovery of in-air temporary threshold shift in a California sea lion (*Zalophus californianus*). J. Acoustic. Soc. Amer. 122: 2916-2924.

Kastak, D. and R.J. Schusterman. 1998. Low-frequency amphibious hearing in pinnipeds: Methods, measurements, noise, and ecology. J. Acoustic. Soc. Amer. 103: 2216-2228.

Kastelein, R. A., Bunskoek, P., Hagedoorn, M., Au, W. W. L., and de Haan, D. 2002. Audiogram of a harbor porpoise (*Phocoena phocoena*) measured with narrow-band frequency-modulated signals. J. Acoust. Soc. Am. 112: 334–344.

Kastelein, R.A., M. Hagedoorn, W.W. Au, and D. de Haan. 2003. Audiogram of a striped dolphin (*Stenella coeruleoalba*). J. Acoustic. Soc. Amer. 113: 1130-1137.

Kastelein, R.A., W.C. Verboom, M. Muijsers, N.V. Jennings, and S. van der Heul. 2005. The influence of acoustic emissions for underwater data transmission on the behaviour of harbour porpoises

(*Phocoena phocoena*) in a floating pen. Mar. Environ. Res. 59: 287–307.

Kastelein, R.A. and P.J. Wensveen. 2008. Effect of two levels of masking noise on the hearing threshold of a harbor porpoise (*Phocoena phocoena*) for a 4.0 kHz signal. Aquat. Mamm. 34: 420-425.

Kenyon, T.N., F. Ladich, and H.Y. Yan. 1998. A comparative study of hearing ability in fishes: the auditory brainstem response approach. Journal of Comparative Physiology A. 182: 307-318.

Ketten, D.R. 1998. Marine mammal auditory systems: a summary of audiometric and anatomical data and its implications for underwater acoustic impacts. NOAA Technical Memorandum NMFS NOAA-TM-NMFS-SWFSC-256, National Marine Fisheries Service, La Jolla, CA. 74 pp.

Ketten, D.R. 2000. Cetacean Ears: Hearing by Whales and Dolphins. In W.W.L. Au, A.N. Popper and R.R. Fay. [eds.]. Springer-Verlag, New York, NY. pp. 43-108.

Kinsler, C.E., A.R. Frey, A.B. Coopens, and J.V. Sanders. 1982. Fundamental of acoustics, 3rd edition. John Wiley and Sons, New York.

Kritzler, H. and L. Wood. 1961. Provisional audiogram for the shark, *Carcharhinus leuces*. Science 133: 1480-1482.

Knudsen, F.R., P.S. Enger, and O. Sand. 1992. Awareness reactions and avoidance responses to sound in juvenile Atlantic salmon, *Salmo salar L*. J. Fish. Biol. 40: 523-534.

Knudsen. F.R., P.S. Enger, and O. Sand. 1994. Avoidance responses to low frequency sound in downstream migrating Atlantic salmon smolt, *Salmo salar L*. J. Fish Biol. 45: 227- 233.

Knudsen, F.R., C.B. Schreck, S.M. Knapp, P.S. Enger, and O. Sand. 1997. Infrasound produces flight and avoidance responses in Pacific juvenile salmonids. J. Fish. Biol. 51: 824-829.

Kosheleva, V. 1992. The impact of air guns used in marine seismic explorations on organisms living in the Barents Sea. Contr. Petro Piscis II `92 Conference F-5, Bergen, 6-8 April, 1992.

Kritzler, H. and L. Wood. 1961. Provisional audiogram for the shark *Carcharhinus leuces*. Science 133: 1480-1482.

Kruse, S. 1991. The interactions between killer whales and boats in Johnstone Strait, BC pp. 48-159 In Pryor, K. and K.S. Norris [eds.]. Dolphin societies: Discoveries and puzzles. University of California Press, Berkeley, CA.

Kryter, K. 1994. The handbook of hearing and the effects of noise. New York: Academic Press.

Laake, J., D. Rugh, and L. Baraff. 1998. Observations of harbour porpoise in the vicinity of acoustic alarms on a set gillnet. U.S. Dept. Commerc., NOAA Tech. Memo. NMFS-AFSC-84 40 pp.

Lawson, J.W. 2009. The use of sound propagation models to determine safe distances from a seismic sound energy source. DFO Can. Sci. Advis. Sec. Res. Doc., 2009/060.

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

Lien, J. 2001. The Conservation Basis for the Regulation of Whale Watching in Canada by the Department of Fisheries and Oceans: A Precautionary Approach. Can. Tech. Rep. Fish. Aquat. Sci. 2363: vi + 38 pp.

Madsen, P.T. 2005. Marine mammals and noise: Problems with root-mean-square sound pressure level for transients. J. Acoustic. Soc. Amer. 117: 3952-3957.

Madsen, P.T., M. Wahlberg, J. Tougaard, K. Lucke, and P.L. Tyack. 2006. Wind turbine underwater noise and marine mammals: Implications of current knowledge and data needs. Mar. Ecol. Prog. Ser. 309: 279–295.

Malme, C.I., P.R. Miles, G.W. Miller, W.J. Richardson, D.G. Roseneau, D.H. Thomson, and C.R. Greene, Jr. 1989. Analysis and ranking of the acoustic disturbance potential of petroleum industry activities and other sources of noise in the environment of marine mammals in Alaska. BBN Rep. 6945; OCS Study MMS 89-0006.

Mann, D.A., D.M. Higgs, W.N. Tavolga, M.J. Souza, and A.N. Popper. 2001. Ultrasound detection by clupeiform fishes. J. Acoust. Soc. Am. 109: 3048–3054.

Mann, D.A., Z. Lu, and A.N. Popper. 1997. A clupeiform fish can detect ultrasound. Nature 389: 341.

Mann, D.A., Z. Lu, M.C. Hastings, and A.N. Popper. 1998. Detection of ultrasonic tones and simulated dolphin echolocation clicks by a teleost fish, American shad (*Alosa sapidissima*). J. Acoust. Soc. Amer. 104: 562-568.

Mate, B.R., R.F. Brown, C.F. Greenlaw, J.T. Harvey, and J. Tempte. 1987. An acoustic harassment technique to reduce seal predation on salmon. pp. 23-36 In B.R. Mate and J.T. Harvey [eds.]. Acoustical Deterrents in Marine Mammal Conflicts with Fisheries. A Workshop held 17-18 February, 1986, Newport, OR. Oregon State Univ. Publ. No. ORESU-W-86-001. 116 pp.

Mate, B.R. and J.T. Harvey. 1987. Acoustical deterrents in marine mammal conflicts with fisheries. A Workshop held 17-18 February, 1986, Newport, OR. Oregon State Univ. Publ. No. ORESU-W-86-001. 116 pp.

McCauley, R.D., J. Fewtrell, A.J. Duncan, C. Jenner, M.-N. Jenner, J.D. Penrose, R.I.T. Prince, A. Adihyta, J. Murdoch, and K. McCabe. 2000. Marine seismic surveys: A study of environmental implications. Austr. Petroleum Prod. Exploration Assoc. J. 40: 692-708.

McCauley, R.D., J. Fewtrell, and A.N. Popper. 2003. High intensity anthropogenic sound damage fish ears. J. Acoust. Soc. America 113: 638– 642.

McCauley, R.D., M.-N. Jenner, C. Jenner, K.A. McCabe, and J. Murdoch. 1998. The response of humpback whales (*Megaptera novaeangliae*) to offshore seismic survey noise: Preliminary results of observations about a working seismic vessel and experimental exposures. Austr. Petroleum Prod. Exploration Assoc. J. 38: 692-707.

McDonald, M.A., J.A. Hildebrand, and S.M. Wiggins. 2006. Increases in deep ocean ambient noise in the Northeast Pacific west of San Nicolas Island, California. J. Acoustic. Soc. Amer. 120: 711–718.

Miles, P.R., C.I. Malme, and W.J. Richardson. 1987. Prediction of Drilling Site-Specific Interaction of Industrial Acoustic Stimuli and Endangered Whales in the Alaskan Beaufort Sea. BBN Rep. 6509; OCS Study MMS 87-0084. 341 pp.

Miller, P.J.O., N. Biassoni, A. Samuels, and P.L. Tyack. 2000. Whale songs lengthen in response to sonar. Nature 405: 903.

Møhl, B. 1968. Auditory sensitivity of the common seal in air and water. J. Auditory Res. 8: 27-38.

Møhl, B., M. Wahlberg, P.T. Madsen, L.A. Miller, and A. Surlykke. 2000. Sperm whale clicks: Directionality and source level revisited. J. Acoust. Soc. Am. 107: 638–648.

Moore, M.M. and G.A. Early. 2004. Cumulative sperm whale bone damage and the bends. Science 306: 2215.

Moore, P.W.B. and R.J. Schusterman. 1987. Audiometric assessment of northern fur seals, *Callorhinus ursinus.* Marine Mammal Science 3: 31–53.

Morton, A.B. 1990. A quantitative comparison of the behaviour in resident and transient killer whales of the central British Columbia coast. Rep. Int. Whal. Commn. (Special Issue 12): 245-248.

Morton, A.B. 2000. Occurrence, photo-identification and prey of Pacific white-sided dolphins (*Lagenorhyncus obliquidens*) in the Broughton Archipelago, Canada 1985–1997. Marine Mammal Science 16: 80–93.

Morton, A.B. and H.K. Symonds. 2002. Displacement of *Orcinus orca* (*L.*) by high amplitude sound in British Columbia. ICES J. Mar. Sci. 59:71-80.

Myberg, A.A. 1990. The effects of noise on the behavior of marine animals. Env. International. 16: 575-586.

Myrick, A.C., Jr., E.R. Cassano, and C.W. Oliver. 1990a. Potential for physical injury, other than hearing damage, to dolphins from seal bombs used in the Yellowfin tuna purse-seine fishery: Results from open-water tests. Admin. Rep LJ-90-07. U.S. Natl. Mar. Fish. Serv., La Jolla, CA. 28 pp.

Myrick, A.C., Jr., M. Fink, and C.B. Glick. 1990b. Identification, chemistry and behavior of seal bombs used to control dolphins in the yellowfin tuna purse-seine fishery in the Eastern Tropical Pacific: Potential hazards. Admin. Rep LJ-90-08. U.S. Natl. Mar. Fish. Serv., La Jolla, CA. 25 pp.

National Institute for Occupational Safety and Health (NIOSH). 1998.Criteria for a recommended standard: Occupational noise exposure (Publication #98-126). U.S. Department of Health and Human Services, NIOSH, Washington, DC.

National Institute Deafness and other Communicative Disorders (NIDCD). 2009. How loud is too loud? (web site available at: http://www.nidcd.nih.gov/health/hearing/ruler.asp).

National Research Council (NRC). 1994. Low-Frequency Sound and Marine Mammals: Current Knowledge and Research Needs. National Academy Press, Washington, DC, 75 pp.

National Research Council (NRC). 2000. Marine Mammals and Low-frequency Sound. Washington, DC: The National Academies Press. 146 pp.

National Research Council (NRC). 2003. Ocean Noise and Marine Mammals. Washington, DC: The National Academies Press. 192 pp.

National Research Council (NRC). 2005. Marine Mammal Populations and Ocean Noise: Determining when noise causes biologically significant effects. Washington, DC: The National Academies Press. 126 pp.

Nedwell, J.R., A.W.H. Turnpenny, J. Lovell, S.J. Parvin, R. Workman, J.A.L. Spinks, and D. Howell. 2007. A validation of the dBht as a measure of the behavioural and auditory effects of underwater noise Subacoustech Ltd Report No. 534R1231 to Chevron Ltd, Total Fina Elf Exploration UK PLC, Shell UK Exploration and Production Ltd, the Department of Business, Enterprise and Regulatory Reform, the Joint Nature Conservation Committee and The UK Ministry of Defence. Published by the Department of Business, Enterprise and Regulatory Reform, 2007. 74 pp.

Norris, K.S. 1969. The echolocation of marine mammals. pp. 391-423 In The Biology of Marine

Mammals. Academic Press, New York. 511pp.

Nowacek, D. P., L. H. Thorne, D. W. Johnston, and P. L. Tyack. 2007. Responses of cetaceans to anthropogenic noise. Mammal. Rev. 37: 81-115.

Offutt, G.C. 1970. Acoustic stimulus perception by the American lobster, *Homarus americanus* (Decapoda). Experientia 26: 1276-1278.

O'Keeffe, D.J. and G.A. Young. 1984. Handbook on the Environmental Effects of Underwater Explosions. NSSWC/WOL TR-83-240. Naval Surface Weapons Center, White Oak Lab., Silver Springs, MD. 207 pp. Def. Tech. Info. Cent. AD-B 093885.

Olesiuk, P.F., Nichol, L.M., M.J. Sowden, and J.K.B. Ford. 1995. Effect of sounds generated by acoustic deterrent device on the abundance and distribution of harbor porpoise (*Phocoena phocoena*) in Retreat Passage, British Columbia. Avail- able from Fisheries and Oceans Canada, Pacific Biological Station, Nanaimo BC V9R 5K6 Canada. 47 pp.

Olesiuk, P.F., L.M. Nichol, M.J. Sowden, and J.K.B. Ford. 2002. Effect of the sound generated by an acoustic harassment device on the relative abundance and distribution of harbor porpoises (*Phocoena phocoena*) in Retreat Passage, British Columbia. Mar. Mamm. Sci. 18: 843-862.

Olsen, K., J. Angell, F. Pettersen, and A. Løvik. 1983. Observed fish reactions to a surveying vessel with special reference to herring, cod, capelin and polar cod. FAO Fish. Rep. 300: 131–138.

Oosterveld, W., A. Polman, and J. Schoonheyt. 1982. Vestibular implications of noise-induced hearing loss. British J. Audition 16: 227-232.

Payne, R. and D. Webb. 1971. Orientation by means of long range acoustic signalling in baleen whales. Annals of the New York Acad. Sci. 188: 110-141.

Pincock, D.G. and F.A.Voegeli. 2002. Quick Course in Underwater Telemetry Systems. Vemco Ltd. Publication. 43 pp.

Polacheck, T. and L. Thorpe. 1990. The swimming direction of harbour porpoises in relationship to a survey vessel. Rep. Intern. Whaling Comm. 40: 463–470.

Popper, A.N. 1980. Sound emission and detection by delphinids. pp. 1-52 In: L. Herman, [ed.]. Cetacean behavior: Mechanisms and functions. Wiley Inter-Science, New York. 463 pp.

Popper, A.N. 2003. Effects of anthropogenic sound on fishes. Fisheries 28, 24–31.

Popper, A.N. and R.R. Fay. 1999. The auditory periphery in fishes. pp. 43-100 In R.R. Fay and A.N. Popper [eds.]. Comparative Hearing: Fish and Amphibians. Springer-Verlag, New York.

Popper, A. N., R.R. Fay, C. Platt, and O. Sand. 2003. Sound detection mechanisms and capabilities of teleost fishes. pp. 3-38 In: S.P. Collin and N.J. Marshall [eds.]. Sensory Processing in Aquatic Environments. New York: Springer-Verlag.

Popper, A.N., J. Fewtrell, M.E. Smith, and R.D. McCauley. 2004. Anthropogenic sound: Effects on the behavior and physiology of fishes. Mar. Technol. Soc. J. 37: 35–40.

Popper, A.N., M.B. Halvorsen, A. Kane, D.L. Miller, M.E. Smith, J. Song, P. Stein, and L.E. Wyscocki. 2007. The effects of high-intensity, low-frequency active sonar on rainbow trout. J. Acoust. Soc. Am. 122: 623–635.

Popper, A.N. and M.C. Hastings. 2009. The effects of anthropogenic sources of sound on fishes. J. Fish.

Biol. 75: 455-489.

Popper, A.N., M. Salmon, and K.W. Horch. 2001. Acoustic detection and communication by decapod crustaceans. J. Comp. Physiol. A 187:83-89.

Popper, A. N., M.E. Smith, P.A. Cott, B.W. Hanna, A.O. MacGillivray, M..E. Austin, and D.A. Mann. 2005. Effects of exposure to seismic airgun use on hearing of three fish species. J. Acoust. Soc. Am. 117: 3958-3971.

Pryor, K. 1987. Behavioral conditioning. In Mate, B.R. and J.T. Harvey [eds.]. Acoustical Deterrents in Marine Mammal Conflicts with Fisheries. pp. 105-110 A Workshop held 17-18 February, 1986, Newport, OR. Oregon State Univ. Publ. No. ORESU-W-86-001.

Richardson, W. J., C.R. Greene Jr., C.I. Malme, and D.H. Thomson. 1995. Marine Mammals and Noise. New York: Academic Press. 576 pp.

Ridgway, S.H. and P.L. Joyce. 1975. Studies on seal brain by radiotelemetry. Rapports et Procès-Verbeaux des Réunions, Conseil International pour l'Exploration de la Mer 169 :1 81-91.

Ridgway, S.H., E.G., Wever, J.G. McCormick, J. Palin, and J.H. Anderson. 1969. Hearing in giant sea turtle (*Chelonia myads*). Proceedings of the National Academy of Sciences 64: 884-890.

Rogers, P. H. and M. Cox. 1988. Underwater sound as a biological stimulus. pp. 131–149 In J. Atema, R.R. Fay, A.N. Popper and W.N. Tavolga [eds.]. Sensory Biology of Aquatic Animals. Springer-Verlag, New York, NY.

Rueggeberg, H. and J. Booth. 1989. Interactions between wildlife and salmon farms in British Columbia: results of a survey. Technical Report Series No. 67, Canadian Wildlife Service, Pacific and Yukon Region, BC. 74 pp.

Sand, O., P.S. Enger, H.E. Karlsen, F.R. Knudsen, and T. Kvernstuen. 2000. Avoidance responses to infrasound in downstream migrating European silver eels, *Anguilla anguilla*. Environ. Biol. Fish. 57: 327-336.

Sand, O. and Karlsen, H.E. 1986. Detection of infrasound by the Atlantic cod. J. Exp. Biol. 125: 197-204.

Sand, O. and Karlsen, H.E. 2000. Detection of infrasound and linear acceleration in fish. Phil. Trans. R. Soc. Lond. B 355: 1295-1298.

Schlundt, C.E.,R.L. Dear, L. Green, D.S. Houser, and J.J. Finneran. 2007. Simultaneously measured behavioral and electrophysiological hearing thresholds in a bottlenose dolphin (*Tursiops truncatus*). J. Acoust. Soc. America 122: 615-622.

Sepulveda, M. and D. Oliva. 2005. Interactions between South American sea lions (*Otaria flavescens* Shaw) and salmon farms in southern Chile. Aquaculture Research 36: 1062–1068.

Shapiro, A.D., J. Tougaard, P.B. Jorgensen, L.A. Kyhn, J.D. Balle, C. Bernardez, A. Fjalling, J. Karlsen, and M. Wahlberg. 2009. Transmission loss patterns from acoustic harassment and deterrent devices do not always follow geometrical spreading predictions. Mar. Mamm. Sci. 25: 53-67.

Shaughnessy, P.D., A. Semmelink, J. Cooper, and P.G.H. Frost. 1981. Attempts to develop acoustic methods of keeping Cape Fur seals, *Arctocephalus pusillus,* from fishing nets. Biological Conservation 21: 141-158

Sivian, L.J. and S.D. White. 1933. On minimum audible sound fields. J. Acoustic. Soc. Amer. 4: 288-321.

Smith, M.E., S. Andrew, and A.N. Popper. 2004. Noise-induced stress response and hearing loss in goldfish, *Carassius auratus*. J. Exp. Biol. 207: 427–435.

Sonny, D., F.R. Knudsen, P.S. Enger, T. Kvernstuen, and O. Sand. 2006. Reactions of cyprinids to infrasound in a lake and at the cooling water inlet of a nuclear power plant. J. Fish Biol. 69: 735-748.

Southall, B. L., A.E. Bowles, W.T. Ellison, J.J. Finneran, R.L. Gentry, C.R. Greene Jr., D. Kastak, D.R. Ketten, J.H. Miller, P.E. Nachtigall, W.J. Richardson, J.A. Thomas, and P.L. Tyack. 2007. Marine mammal noise exposure criteria: Initial scientific recommendations. Aquat. Mamm. 33: 411-522.

Strong, M., E. Trippel, D. Clark, J. Neilson, and B. Chang. 1995. Potential impacts of use of acoustic deterrent devices (ADDs) on marine mammals in the Quoddy region based on a study conducted in British Columbia. DFO Atlantic Fisheries Res. Doc. 95/127. 16 pp.

Szymanski, M.D., D.E. Bain, K. Kiehl, S. Pennington, S. Wong, and K.R. Henry. 1999. Killer whale (*Orcinus orca*) hearing: auditory brainstem response and behavioral audiograms. J. Acoustic. Soc. Amer.a 106: 1134–1141.

Tavolga, W.N. 1964. Marine Bio-Acoustics. Pergammon Press, New York. 413 pp.

Taylor, V. J., D. W. Johnston, and W. C. Verboom. 1997. Acoustic harassment device (AHD) use in the aquaculture industry and implications for marine mammals. Proceedings of the Institute of Acoustics, 19: 267-275.

Terhune, J.M. 1988. Detection thresholds of a harbour seal to repeated underwater high-frequency, short-duration sinusoidal pulses. Can. J. Zool. 66: 1578-1582.

Terhune, J.M. 1989. Underwater click hearing thresholds of a harbor seal. Aquat. Mamm. 15: 22–26.

Terhune, J.M., C.L. Hoover, and S.R. Jacobs. 2002. Potential detection and deterrence ranges by harbor seals of underwater acoustic harassment devices (AHD) in the Bay of Fundy, Canada. J. World Aquacult. Soc. 33: 176-183.

Terhune, J.M. and K. Ronald. 1972. The harp seal *Pagophilus groenlandicus* (Erxleben, 1777). III. The underwater audiogram. Can. J. Zool. 50: 565–569.

Terhune, J.M. and K. Ronald. 1975. Underwater hearing sensitivity of two ringed seals. Can. J. Zool. 53: 227-231.

Terhune, J. M and S.D. Turnbull. 1995. Variation in the psychometric functions and hearing thresholds of a harbour seal. pp. 81-93. In R.A. Kastelein, J. A. Thomas and P. E. Nachtigall [eds.]. Sensory Systems of Aquatic Mammals. . DeSpil Publishers, Woerden, The Netherlands.

Thiery, L., and C. Meyer-Bisch. 1988. Hearing loss due to partly impulsive industrial noise exposure at levels between 87 and 90 dB (A). J. Acoustic. Soc. Amer. 84: 651-659.

Thomas, J., J. Pawloski, and W.W.L. Au. 1990. Masked hearing abilities of a false killer whale (*Pseudorca crassidens*) in Sensory abilities of Cetaceans, edited by J. Thomas and R. Kastelein Plenum, New York, pp. 395–404.

Thompson, D., M. Sjoberg, M.E. Bryant, P. Lovell, and A. Bjorge. 1998. Behavioural and physiological responses of harbour (*Phoca vitulina*) and grey (*Halichoerus grypus*) seals to seismic surveys. Report to European Commission of BROMMAD Project. MAS2 C7940098.

Tougaard, J. Carstensen, J. Teilmann, H. Skov, and P. Rasmussen. 2009. Pile-driving zone of responsiveness extends beyond 20 km for harbour porpoises (*Phocoena phocoena* (*L.*)). J. Acoust.

Soc. Am. 126: 11-14.

Trippel, E.A. M.B. Strong, J.M. Terhune, and J.D. Conway. 1999. Mitigation of harbour porpoise (*Phocoena phocoena*) by-catch in the lower Bay of Fundy. Can. J. Fish. Aquat. Sci. 56: 113-123.

Turnbull, S.D. and J.M. Terhune. 1990. White noise and pure tone masking of pure tone thresholds of a harbour seal listening in air and underwater. Can.J. Zool. 68:2090-2097

Tyack, P.L. 1998. Acoustic communication under the sea. pp. 163-220. In Hopp, S.L.. M.J. Owren, and C.S. Evans [eds.]. Animal Acoustic Communication. Berlin: Springer-Verlag.

Tyack, P.L. 2008. Implications for marine mammals of large-scale changes in the marine acoustic environment. J. Mamm. 89: 549-558.

Urick, R.J. 1983. Principles of Underwater Sound, 3rd ed., McGraw-Hill, NewYork. 423 pp.

Vabø, R., K. Olsen, and I. Huse. 2002. The effect of vessel avoidance of wintering Norwegian spring spawning herring. Fish. Res. 58: 59–77.

Wahlberg, M. 2006. Sound propagation of signals from two pingers and an acoustic harassment device in shallow waters. Report for Naturens Stemme and Aarhus University. 11p.

Ward, W.D. 1997. Effects of high-intensity sound. pp: 1497-1507 In M.J. Crocker [ed.]. Encyclopedia of acoustics, Volume III. John Wiley & Sons, New York.

Wartzok, D. and D.R. Ketten. 1999. Marine mammal sensory systems. pp 117-185 In J.E. Reynolds II and S.A. Rommel [eds.]. Biology of Marine Mammals. Washington, DC: Smithsonian Institution Press.

Wartzok, D., A.N. Popper, J. Gordon, and J. Merrill. 2004. Factors affecting the responses of marine mammals to acoustic disturbance. Marine Technology Soc. J. 37:6-15.

Watkins, W. A. and D. Wartzok. 1985. Sensory biophysics of marine mammals. Mar. Mamm. Sci. 1:219-260.

Webb, J. F., R.R. Fay, and A.N. Popper [eds.]. 2008. In Fish Bioacoustics. Springer Science and Business Media New York, NY.

Williams, R. M., A. W. Trites, and D.E. Bain. 2002. Behavioural responses of killer whales (*Orcinus orca*) to whale-watching boats: Opportunistic observations and experimental approaches. J. Zool. (London) 256: 255-270.

Wilson, B. and L.M. Dill. 2002. Pacific herring respond to simulated odontocete echolocation sounds. Can. J. Fish. Aquat. Sci. 59: 542–553.

Wursig, B. and G.A. Gailey. 2002. Marine mammals and aquaculture: conflicts and potential resolutions. pp. 45-59 In R.R. Stickney and J.P. Mcvey [eds.]. Responsible Marine Aquaculture. CABI Publishing, UK. 416 pp.

Wursig, B., C.R. Greene, Jr., and T.A. Jefferson. 2000. Development of an air bubble curtain to reduce underwater noise of percussive piling. Marine Environmental Research 49: 79-93

Wysocki, L.E., J.W. Davidson III, M.E. Smith, A.S. Frankel, W.T. Ellison, P.M. Mazik, A.N. Popper, and J. Bebak. 2007. Effects of aquaculture production noise on hearing, growth, and disease resistance of Rainbow trout, *Oncorhynchus mykiss*. Aquaculture 272: 687-697.

Yelverton, J.T. 1981. Underwater explosion damage risk criteria for fish, birds, and mammals.

Manuscript presented at 102nd Meet. Acoust. Soc. Am., Miami Beach, FL. Dec. 1981.

Yelverton, J.T., D.R. Richmond, E.R. Fletcher, and R.K. Jones. 1973. Safe distances from underwater explosions for mammals and birds. DNA 3114T. Rep. from Lovelace Foundation for Medical Educ. and Res., Albuquerque, NM for Defense Nuclear Agency, Washington, DC. 67 pp. NTIS AD-766952.

Yost, W.A. 2000. Fundamentals of Hearing: An Introduction, 4th edition. Academic Press, New York. 349 pp.

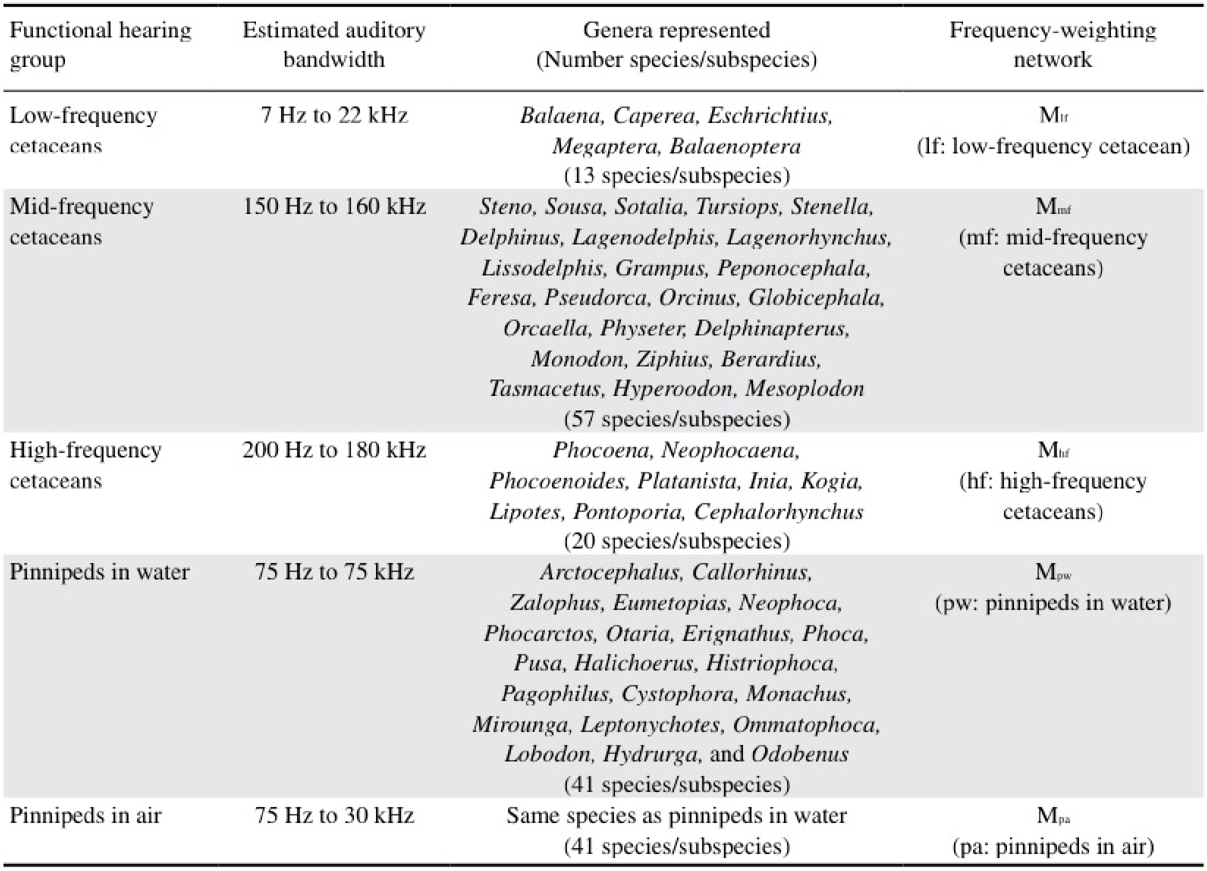
Yuen, M.M.L., P.E. Nachtigall, M. Breese, and A.Y. Supin. 2005. Behavioral and auditory evoked potential audiograms of a false killer whale (*Pseudorca crassidens*). J. Acoust. Soc. America 118: 2688-2695.

Yurk H. and A.W. Trites. 2000. Experimental attempts to reduce predation by harbor seals on out-migrating juvenile salmonids. Trans. Amer. Fish. Soc. 129:1360-1366.

Table 1. *General overview of the sources, amplitude, characteristics and prevalence of noise associated with aquaculture in Canada.*

Noise Source	Characteristics			Prevalence		References
	Intensity or Amplitude	Type	Frequency	Geographic	Temporal	
Predator Deterrents						
Cracker shells	170-235 dB re: 1 $\mu Pa^2 \cdot s$	Single Pulse	Broad 10-400 Hz	Formerly Common, Not Widely Used Currently	Intermittent	Awbrey and Thomas 1987
Sea lion bombs	190 dB re: 1 $\mu Pa^2 \cdot s$ 220 dB re: 1 μPa @ 1 m p-p	Single Pulse	Broad 200 Hz – 10 kHz	Rarely Used	Intermittent	Awbrey and Thomas 1987; Myrick et al. 1990b
Acoustic Harassment Devices	174-203 dB re: 1 $\mu Pa^2 \cdot s$ 191-235 dB re: 1 μPa @ 1 m p-p	Multi-Pulse Continuous	Narrow 10 kHz or 38 kHz	Potentially Widespread If Allowed	Continuous	Haller and Lemon (1994)
Pulsed Power Deterrents	1-3 kJ 240 dB re: 1 μPa @ 1 m at 1.8 kJ	Multi-Pulse	N/A	Under Development - Tests At Full Power or With Animals Have Not Been Approved		NMFS[7]
Operational Noise						
Large Outboard Motor	175 dB re:1 μPa @ 1 m rms	Non-Pulse	Broad 100 Hz – 20 kHz	Widespread	Routine	Richardson et al. 1995
Zodiac (5m 25hp)	152 dB re:1 μPa @ 1 m rms	Non-Pulse	Broad Peak at 6 kHz	Widespread	Routine	Malme et al. 1989
Crew Boat (16m)	156 dB re:1 μPa @ 1 m rms	Non-Pulse	Broad Peak at 90 Hz	Widespread	Routine	Greene 1985
Tugboat (Range of Loads and Speeds)	161 - 170 dB re:1 μPa @ 1 m rms	Non-Pulse	Broad 1-5 kHz	Widespread	Occasional	Miles et al. 1987
Fishing Boat (12 m at 7 Knots)	151 dB re:1 μPa @ 1 m rms	Non-Pulse	Broad 250 Hz – 1 kHz	Widespread	Routine	Miles et al. 1987
Whale-Watching Boats (Slow to ~50 $km \cdot hr^{-1}$)	145-169 dB re:1 μPa @ 1 m rms	Non-Pulse	Broad 100 Hz - 20 kHz	Widespread	Routine	Erbe 2002
Sonar (Fish-Finders)	210-250	Multi-Pulse (Very Brief)	Narrow 50 & 200 kHz	Widespread	Routine	Richardson et al. 1995; NRC 2003
Generators (125 kW to 2,000 kW)	86-99 dB re: 20μPa @ 1 m rms	Non-Pulse	Broad	Widespread	Continuous	ASHRAE 2002
Chain saws, pressure washers, etc.	90-110 dB re: 20μPa @ 1 m rms	Non-Pulse	Broad 630 Hz – 2 kHz	Widespread	Routine	NICDC 2009
Construction and Demolition						
Pile-Driving	Up to 235 dB re: 1 μPa @ 1 m p-p	Multi-Pulse	Broad Peak at 160 Hz	Localized	Rare	Madsen et al. 2006; Tougaard et al. 2009
Explosives	Variable But High	Single Pulse Shock Wave	1-100 Hz	Localized	Rare	Richardson et al. 1995

Table 2. *Functional marine mammal hearing groups, auditory bandwidth (estimated lower to upper hearing thresholds) and group-specific (M) frequency-weightings (from Southall et al. 2007).*



Functional hearing group	Estimated auditory bandwidth	Genera represented (Number species/subspecies)	Frequency-weighting network
Low-frequency cetaceans	7 Hz to 22 kHz	*Balaena, Caperea, Eschrichtius, Megaptera, Balaenoptera* (13 species/subspecies)	M_{lf} (lf: low-frequency cetacean)
Mid-frequency cetaceans	150 Hz to 160 kHz	*Steno, Sousa, Sotalia, Tursiops, Stenella, Delphinus, Lagenodelphis, Lagenorhynchus, Lissodelphis, Grampus, Peponocephala, Feresa, Pseudorca, Orcinus, Globicephala, Orcaella, Physeter, Delphinapterus, Monodon, Ziphius, Berardius, Tasmacetus, Hyperoodon, Mesoplodon* (57 species/subspecies)	M_{mf} (mf: mid-frequency cetaceans)
High-frequency cetaceans	200 Hz to 180 kHz	*Phocoena, Neophocaena, Phocoenoides, Platanista, Inia, Kogia, Lipotes, Pontoporia, Cephalorhynchus* (20 species/subspecies)	M_{hf} (hf: high-frequency cetaceans)
Pinnipeds in water	75 Hz to 75 kHz	*Arctocephalus, Callorhinus, Zalophus, Eumetopias, Neophoca, Phocarctos, Otaria, Erignathus, Phoca, Pusa, Halichoerus, Histriophoca, Pagophilus, Cystophora, Monachus, Mirounga, Leptonychotes, Ommatophoca, Lobodon, Hydrurga,* and *Odobenus* (41 species/subspecies)	M_{pw} (pw: pinnipeds in water)
Pinnipeds in air	75 Hz to 30 kHz	Same species as pinnipeds in water (41 species/subspecies)	M_{pa} (pa: pinnipeds in air)

Table 3. *Severity scale for ranking observed behavioural responses of free-ranging marine mammals and laboratory animals to various types of anthropogenic noise (from Southall et al. 2007).*

Response score[1]	Corresponding behaviors (Free-ranging subjects)[2]	Corresponding behaviors (Laboratory subjects)[2]
0	- No observable response	- No observable response
1	- Brief orientation response (investigation/visual orientation)	- No observable response
2	- Moderate or multiple orientation behaviors - Brief or minor cessation/modification of vocal behavior - Brief or minor change in respiration rates	- No observable negative response; may approach sounds as a novel object
3	- Prolonged orientation behavior - Individual alert behavior - Minor changes in locomotion speed, direction, and/or dive profile but no avoidance of sound source - Moderate change in respiration rate - Minor cessation or modification of vocal behavior (duration < duration of source operation), including the Lombard Effect	- Minor changes in response to trained behaviors (e.g., delay in stationing, extended inter-trial intervals)
4	- Moderate changes in locomotion speed, direction, and/or dive profile but no avoidance of sound source - Brief, minor shift in group distribution - Moderate cessation or modification of vocal behavior (duration ≈ duration of source operation)	- Moderate changes in response to trained behaviors (e.g., reluctance to return to station, long inter-trial intervals)
5	- Extensive or prolonged changes in locomotion speed, direction, and/or dive profile but no avoidance of sound source - Moderate shift in group distribution - Change in inter-animal distance and/or group size (aggregation or separation) - Prolonged cessation or modification of vocal behavior (duration > duration of source operation)	- Severe and sustained changes in trained behaviors (e.g., breaking away from station during experimental sessions)
6	- Minor or moderate individual and/or group avoidance of sound source - Brief or minor separation of females and dependent offspring - Aggressive behavior related to noise exposure (e.g., tail/flipper slapping, fluke display, jaw clapping/gnashing teeth, abrupt directed movement, bubble clouds) - Extended cessation or modification of vocal behavior - Visible startle response - Brief cessation of reproductive behavior	- Refusal to initiate trained tasks
7	- Extensive or prolonged aggressive behavior - Moderate separation of females and dependent offspring - Clear anti-predator response - Severe and/or sustained avoidance of sound source - Moderate cessation of reproductive behavior	- Avoidance of experimental situation or retreat to refuge area (≤ duration of experiment) - Threatening or attacking the sound source
8	- Obvious aversion and/or progressive sensitization - Prolonged or significant separation of females and dependent offspring with disruption of acoustic reunion mechanisms - Long-term avoidance of area (> source operation) - Prolonged cessation of reproductive behavior	- Avoidance of or sensitization to experimental situation or retreat to refuge area (> duration of experiment)
9	- Outright panic, flight, stampede, attack of conspecifics, or stranding events - Avoidance behavior related to predator detection	- Total avoidance of sound exposure area and refusal to perform trained behaviors for greater than a day

[1] Ordinal scores of behavioral response severity are not necessarily equivalent for free-ranging vs laboratory conditions.

[2] Any single response results in the corresponding score (i.e., all group members and behavioral responses need not be observed). If multiple responses are observed, the one with the highest score is used for analysis.

FIGURES

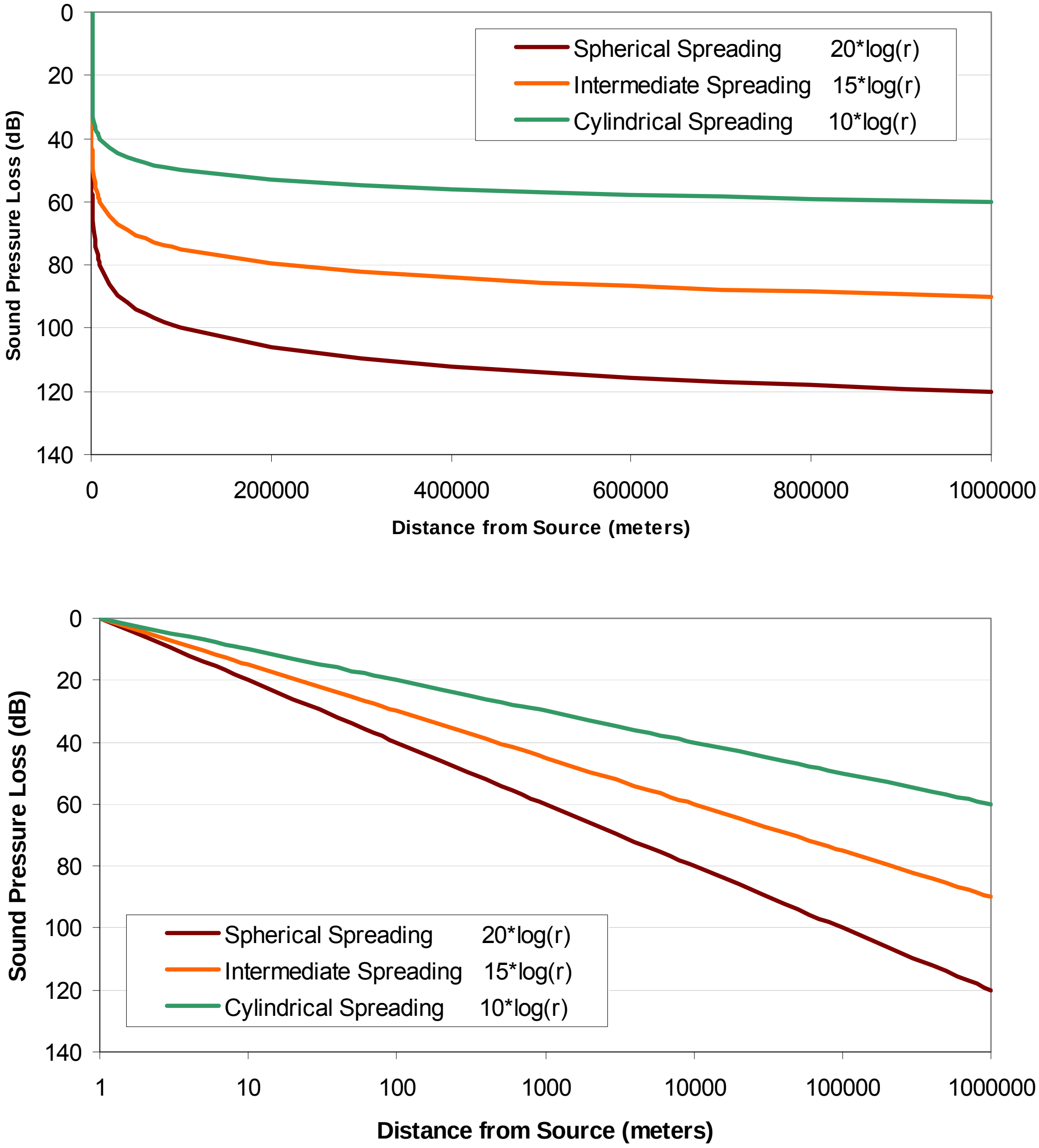


Figure 1. *Drop in sound pressure levels (dB) as a function of distance from the sound source (metres) as predicted by spherical [20·log(r)], intermediate [15·log(r)] and cylindrical [10·log(r)] spreading models. The top panel shows the decline on a linear scale, illustrating the abrupt drop in sound levels close to the sound source (30-60 dB in the first 1,000 metres) and more gradual drop in sound levels with increasing distance from the sound source. The bottom panel shows the decline on a logarithmic scale.*

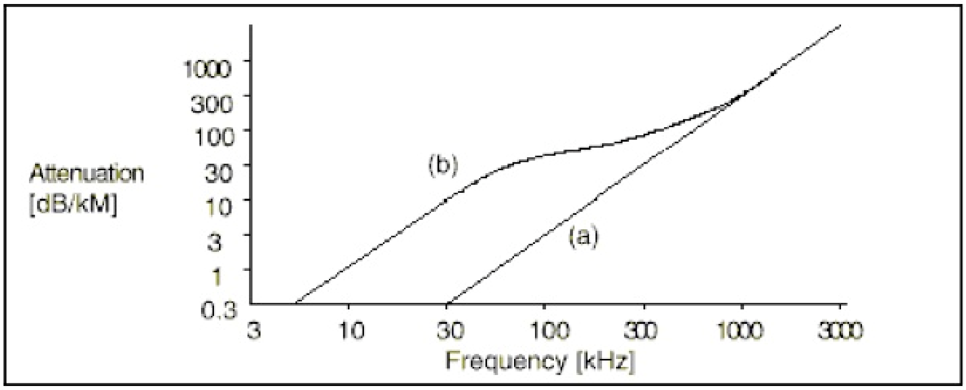


Figure 2. *Absorption coefficient (α) for sound in (a) freshwater, and (b) saltwater as a function of frequency (from Pincock and Voegeli 2002).*

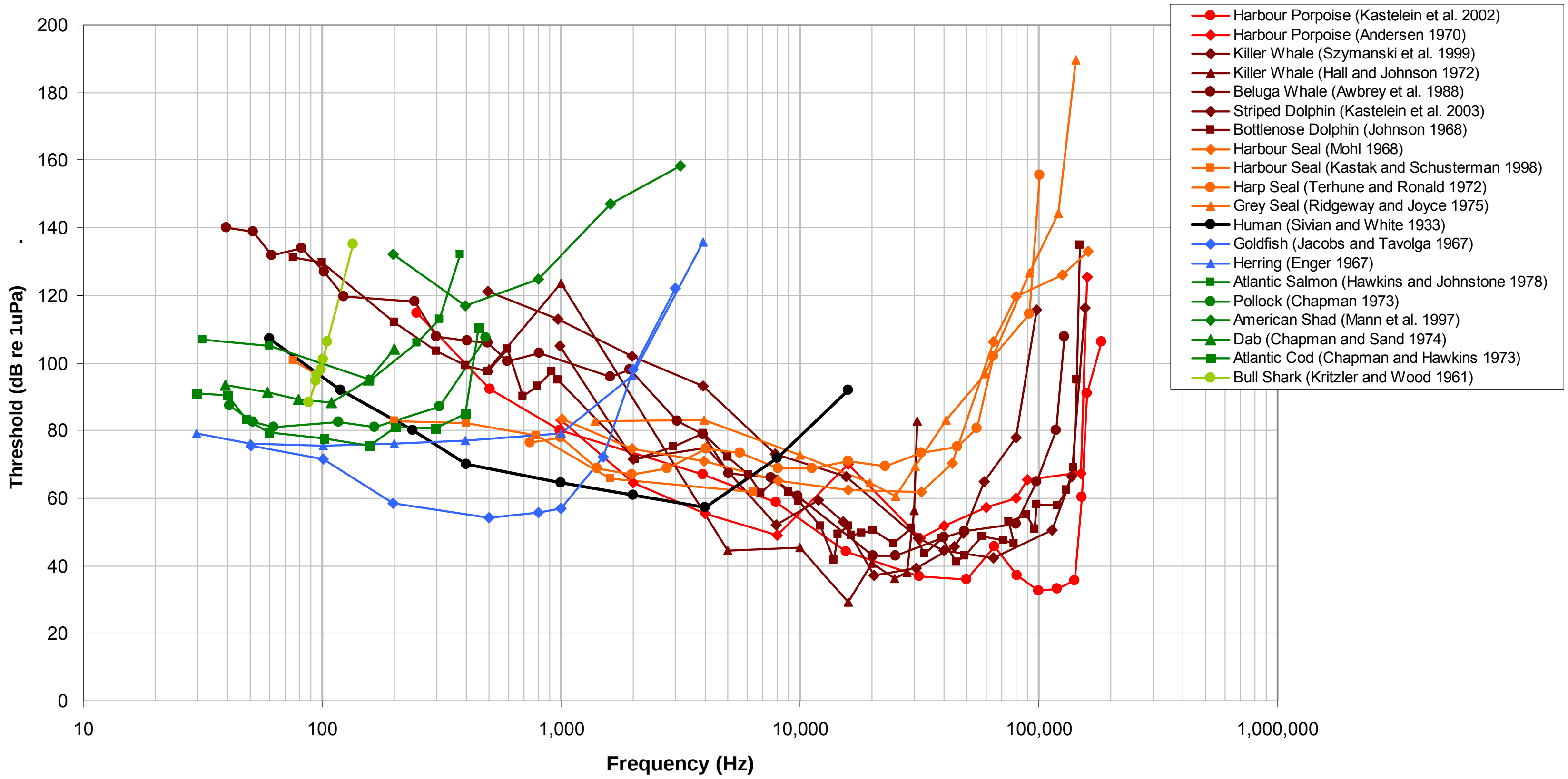


***Figure 3**. Audiograms showing hearing sensitivity as a function of frequency for various aquatic animals. Fish with generalized hearing structures are shown in green, fish with specialized hearing structures in blue, pinnipeds in orange, mid-frequency cetaceans in purple, and high-frequency cetaceans in red. For comparison, the hearing sensitivity of humans is shown in black.*

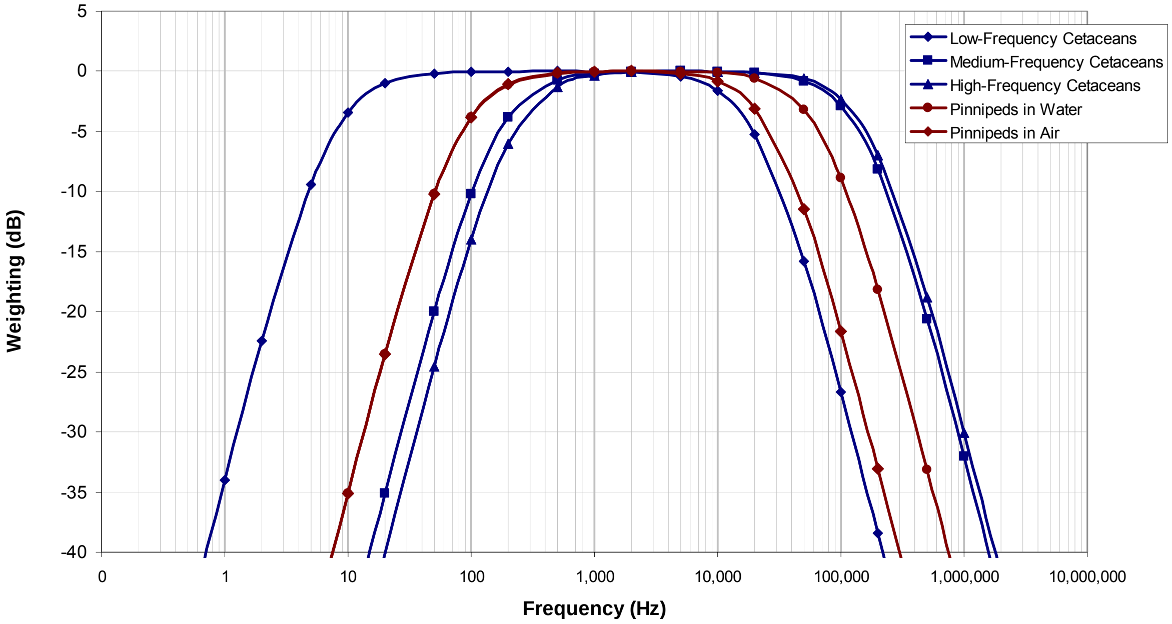


Figure 4 *M-weighting functions for de-emphasizing frequencies outside the range of greatest sensitivity for the 5 functional groups of marine mammals (after Southall et al. 2007). The M-weights are considered to be precautionary in that the functions tend to be flat over the entire hearing range of each group, and only frequencies outside the hearing range are de-emphasized.*

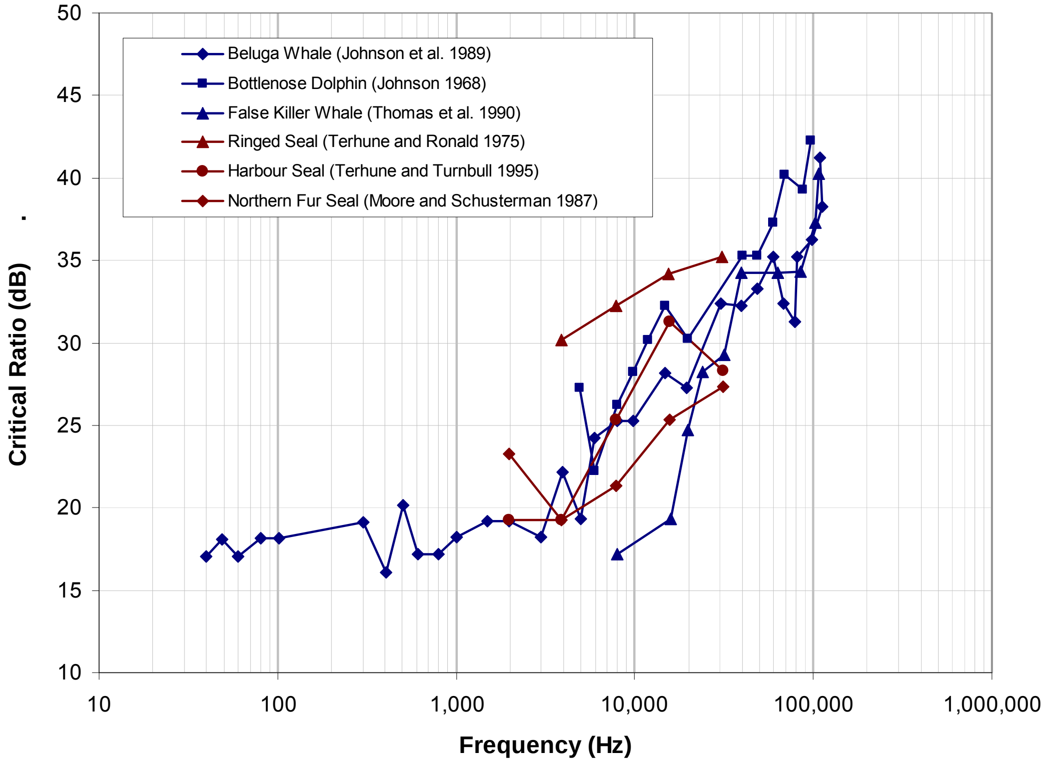


Figure 5 *Critical Ratios for representative cetaceans (blue) and pinnipeds (red) as a function of frequency.*

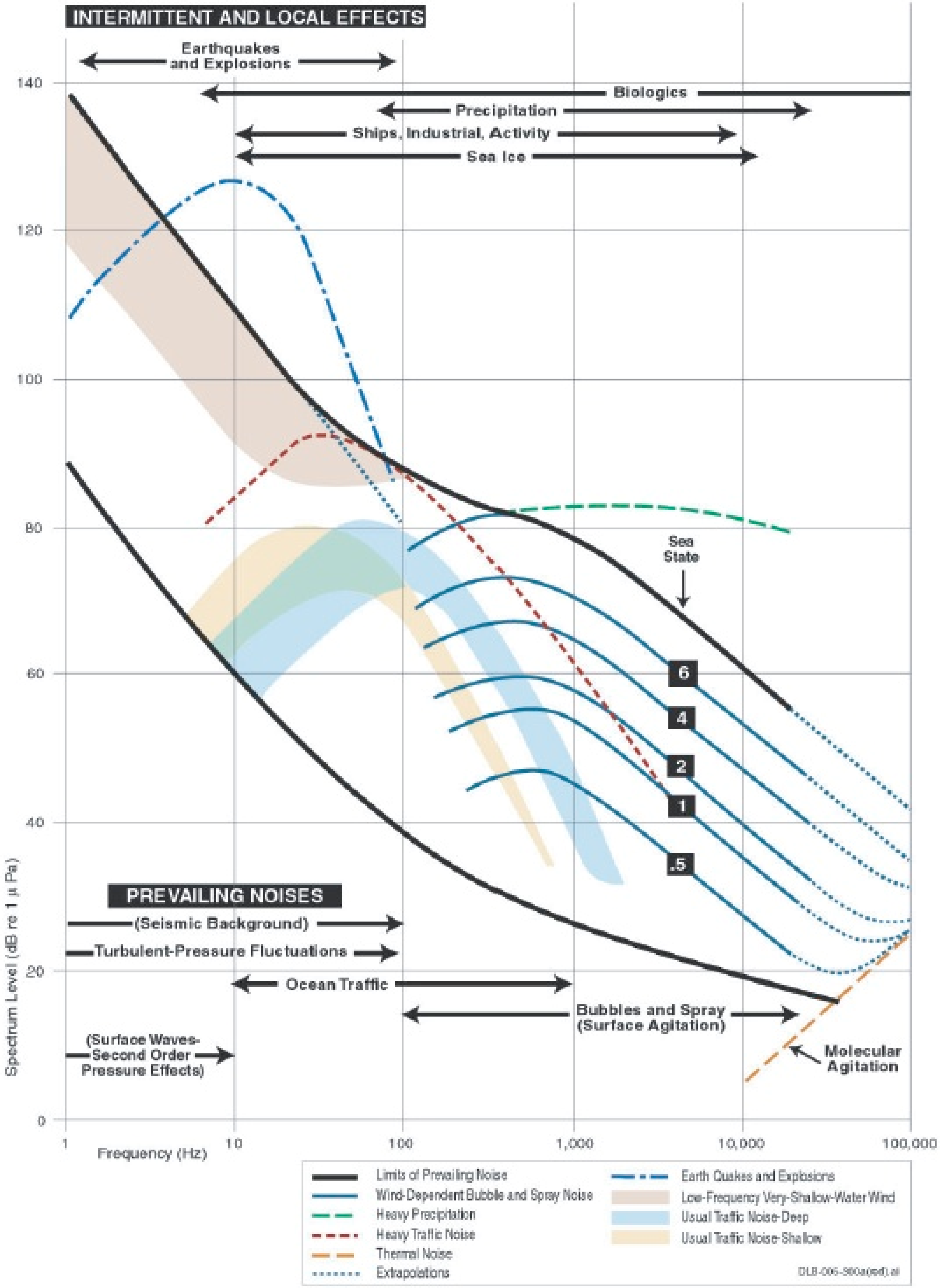


Figure 6. *Spectral curves showing marine ambient noise resulting from weather, wind, geologic activity and commercial shipping (adapted from Wenz 1962 from NRC 2003).*

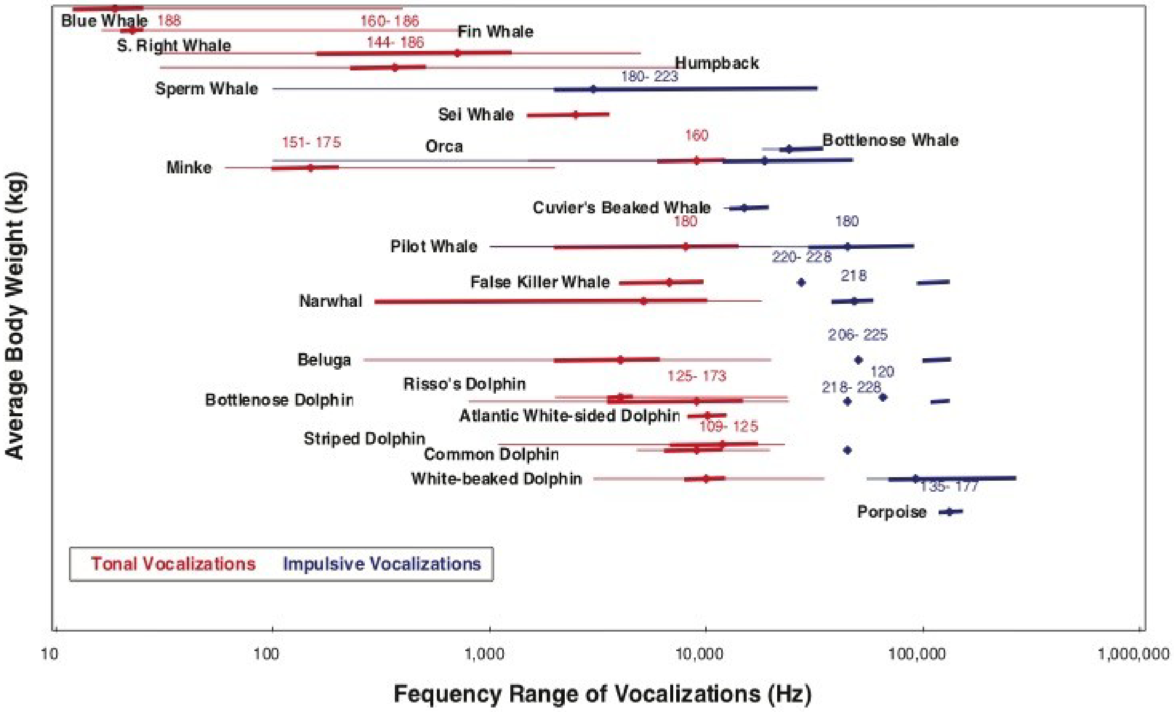


***Figure 7**. Range of sounds produced by cetaceans. Tonal vocalizations are plotted in red; impulsive vocalizations are shown in blue. The thicker lines indicate the frequencies near maximum energy, and the thin lines indicate the total range of frequencies in vocalizations. Numbers above the line indicate measured source levels in dB re µPa at 1m. Body weight data are taken from Table 10.1 in Boness et al. (2002); vocalization data are summarized from Table 4.1 in Wartzok and Ketten (1999) updated with additional information from Hooker and Whitehead (2002), Frantzis et al. (2002), Møhl et al. (2000), and Rasmussen et al. (2002) (from NRC 2003).*

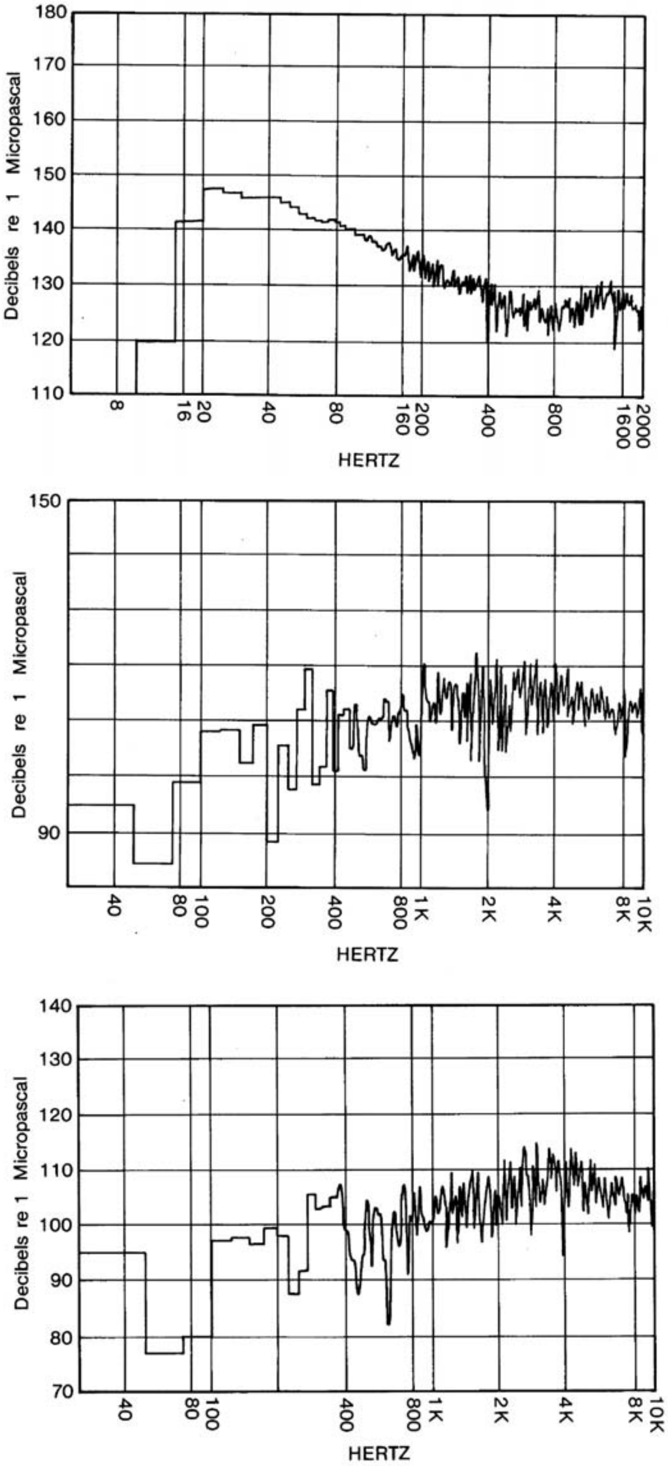


Figure 8 *Relative spectral energy for a seal bomb (top), flare gun fired cracker shell (middle) and shotgun fired cracker shell (bottom) (from Awbrey and Thomas 1987). Measurements were made at various distances, and are intended to illustrate the relative spectral energy rather than absolute energy levels (see text for signal strength levels extrapolated to 1 metre from source).*

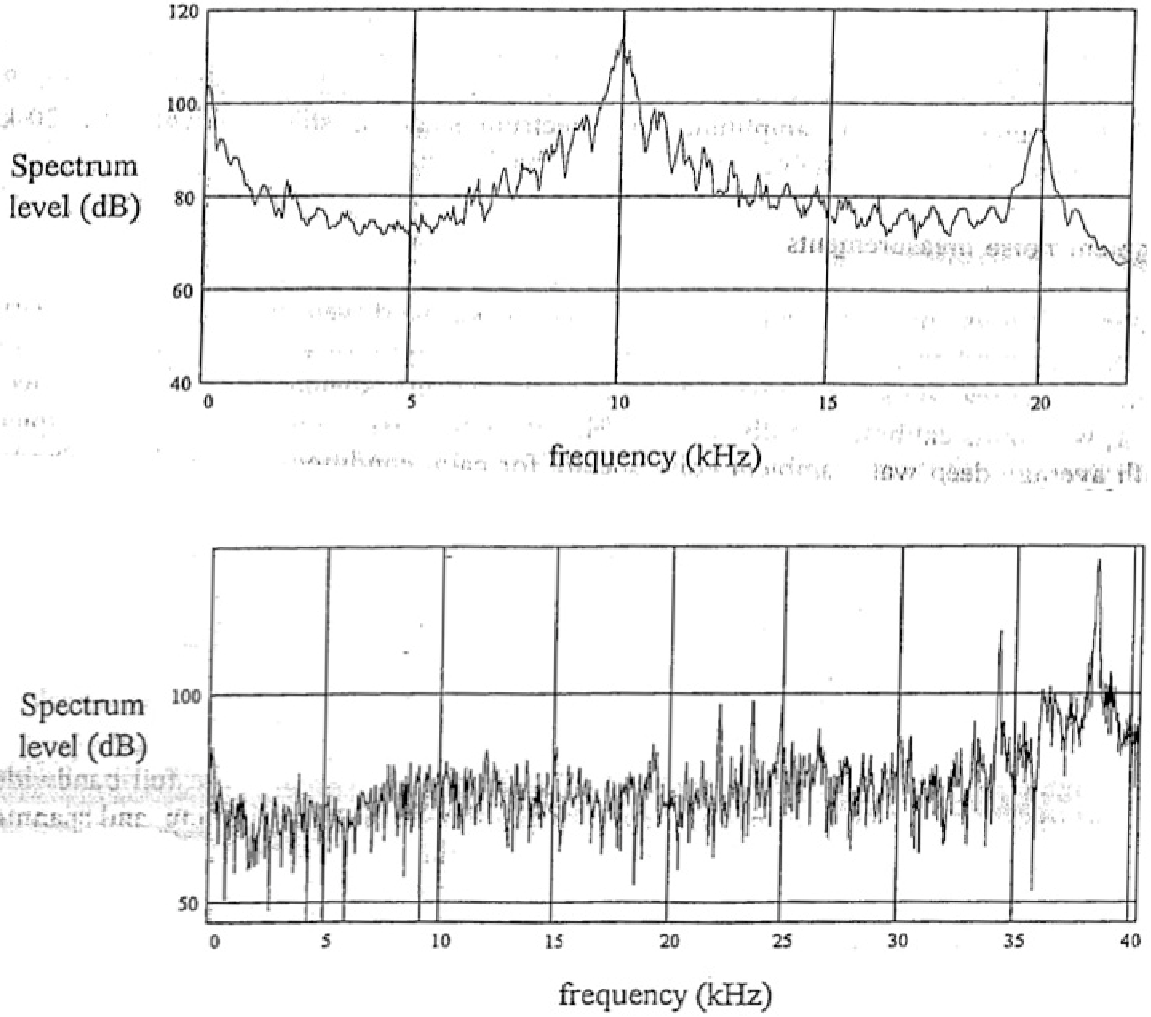


Figure 9. *Relative spectral energy of an AirMar (top) and Ferranti-Thomposon (bottom) AHD (from Haller and Lemon 1994). Measurements were made at various distances, and are intended to illustrate the relative spectral energy rather than absolute energy levels (see text for signal strength levels extrapolated to 1 metre from source).*

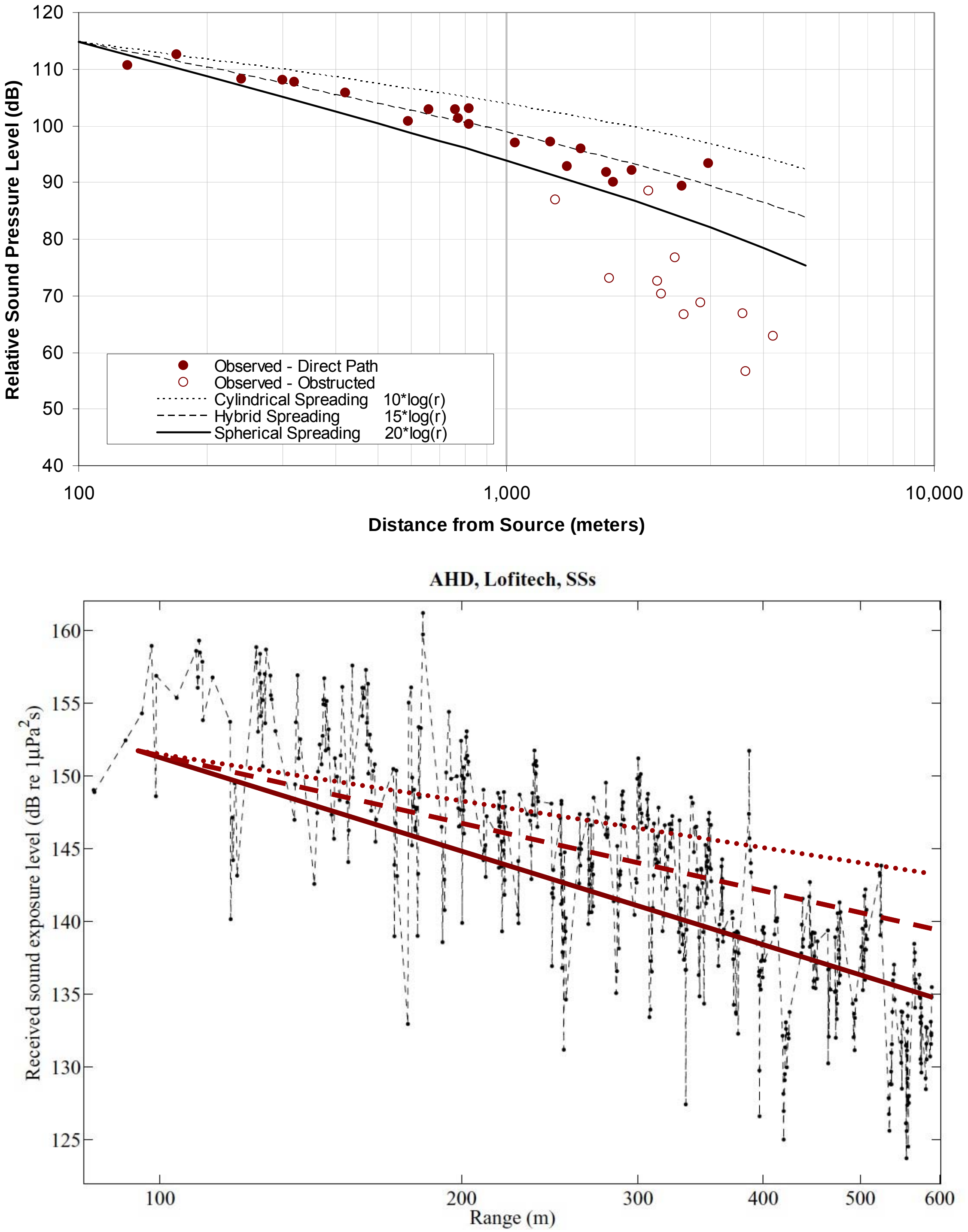


***Figure 10**. Propagation of AHD sounds relative compared to spherical (solid), hybrid spherical-cylindrical (dashed) and cylindrical (dotted) spreading models with allowance for absorption as per equation [7]. The top panel shows long-range propagation for an AirMar device, indicating that loss tends to follow a hybrid model in an unobstructed path (data from Haller and Lemon 1994). The bottom panel illustrates fine-scale variability for an Lofitech AHD (from Shapiro et al. 2009).*

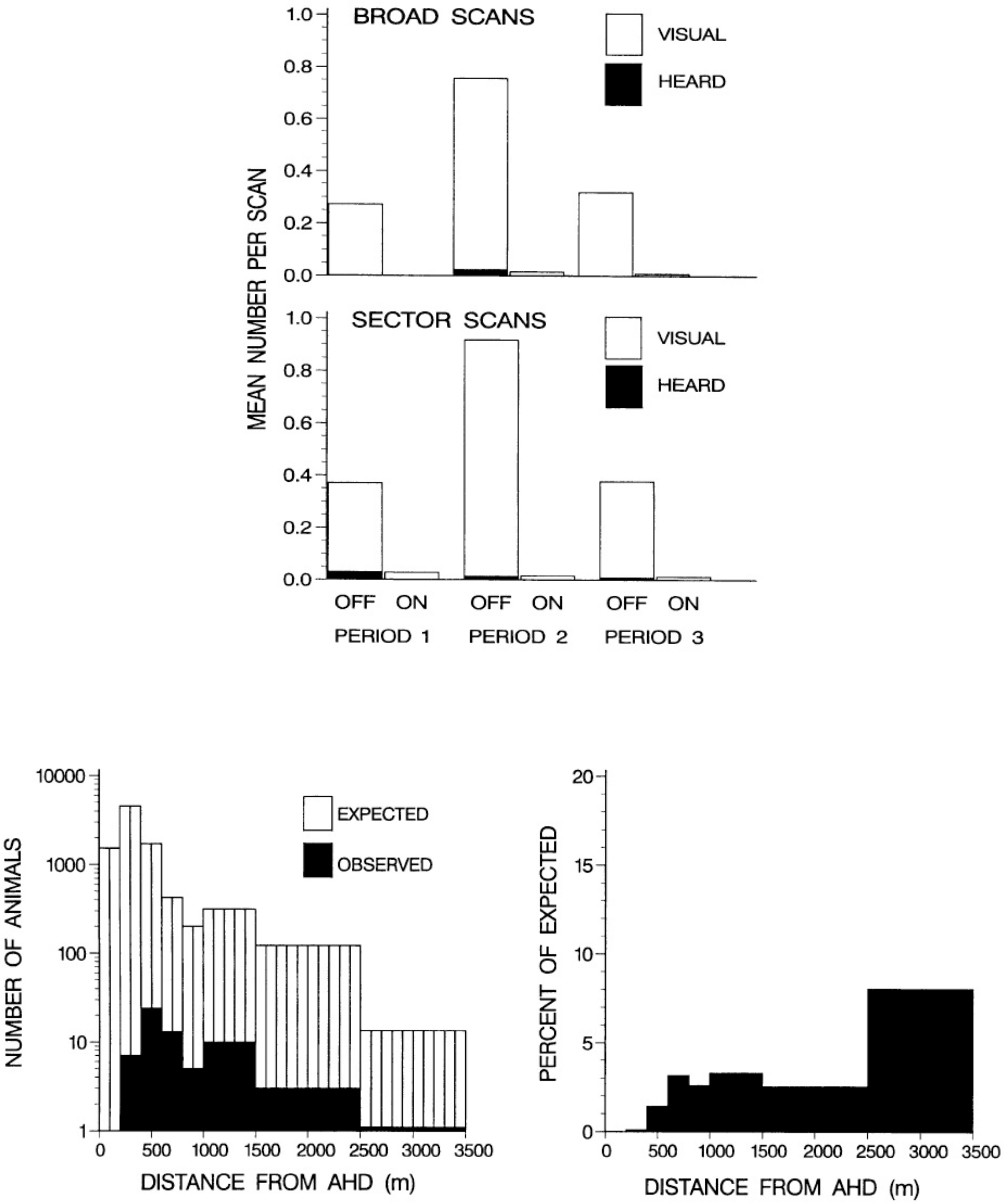


Figure 11. *Decline in the abundance (top) and relative change in distribution (bottom) of harbour porpoise in the study area during periods an AirMar AHD was activated compared to control periods (from Olesiuk et al. 2002).*

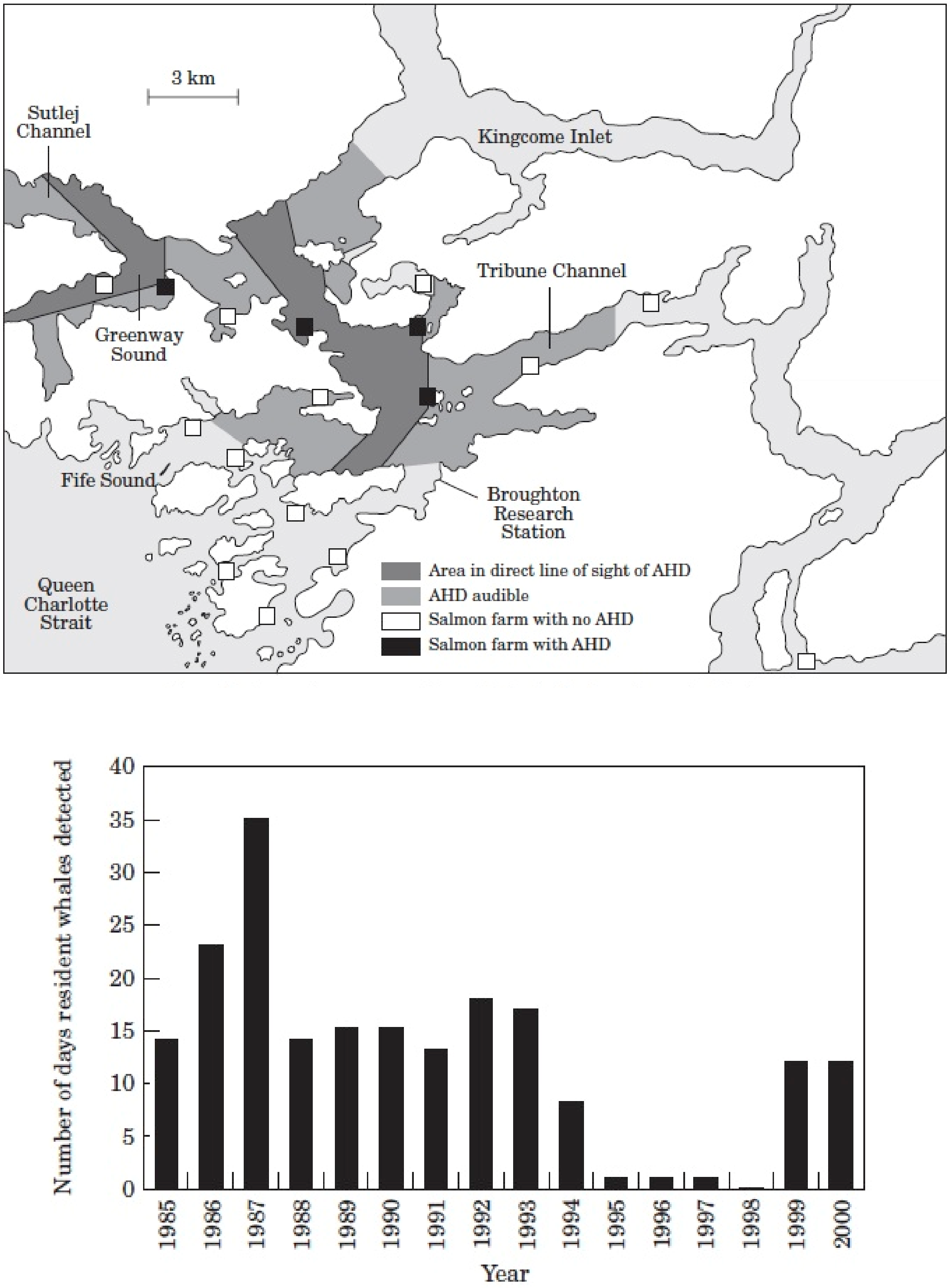


***Figure 12**. Map showing approximate coverage of AHD sounds in the Broughton Archipelago (top), and change in frequency of resident (fish-eating) killer whale sightings when during the 1994-1999 AHD deployment period (from Morton and Symonds 2002).*

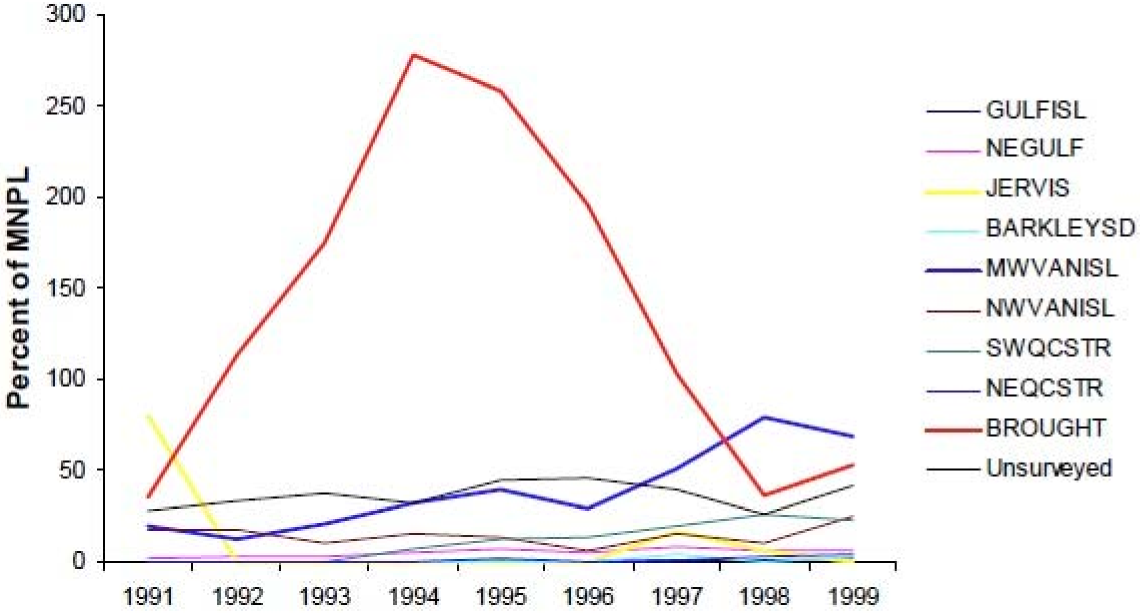


Figure 13. *Level of harbour seals predator control kills at salmon farms in British Columbia during 1991-1999. Kills are expressed as a percentage of maximum net productivity levels (i.e., the number of kills that could be sustained without depleting the population) (from Jamieson and Olesiuk 2001).*

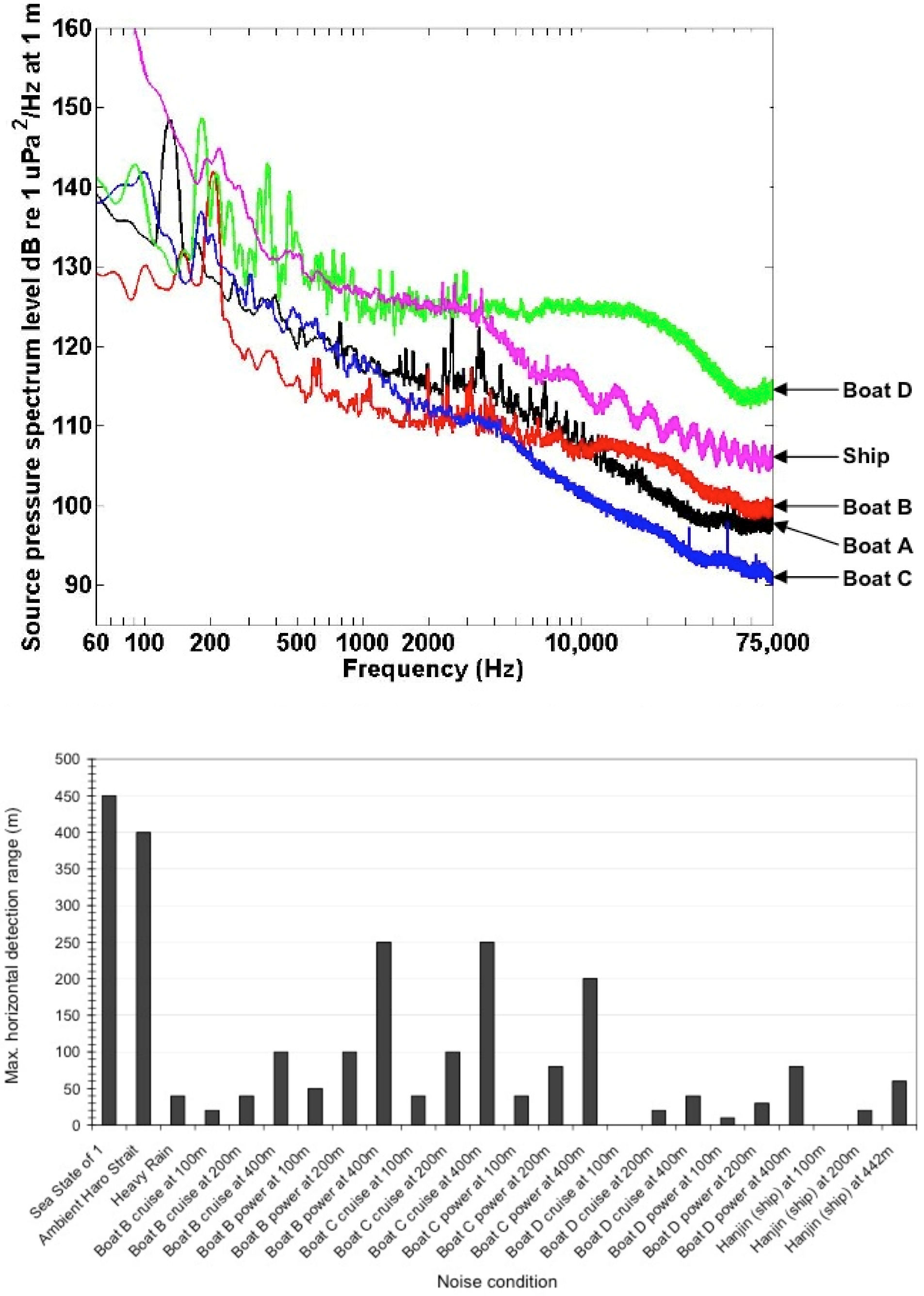


Figure 14. *Spectrum levels for various vessels (top): Boat A was a 20 ft fiberglass monohull with 90 HP outboard at power-up; Boat B was a 29 ft aluminium monohull with twin 225 HP outboard motors @24 kts; Boat C was a 38 foot aluminium catamaran with jet drives at 31 kts; Boat D was a 50 foot monohull with 3 inboard/outboard stern drives at 23 kts; and the ship was a 290 metre container ship (Hildebrand et al. 2006 cited in Holt 2008). Bottom panel shows the estimated change in range that a killer whale using echolocation would be able to detect a chinook salmon as a function of background noise (see Holt 2008 for details).*

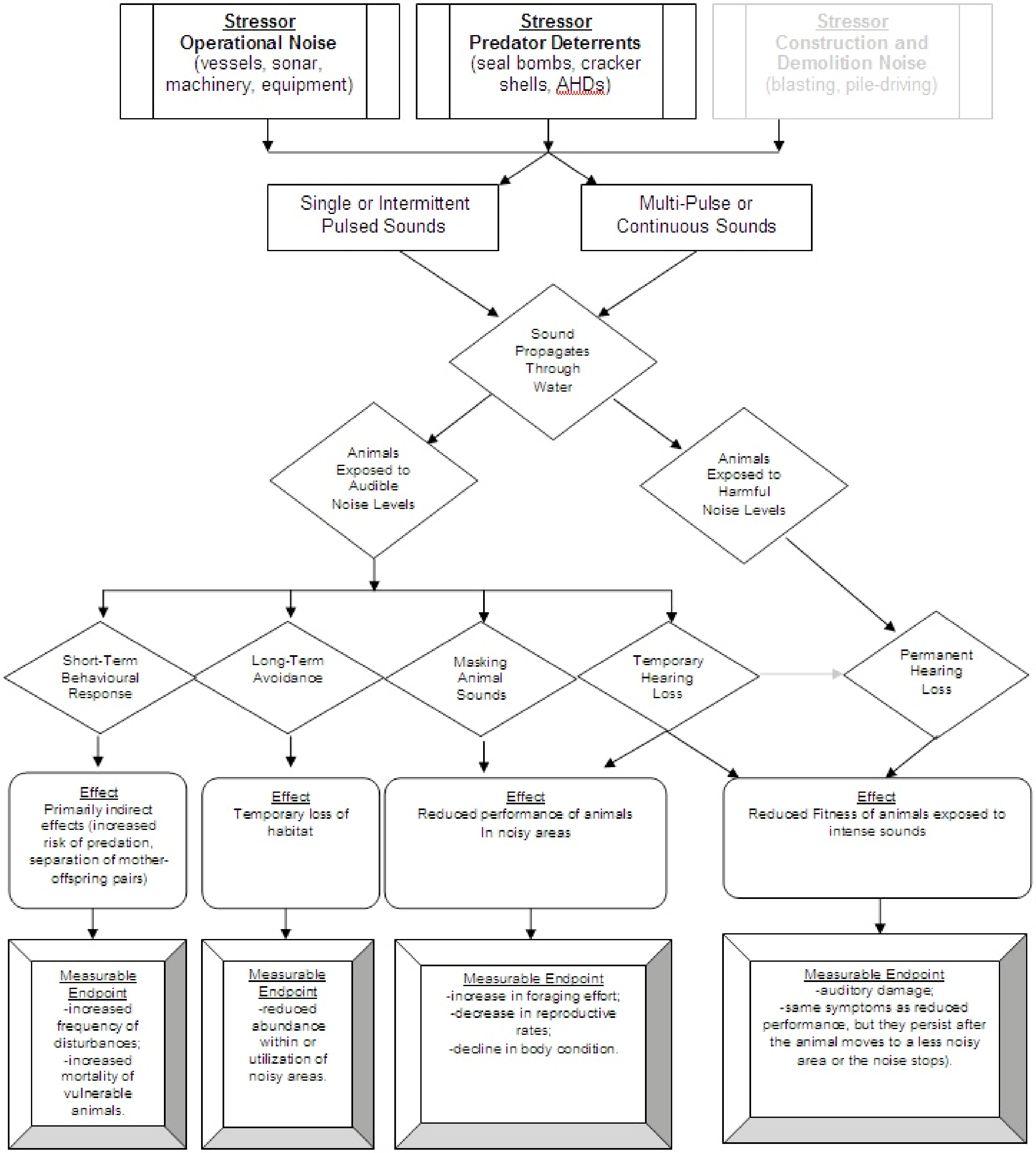


Figure 15. *Schematic illustrating pathways of effects of noise associated with aquaculture on natural aquatic ecosystems in Canada. Noise associated with construction and demolition was considered outside the scope of this review, so is shown in grey.*