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Sound Exposure and Southern Resident Killer Whales *(Orcinus orca)*:

A Review of Current Knowledge and Data Gaps

February 2008

U.S. DEPARTMENT OF COMMERCE
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A Review of Current Knowledge and Data Gaps

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Executive Summary

This document reviews what is currently known about potential acoustic impacts on endangered Southern Resident killer whales (SRKWs). Killer whales (*Orcinus orca*) use sound for echolocation, social communication, and passive listening. Ambient noise, including that from natural and anthropogenic sources, has the potential to interfere with the reception and use of these important biological sounds. Significant sources of anthropogenic sounds that contribute to ambient background noise in critical habitats of SRKWs include sonar, acoustic harassment devices, vessel traffic, and construction noise.

Most measurements of ambient sounds made in SRKW habitat are greatly influenced by vessel traffic that, at close ranges, raises noise levels significantly above ambient levels. In order to address potential acoustic impacts, particularly from anthropogenic sources, this document reviews parameters of sound that are pertinent to the auditory capabilities of killer whales and various studies on noise effects in killer whales and other dolphins. The latter includes auditory ramifications such as auditory masking or hearing loss and behavioral effects such as disruption of foraging events or avoidance of an area. With this information, the document then incorporates information on the soundscape of SRKW habitat and defines zones of audibility, responsiveness, masking, and hearing loss and addresses the likelihood of acoustic impacts on the SRKW population.

Lastly, recommendations are made for future work in order to address gaps in information that, if available, would increase confidence in predicting the likelihood of acoustic impacts on SRKWs.

Acknowledgments

Sue Moore, Brad Hanson, Dawn Noren, Candice Emmons, and Mike Ford provided valuable feedback on writing this report. This work was funded by the National Research Council Postdoctoral Research Associateships Program and the Northwest Fisheries Science Center.

Abbreviations and Acronyms

ABR	auditory brainstem response
AHD	acoustic harassment device
CR	critical ratio
dB	decibel
DL	difference limen
DI	directivity index
EL	echo level
Hz	hertz
kHz	kilohertz
μPa	micropascal
ms	milliseconds
MTTS	masked temporary threshold shift
NL	noise level
Pa	pascal
PAL	passive acoustic listener
PCB	polychlorinated biphenyl
PTS	permanent threshold shift
RMS	root-mean-square
RL	received level
SL	source level
SEL	sound exposure level
SPL	sound pressure level
SRKW	Southern Resident killer whale
TL	transmission loss
TS	target strength
TTS	temporary threshold shift
VTOSS	Vessel Traffic Operations Support System

Introduction

Southern Resident killer whales (SRKWs) are an endangered population of approximately 85 individuals that spend the summer in inland waters surrounding British Columbia and Washington State. These killer whales (*Orcinus orca*) are fish eaters that typically feed on Pacific salmon (*Oncorhynchus* spp.). They actively use sound for echolocation and vocal communication. Additionally, they can glean information about the environment such as the presence of prey from passive listening. Both natural and anthropogenic sounds have the potential to impact the use of biologically important acoustic signals by SRKWs. Concern about anthropogenic sound exposure, such as those produced by vessels or military sonar, has provided the impetus to study and describe the acoustic environment that Southern Residents inhabit and the effects of sound exposure on their auditory system, behavior, and physiology.

The purpose of this paper is to review what is currently known about killer whale auditory capabilities and the use of sound by killer whales, the characteristics of sound in their environment, and effects of sound exposure in killer whales and other dolphins in order to address potential acoustic impacts on the SRKW population. As is the case for all marine mammal groups, it is extremely difficult to address acoustic effects that might have indirect or small but consistent consequences at the population level, as opposed to those with immediate and sometimes extreme outcomes. The life history patterns and habitats of marine mammals in general make this assessment extremely challenging (NRC 2005). In most cases, there is insufficient empirical data on which to draw in order to address acoustic impacts at the population level. A review of the current data and their limitations allows the opportunity to assess data gaps, which are summarized at the end of this document.

Basic Concepts

Given the various ways that sound energy is quantified, it is necessary to review some basic acoustical concepts. Sound is essentially generated when a vibrating object sets molecules in a medium adjacent to that object into motion. Sound amplitude or what is perceived as loudness is directly related to the amount of pressure generated by the vibrating object. In a compressible medium, the motion of molecules produces positive pressure where there is condensation and negative pressure where there is rarefaction of molecules. The intervals of condensation and rarefaction typically occur in a cyclical fashion. In a plane progressive wave of sound (when the acoustic pressure is the same in all planes perpendicular to the direction of propagation), the instantaneous pressure, p, generated in a compressible fluid can be described by

$$p = \rho c u \qquad (1)$$

where ρ equals the fluid density, c equals the speed of sound, and u equals the particle velocity. Acoustic pressure is typically measured as the root-mean-square (RMS) pressure average over the duration of the sound. For impulsive sound such as pile driving strikes or biosonar clicks, peak sound pressure (the range from zero to the greatest pressure of the signal) or peak-to-peak sound pressure (the range of the most positive to the most negative pressure of the signal) are often reported instead, since it is difficult to define an appropriate duration over which to average the signal's pressure (Madsen 2005). Pressure is typically reported in units of pascals (Pa) or micropascals (µPa). In a plane progressive wave, sound intensity is described by the sound power per unit area and is a product of the sound pressure and particle velocity by

$$I = pu \tag{2}$$

and substituting *u* from first equation, intensity of the sound, *I*, is related to *p* by

$$I = p(p/\rho c) = p^2/\rho c \tag{3}$$

where *p* is the RMS pressure average over the duration of the sound. Intensity is typically reported in units of watts per square meter. Sound levels are most often described in units of decibel (dB), which is traditionally defined as a power or intensity ratio. Sound intensity level in decibels is as follows:

$$dB = 10 \log_{10} (I_1/I_2) \tag{4}$$

where I_1 is the intensity of the sound of interest and I_2 is a reference intensity. In the case of a plane wave, sound pressure which is typically what is measured by a microphone or hydrophone may also be used to measure the sound's magnitude in dB. Because sound intensity is proportional to pressure squared, sound pressure level (SPL) in dB is given by

$$dB = 10 \log_{10} (p_1^2)/(p_2^2) = 20 \log_{10} (p_1/p_2) \tag{5}$$

where p_1 is the pressure of the sound of interest and p_2 is typically the standard reference pressure for a given medium. In water the reference is usually 1 µPa. SPLs in this document are referenced to the underwater convention (re 1 µPa) based on RMS measurements unless otherwise noted. This reference pressure is different from the standard used to measure sound pressure levels in air. Thus a dB (re 1 µPa) underwater is not equivalent to a dB (re 20 µPa) measured in air. Pulsed sounds such as explosions, seismic air gun pulses, or pile driving impacts are often measured in terms of their energy and not just pressure or intensity. Energy measures include time as a dimension and are also used to quantify sound exposure when both amplitude and duration of exposure is important. Energy is proportional to the time integral of the pressure squared and in dB sound exposure levels (SELs) has the units of dB re 1 μPa^2s.

Frequency Bandwidth

Amplitude, intensity, or energy measurements in dB are always dependent on the measurements integrated across a frequency bandwidth. Broadband SPL measurements (overall SPL) will be different from those based on one octave, one-third octave, narrower band, and spectral density level measurements. For example, sound pressure spectral density gives the mean squared pressure of a sound within a given frequency bandwidth divided by the measurement bandwidth and the units are pressure square per hertz. The decibel unit based on spectral density level is dB re 1 μPa^2 /Hz (or re 1 µPa $/Hz^{1/2}$). On the other hand, one-third octave levels in dB are based on the mean square pressure level for each one-third octave band.

An octave is a factor of two in frequency and sound levels are often reported in one-third octave bands because the effective filter bandwidth of the auditory system in humans and some animals is approximated by one-third octave (Richardson et al. 1995, but see the Auditory Capabilities section and Auditory Masking subsection below). The term noise in this document usually refers to the general definition of unwanted sound. For a further review of these concepts, see Urick (1983), Richardson et al. (1995), and the Web site http://www.dosits.org/science/intro.htm, hosted by the University of Rhode Island.

Southern Resident Killer Whales

Killer whales are the largest cetacean in the dolphin family, Delphinidae. The three identified ecotypes of killer whales in the northeastern Pacific Ocean are residents, transients, and offshores. While there is considerable overlap in their geographic range, these ecotypes are genetically distinct and do not appear to interbreed. The differences between ecotypes also extend to their morphology, foraging ecology, behavior, and acoustic repertoire. For example, residents are generally fish eaters while transients are generally mammal eaters (Ford et al. 2000) and less is known about the diet of offshores. Residents tend to live in larger, more stable groups consisting of multigenerational, matrilineal-related kin, while transients live in smaller, less stable groups usually consisting of females and a few offspring (Ford et al. 2000). Residents tend to be more vocal, particularly when foraging and socializing, while transients are acoustically cryptic presumably because their prey can hear within the frequency range of their sound emissions (Barrett-Lennard et al. 1996, Deecke et al. 2002, Deecke et al. 2004).

Along the U.S. and Canadian west coast, there are currently four communities of resident killer whales that have been identified: Northern, Southern, Southern Alaska, and Western Alaska Residents (Krahn et al. 2004). SRKWs consist of three pods—J, K, and L pod—found during the late spring to early autumn in the inland waters of Washington State and British Columbia. Members are individually identified based on photo identification records of natural markings. Like all marine mammals, they are long-lived and slow to mature.

Both male and female resident killer whales of the area do not become sexually mature until an average age of 15 years and females produce an average of 5.5 surviving offspring (Olesiuk et al. 1990). Annual population censuses indicate that SRKW numbers experienced a population decline in the mid to late 1990s. This distinct population segment of killer whales has been listed under the Endangered Species Act of 1973 (ESA) as endangered (NMFS 2005b) with three factors believed to be related to their decline: food availability, contaminant loads, and vessel and noise interactions (Krahn et al. 2004).

As part of the ESA listing, NOAA's National Marine Fisheries Service (NMFS) is responsible for designating critical habitat for SRKWs. As shown in Figure 1, this includes the following areas but excludes spaces around U.S. military sites: 1) core summer area of U.S. waters up to the border of Canada and surrounding the San Juan Islands, 2) Puget Sound excluding Hood Canal, and 3) U.S. waters within the Strait of Juan de Fuca (NMFS 2006). The following two sections will review information on the use of sound by killer whales in general and the acoustic scene of the areas defined as critical habitat for SRKWs in particular.

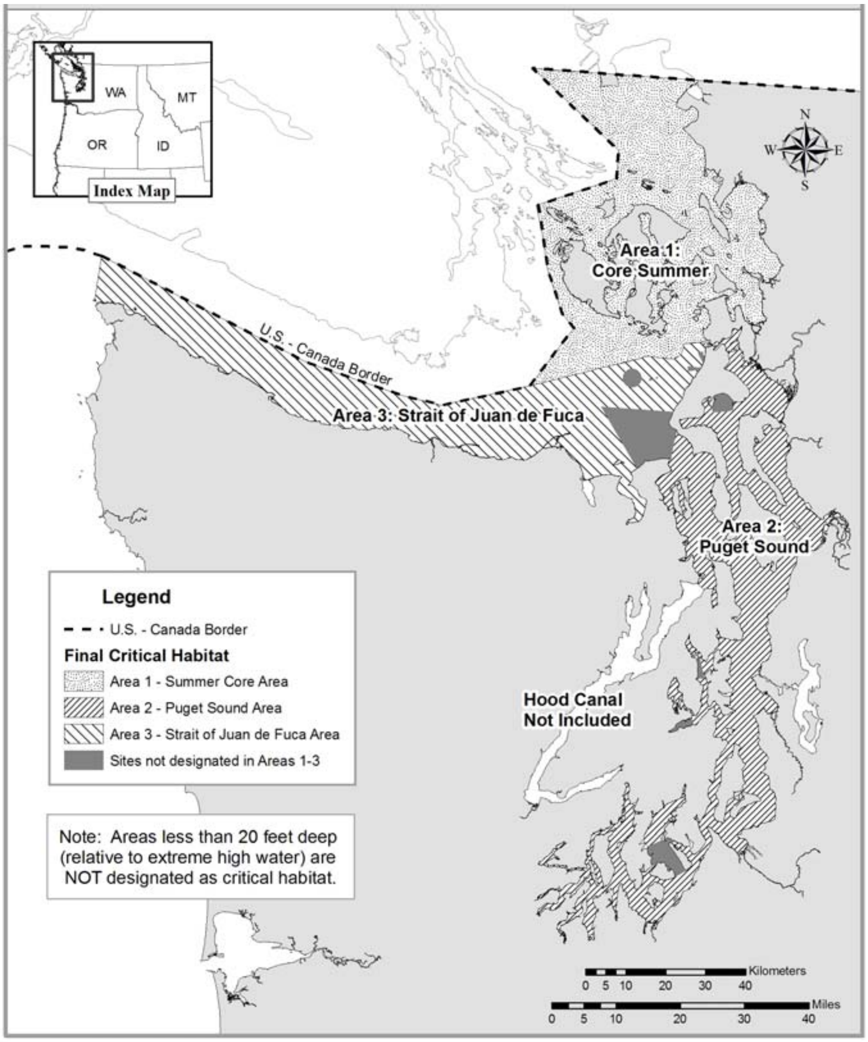

Figure 1. Critical habitat designation for Southern Resident killer whales (NMFS 2006).

Killer Whale Sound Production and Function

Killer whales produce a wide variety of clicks, whistles, and pulsed calls (Schevill and Watkins 1966, Ford 1989, Thomsen et al. 2001). Clicks are echolocation signals that are produced individually or in click trains. Individual clicks produced by Northern Resident killer whales are relatively broadband, short (0.1–25 milliseconds [ms]), and range in frequency from 8 to 80 kHz with an average center frequency of 50 kHz and an average bandwidth of 40 kHz (Au et al. 2004). These broadband biosonar signals are predominantly used for sensing objects such as prey in the environment and are produced by whales foraging on salmon at peak-to-peak source levels ranging from 195 to 225 dB re 1 µPa at 1 m (Au et al. 2004). There may be a considerable amount of variation in source level and frequency content of killer whale biosonar emissions depending on prey type. For example, Norwegian killer whales feeding on herring produced echolocation clicks that were lower in mean center frequency and source level, although there was some overlap in the range of these biosonar parameters (Simon et al. 2007). SRKWs also feed on salmon and it is likely that their biosonar emissions are similar to those of Northern Residents.

Whistles are tonal, nonpulsed signals that are relatively longer in duration (0.06–18 seconds [s]) and lower in frequency (0.5–10.2 kHz, Thomsen et al. 2001). Whistles are most often heard during close-range social activities but not as often during foraging and traveling (Thomsen et al. 2002). Whistles produced by Northern Residents typically have source levels ranging from 133 to 147 dB re 1 µPa at 1 m (Miller 2006).

Pulsed calls are the most commonly observed type of sounds emitted by killer whales and are categorized as discrete (stereotyped), variable, or aberrant (Ford 1989). These calls produced by both Northern and Southern residents are relatively long (600–2,000 ms), appear harmonically rich, and range in frequency between 1 and 10 kHz; but, those with high frequency components may contain harmonics up to 30 kHz (Ford 1989, Miller 2002). Variable calls are produced at source levels ranging from 133 to 165 dB while stereotyped calls are produced at source levels ranging from 135 to 168 dB re 1 µPa at 1 m (Miller 2006). These calls are most often used when killer whales are foraging and traveling and likely function to maintain social cohesion among pod members in the absence of other sensory information (Ford 1989, Miller 2002).

Killer whale communities have different discrete call repertoires, both among and between ecotypes (Ford 1991, Ford and Ellis 1999). Discrete calls are highly stereotyped and repetitive, and have pod-specific qualities that are stable over time (Ford 1989, Foote 2005). For example, each of the three pods among the Southern Resident population uses one or two discrete call types more than 50% of the time (Ford 1989, Foote 2005).

Given the biological significance of sound for biosonar and social functions in killer whales, it is important to address how ambient noise, from a variety of sources acting alone or in combination with other threats, might impact the population structure of SRKWs. In order to assess noise impacts on this population, it is necessary to consider several parameters of sound that are pertinent to the auditory capabilities of these whales. Such factors include amplitude, duration, and spectral characteristics of the source, as well as how it propagates in the underwater environment.

Characterizing Sound and Propagation

Ambient noise is essentially the background din. Several sources of sound contribute to ambient noise levels in the ocean and coastal marine environments (for review, see Richardson et al. 1995). Natural sounds include those produced from abiotic and biotic sources. Abiotic sounds arise from activities related to weather (i.e., wind, waves, and rain), seismic activity, underwater slides, and currents. Some of these sources can substantially increase ambient noise levels. For example, heavy precipitation can raise levels from a few to 20 dB re $1\mu Pa^2/Hz$ or more between 1 and 20 kHz (Wenz 1962, Nystuen et al. 1993). Biotic sources of noise include those produced by marine mammals, fish, and snapping shrimp (*Alpheus* and *Synalpheus* spp.). For example, ambient noise in some areas is dominated by snapping shrimp in frequencies that overlap with those of echolocating dolphins (Au et al. 1985).

Anthropogenic sounds that contribute to overall ambient noise include active sources (e.g., air guns, sonars) and those that are by-products of various human activities in the ocean. The latter include oil drilling, construction, and vessel traffic. Active sources include those used for military tactics, seismic surveys, fisheries, and oceanographic research. In the areas proposed as SRKW critical habitat, almost all of these sources of underwater sound are present.

Sound Propagation Variables

Sound propagated in sea water is subject to a number of variables that degrade the signal's amplitude over distance and thus affect potential received levels. The transmission loss of sound occurs due to two primary factors: spreading and attenuation. If a source is allowed to spread in all directions equally, then a spherical spreading law may be used to describe transmission loss. This might be an appropriate assumption if a source emanates in deep water in the open ocean. However, even oceanographic features of the water column can create less than spherical spreading conditions. This is because the speed of sound varies based on temperature, salinity, and pressure (depth).

Sound will also be attenuated as it travels over a distance due to absorption of sound by sea water and scattering due to particles. Sound is also reflected at the sea surface and from the bottom, creating instances of constructive or destructive interference. The details of spreading and attenuation of underwater sound are reviewed in Urick (1983). In shallow water environments such as those inhabited by SRKWs, the parameters that affect sound propagation can vary considerably in space and time. The acoustic environment in shallow water, therefore, is often described as complex.

Ambient Noise

Several investigations have reported ambient noise measurements in the areas designated as SRKW critical habitat and adjacent Canadian waters known to be important SRKW habitat.

Many of these studies have focused on describing sound levels generated by vessel propulsion. Noise from vessel propulsion is usually generated by propeller cavitation (producing broadband noise), propeller "singing" (producing tonal sounds and harmonics related to the propeller blade rate), and auxiliary machinery (such as pumps and rotating shafts) with the amount of noise related to vessel size, speed, and mode of operation (Richardson et al. 1995). Data from the following peer-reviewed articles and government reports[*] are examined in this section: Veirs and Veirs (2005), Nystuen (2006), Erbe (2002), Hildebrand et al. (2006), Laughlin (2005), and Jones and Wolfson (2006).

Veirs and Veirs (2005) report average sound pressure levels (SPL dB re 1µPa measured from 0.1 to 15 kHz) and power spectra (converted to noise spectral density levels in dB re $1\mu Pa^2/Hz$) of ambient sound recorded on the west side of San Juan Island in Haro Strait (lat 48°33′25″ N, long 123°10′23″ W). All measurements were reported as received levels between 0.1 and 15 kHz, thus propagation loss was not determined from source to receiver. The frequency range of these measurements is pertinent for the lower frequency range of killer whale hearing relevant to the reception of communication calls (see the Auditory Capabilities section below). Measurements were made on hydrophones that were cross-calibrated with a calibrated hydrophone and projector rented from the Naval Undersea Warfare Center in Newport, Rhode Island.

Veirs and Veirs (2005) reported average SPLs over hourly, daily, and monthly time periods from April 2004 to November 2005. Half-hourly averaged SPLs ranged from 95 to 130 dB with an overall half-hour SPL average of 115 dB. Broadband ambient sound levels were highly influenced by large vessels, such as commercial ships, that increased SPLs between 20 and 25 dB over a 10–30 minute period and to a lesser extent by smaller vessels, such as motor boats, that increased SPLs by 15–20 dB (Veirs and Veirs 2005). During the summer (July and August), these smaller vessels contributed more to the overall ambient levels during the day, raising hourly SPL averages by 2–4 dB compared to nonsummer daytime hours. As a consequence, there was a more pronounced diurnal pattern in the summer but it only amounted to about a 2 dB difference in 12-hour averaged SPLs.

Nighttime SPLs did not exhibit much seasonal variation, since larger ships operate during all months of the year. Monthly averaged SPLs ranged from 114.5 to 117.5 dB and were generally lowest from November through April and highest from June through August. Cumulative distributions of SPLs were also provided from archived measurements taken as 2-second averages (Figure 2). These distributions illustrate that 2-second SPL averages are greater than or equal to 120 dB 50% of the time during summer days, 30% of the time during summer nights, and 20% of the time during the winter (Veirs and Veirs 2005).

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[*] Government research reports are based on work contracted through the Northwest Fisheries Science Center Marine Mammal Program and go through a limited peer review process. These reports are online at http://www.nwfsc.noaa.gov/research/divisions/cbd/marine_mammal/research.cfm or can be requested by contacting the report author(s).

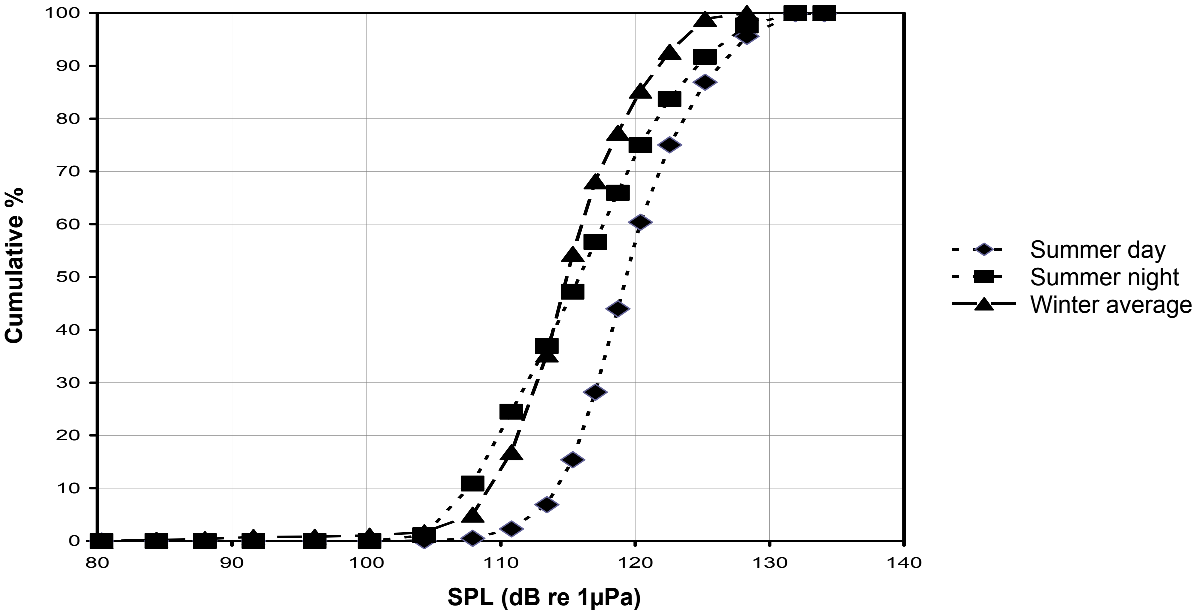

Figure 2. Cumulative distributions of 2-second average SPLs (0.1–15 kHz). Summer day includes SPL measurements during the hours of 0800–2000 and summer night includes SPL measurements from the other 12 of the 24 hour cycle (based on Figure 6 of Veirs and Veirs 2005).

Ambient noise levels were further described by frequency and showed similar trends for ambient levels recorded underwater elsewhere (Wenz 1962). For both July and November 2005, the highest spectrum levels occurred for the lowest frequencies, with peaks of about 82 dB re 1μPa^2/Hz occurring between 400 and 500 Hz, then generally decreased at a rate of 5 dB for each doubling of frequency (i.e., per octave) from 500 Hz to 5 kHz and at a rate of 6 dB for each doubling of frequency from 5 to 20 kHz (Figure 3). Between 1 and 20 kHz, there was a more pronounced difference in spectral levels (up to 5 dB re 1μPa^2/Hz) between daytime and nighttime hours in July compared to November. Below 1 kHz, very little seasonal variation in spectral levels was observed (Veirs and Veirs 2005).

Ambient sounds were also recorded from acoustic moorings called passive aquatic listeners (PALs) during the late spring and summer of 2005 from two areas off Cape Flattery, Washington (Nystuen 2006). Mean sound spectra from these moorings showed that close ships dominated the sound field below 10 kHz while rain and drizzle were the dominant sound sources above 20 kHz during a week in April 2005. As the dominant sound source, relative spectral density levels of close ships decreased with increasing frequency between 1 and 10 kHz while sound from rain had a slighter decrease with increasing frequency for spectral density levels between 20 and 50 kHz (Nystuen 2006). Furthermore at this location, shipping noise dominated the sound field approximately 10% to 30% of the time but was dependent on weather; that is, when the weather was poor, less shipping noise was present. From April to July the shipping activity increased while rain activity, which dominated the sound field from 1 to 10% of the time, decreased (Nystuen 2006).

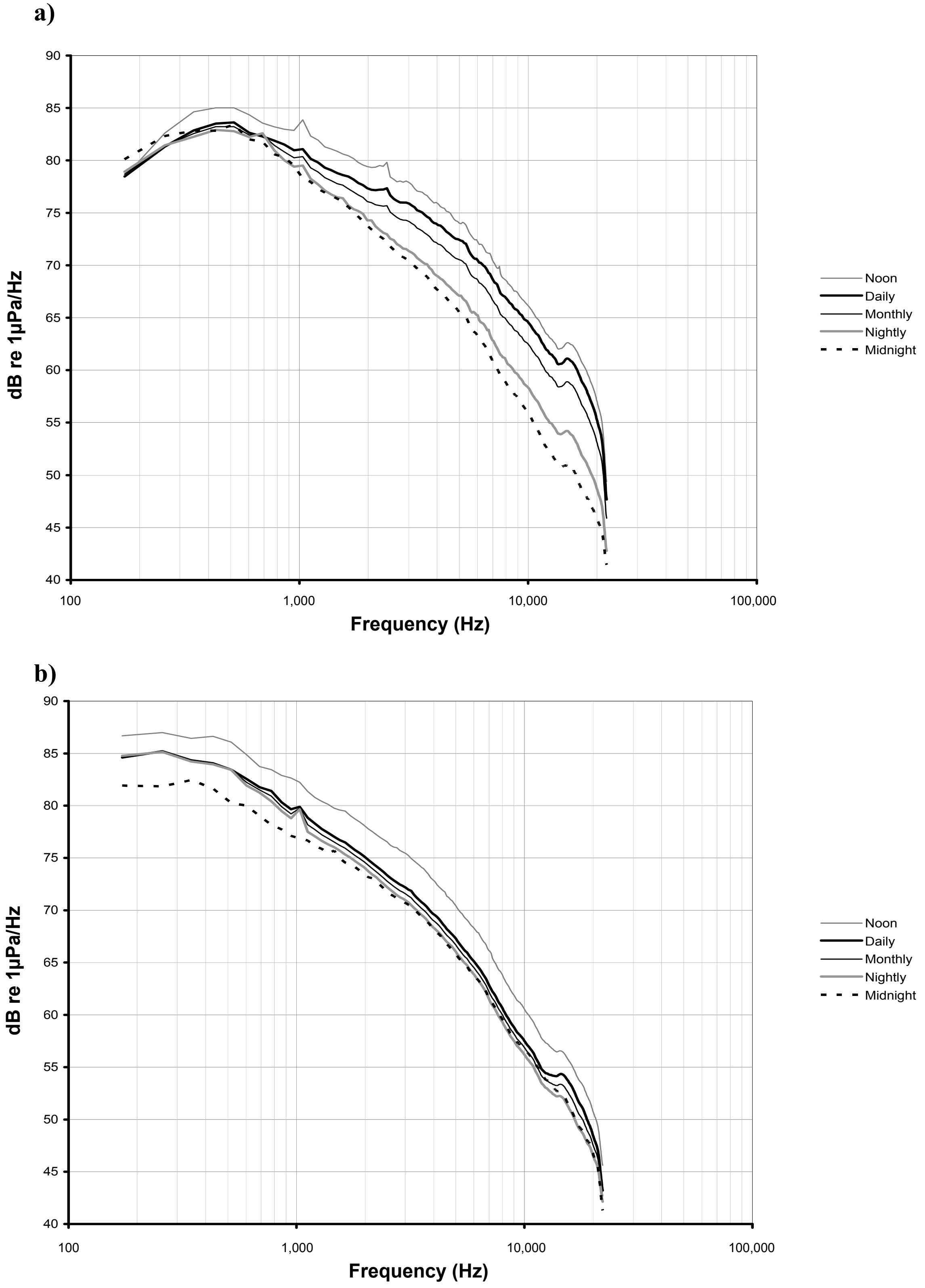

Figure 3. Spectrum levels in Haro Strait in a) July 2005 and b) November 2005 (based on Figure 7 and Figure 8 in Veirs and Veirs 2005).

Whale Watching Sound Propagation

Erbe (2002) reported sound level measurements in the presence and absence of whale watching vessels in Brotchi Ledge outside of Victoria Harbor (Canada) and in Haro Strait on the west side of San Juan Island (United States). Sound levels of whale watching inflatable and noninflatable motorboats were based on recordings made between 0.1 and 20 kHz and lasting 10–15 seconds. Vessel source levels were estimated using propagation models based on ray theory which included variables based on sound speed profiles, absorption loss by the sediment, and frequency-dependent absorption by ocean water (Erbe 2002).

In general, inflatables with larger engines had higher source levels, ones with stern drives were considerably quieter than those with outboards, and inflatables were slightly louder than motorboats with inboards or stern drives. The relationship between source level and vessel speed was nonmonotonic and attributed to vessel speed measurement error of the radar gun (Erbe 2002). At speeds of approximately 50 km/hour, average broadband source levels measured between 0.1 and 20 kHz were 162 and 159 dB re 1 μPa at 1 m for inflatables and motorboats, respectively (Erbe 2002). Noise levels were higher by about 15–27 dB re 1 μPa (based on one-twelfth octave band level analysis) in the presence of about five vessels (mix of inflatables and motorboats operating within 400 m of the hydrophone) compared to those measured in their absence (at a sea state of one-half, Erbe 2002).

Hildebrand et al. (2006) also reported source level measurements (as spectra in dB re 1 $\mu Pa^2/Hz$ at 1 m) for a variety of vessels of different sizes, propulsion systems, and operational speeds in Haro Strait. Measurements include those made from a calibrated broadband recording system (1–75 kHz). The extended frequency range of the vessel noise is pertinent for the higher frequency range of killer whale hearing relevant to the reception of echolocation signals (see the Auditory Capabilities section below). Received levels were measured at ranges that varied between 125 and 442 m for idle, normal cruise speed (17–31 knots), and power acceleration to full speed. Source level spectra for 10-second averages were calculated at 1 m assuming spherical spreading loss and using absorption coefficients from the equation of Ainslie and McColm (1998).

For power up conditions, the 10-second interval was chosen to include the highest sound pressure level. Noise spectra were not sufficiently above background levels to make accurate measurements of vessel noise levels under controlled conditions while vessels were idle despite efforts to collect data under conditions where no other vessels in the local vicinity were present. Thus data collected under idle conditions were not considered to be accurate measurements of real vessel noise (Hildebrand et al. 2006).

Noise source spectra from four whale watching vessels reported by Hildebrand et al. (2006) are summarized here. Boat A is a 28-foot fiberglass monohull with twin 90-horsepower, four-stroke outboard motors. Only the power up measurements produced spectral levels that were sufficiently above ambient noise levels. Under these traveling conditions, source spectral levels were approximately 118 dB re 1 $\mu Pa^2/Hz$ at 1 m between 4 and 15 kHz, then generally decreased at a rate of approximately 5 dB per octave up to 50 kHz. Another small whale watching vessel, Boat B with a 29-foot aluminum monohull and twin 225-horsepower outboard motors, produced source levels at a cruise speed of 24 knots that were significantly above ambient conditions between 100 and 300 Hz and between 6 and 75 kHz. Another medium-sized whale watching vessel, Boat C, is a 38-foot aluminum catamaran with jet drives. While cruising at a speed of 31 knots, Boat C produced source spectral levels that were generally lower than the smaller vessel with no measurable rotating equipment noise at the lower frequencies. The difference between idle and power acceleration noise was less than 10 dB.

Source levels for a larger whale watching vessel, Boat D, a 50-foot monohull vessel with three inboard/outboard (stern) drives, produced the highest source spectral levels that generally ranged between 110 and 145 dB re 1 $\mu Pa^2/Hz$ at 1 m between 60 and 1,000 Hz, with higher peaks at some frequencies in this range while idling. At the same speed, this vessel produced source levels that ranged between 115 and 110 dB re 1 $\mu Pa^2/Hz$ at 1 m between 1 and 5 kHz, then generally decreased at a rate of 5 dB/octave between 5 and 75 kHz. When Boat D was operating at a cruise speed of 23 knots, source spectral levels generally ranged between 125 and 145 dB re 1 $\mu Pa^2/Hz$ at 1 m between 60 and 1,000 Hz, with higher peaks at some frequencies in this range as well. At the same speed, this vessel produced source spectral levels of about 125 dB re 1 $\mu Pa^2/Hz$ at 1 m between 1 and 20 kHz, then generally decreased at a rate of approximately 10 dB/octave up to 75 kHz.

When Boat D was powering to cruise, source spectral levels generally ranged between 122 and 145 dB re 1 $\mu Pa^2/Hz$ at 1 m between 60 and 1,000 Hz, with the highest peak of 148 dB re 1 $\mu Pa^2/Hz$ at 1 m between 100 and 200 Hz. At the same speed, Boat D produced source spectral levels that ranged between 120 and 128 dB re 1 $\mu Pa^2/Hz$ at 1 m between 1 and 20 kHz, then generally decreased at a rate of approximately 10 dB/octave up to 60 kHz (Hildebrand et al. 2006).

Hildebrand et al. (2006) also opportunistically measured the source spectral levels of a 290 m long Korean container ship, the MV *Hanjin Marseilles*. Source spectral levels ranged from 135 to 165 dB re 1 $\mu Pa^2/Hz$ at 1 m between 60 and 1,000 Hz when measured from the low frequency hydrophone and showed a more or less linear decrease in level of approximately 10 dB/octave across this frequency range. At 442 m from the recording equipment, the received spectrum levels of the container ship ranged between 42 and 110 dB re 1 $\mu Pa^2/Hz$ between 0.06 and 75 kHz. Even at the highest frequencies, received levels were approximately 20 dB above ambient levels (that were measured on a different day) as shown in Figure 4 (Hildebrand et al. 2006).

A comparison of source spectral levels for all vessels operating at cruise speed is shown in Figure 5 (Hildebrand et al. 2006). Above 2 kHz, the largest whale watching vessel, Boat D, produced the highest source spectral levels, while Boat C produced the lowest source spectral levels. Boat C presumably produced the lowest noise levels at higher frequencies because of its jet drive system. The MV *Hanjin Marseilles* produced surprisingly significant levels of noise above 2 kHz that, with the exception of Boat D, were higher in level compared to other whale watching vessels.

From the results of Hildebrand et al. (2006) and Erbe (2002), noise generated by whale watching vessels as well as other vessel types is dependent on a combination of size, engine type, and operating speed.

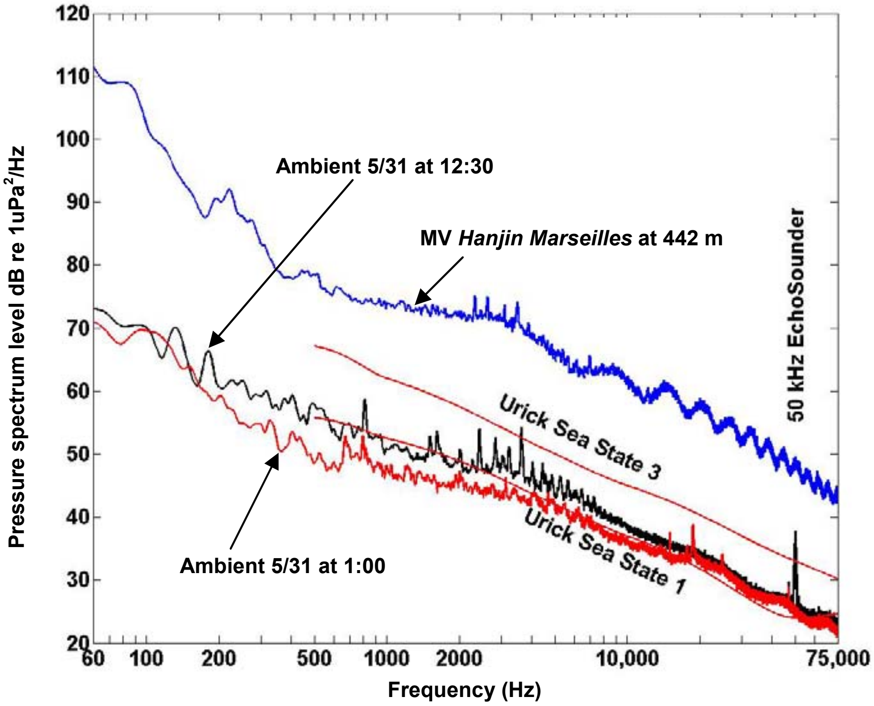

Figure 4. Received spectral levels of a container ship (MV *Hanjin Marseilles*) traveling at 21 knots as it passed 442 m from the recording equipment. Ambient conditions the day after the *Hanjin* passed in the Haro Strait as well as average sea states of 1 and 3 in deep water reported by Urick (1983) are also shown for comparison (adapted from Hildebrand et al. 2006).

Other Anthropogenic Sound Sources

While most studies thus far reviewed have focused on vessel sounds, other sources of anthropogenic sounds in SRKW habitat include those associated with marine construction such as pile driving and dredging. Sound exposure from pile driving has received particular attention given the loud impulsive (transient with rapid rise time) nature of such construction activity. As with other types of sounds, exposure from pile driving will depend on a number of factors including the size and type of pile driving equipment, the intensity of the driving activity, its duty cycle, and the local environment. Sound level measurements were made of pile driving activity associated with the restoration of the Friday Harbor ferry terminal. Average peak levels ranged between 180 and 215 dB re 1 µPa between 0 and 10 kHz (average RMS levels: 166–196 dB re 1 µPa, average SELs: 171–187 dB re 1 $\mu Pa^2 s$) that varied depending on the condition (Lauglin 2005).

In some cases, mitigation measures are taken to reduce the overall level of construction noise introduced into the environment. For example, air bubble curtains, in which a perforated air hose is placed outside the pile, generates bubbles as air is passed through the hose. Bubble curtain designs include single, dual, multiple ring, and tree curtains. In ideal cases, the bubble curtain effectively absorbs and reduces the magnitude of the pressure waves generated from

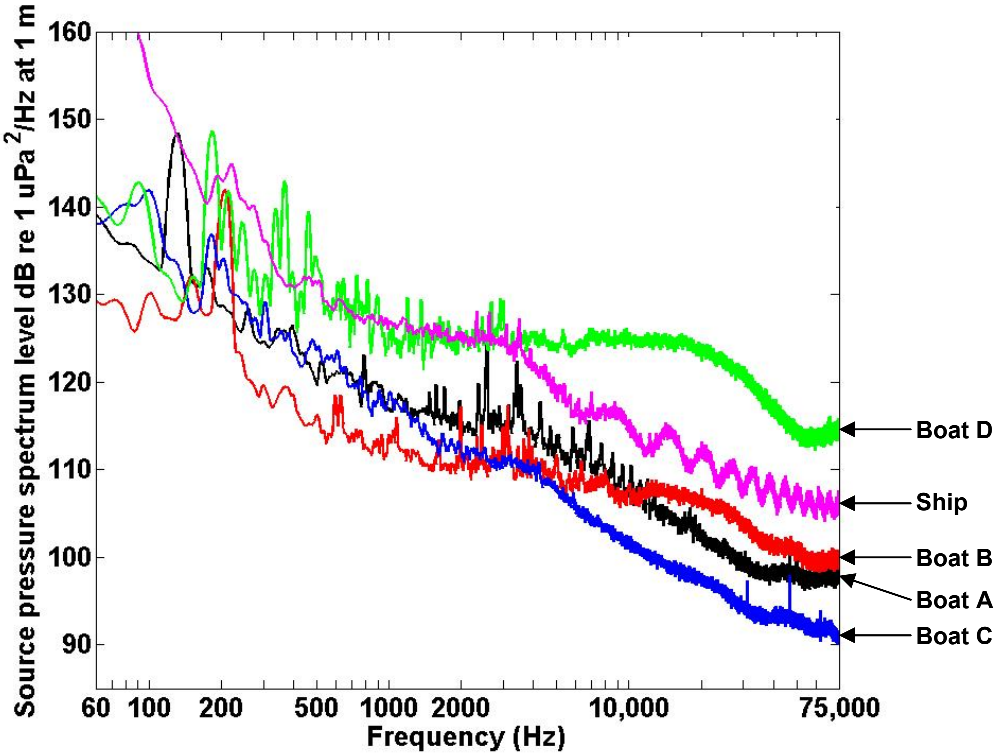

Figure 5. Spectrum source levels of all vessels at cruise speeds recorded on the broadband hydrophone system (adapted from Hildebrand et al. 2006).

driving the pile into the ground (Wursig et al. 2000). However, many factors can influence the effectiveness of the bubble curtain such as water currents, bathymetry, and tide levels. The average reduction in sound levels from multiple ring bubble curtain deployment during pile driving at the Friday Harbor ferry terminal ranged between 1 and 3 dB and the maximum reduction was 16 dB (Laughlin 2005). The results depended on the individual piles driven, the extent of the bubble curtain, and external factors such as ambient noise, given that these tests were not conducted in controlled conditions (Laughlin 2005).

Modeling Sound Propagation

Because many areas frequented by SRKWs are complex shallow water acoustic environments, it is necessary to consider how transmission loss from source to receiver will affect the sound levels potentially reaching killer whales in the area. Jones and Wolfson (2005) modeled propagation of sound emanating from large vessels in the main shipping lanes in the Haro Strait, focusing on numerical estimates in the open channel. Predicted values of received sound levels were compared to field measurements made by PALs collected in spring and summer 2004 (Nystuen 2006). Information on the position, course, and speed of large vessels (≥ 20 m in length) was obtained from the Vessel Traffic Operations Support System (VTOSS) of the Canadian Coast Guard Services. Information for 27 May–30 June 2004 from VTOSS illustrated that 24% of the vessels larger than 20 m were bulk carriers, 15% were tugs, 13% were container ships, 10% were ferries, 9% were fishing vessels, and a smaller percentage were a variety of other vessels types.

Correlation of specific acoustic recordings with data from the VTOSS database provided information about the sound propagation of ships based on type, location, speed, and orientation. For example, when a large cargo ship passed to the side of the PAL mooring at a range of 1 km, received levels were approximately 85 and 75 dB (re 1µPa) at 3.6 and 10.4 kHz, respectively. In contrast, when the same ship passed to the side of the PAL mooring at a range of 4–5 km, the spectral levels of the same frequencies were about 20 dB lower.

In Jones and Wolfson (2006), modeling and analysis were focused at the frequency of 3.6 kHz to represent the frequency ranges between 1 and 10 kHz. Acoustic propagation modeling was described using two-dimensional parabolic equation numerical methods in which only reverberation in the forward direction was considered while backscattering was neglected (Jones and Wolfson 2006). This model is especially vulnerable to uncertainties in bathymetry, sea surface roughness, bottom substrate, and variables affecting sound speed profiles.

Characteristics of these parameters along with appropriate assumptions were incorporated in the model for analysis. Sound speed profiles obtained from conductivity, temperature, and depth measurements collected in May and June in 1990 to 2002 illustrated relatively little variation in sound speed with depth and locations in the Haro Strait (Jones and Wolfson 2006). Model simulations were performed for a single ship at 10 ship positions, each position separated by 1 km as it passed by the stationary mooring. The model was run with three different categories of sediment type (smoother sand/mud sediment, rough sand/mud sediment, rough rock/sand sediment) and three levels of wind speed/sea surface roughness (0, 5, and 10 m/s).

In all cases, the model results were consistent with the measured results when an estimated ship source level of 175 dB at 3.6 kHz was used. However, the results for the different cases of sediment and sea surface conditions varied within 3 dB of each other, implying these parameters do not strongly influence model results (Jones and Wolfson 2005). Furthermore, variability of the model results did not increase with increased sea surface roughness. The investigators hypothesize that the short distance between ship and receiver results in bathymetric effects that dominate the results while bottom and surface conditions contribute little to propagation effects.

Model results were also obtained for the average positions of ships traveling in the northern shipping lanes of the Haro Strait using VTOSS database information. The closest distance to the mooring in this situation was approximately 2 km while the farthest was approximately 7 km. Predicted received levels at the farthest point were lower by about 15 dB at 3.6 kHz compared to the closest point based on this model. Given the complexity of the acoustic environment of the Haro Strait, the report recommends that acoustic modeling is best used as a complement to field measurements (Jones and Wolfson 2006). Other recommendations include correlating seasonal and regional measurements of background sound levels with records of killer whale locations and direct measurements of broadband source levels of large ships in the main shipping lanes.

To summarize, most measurements of ambient sounds levels in SRKW habitat are strongly influenced by vessel traffic that can reach broadband received levels of up to 130 dB and source levels that average approximately 160 dB (Erbe 2002, Veirs and Veirs 2005). In addition various vessels, including those used for whale watching activities as well as larger ships, can raise noise levels significantly above ambient levels even at frequencies up to 75 kHz (Hildebrand et al. 2006). Furthermore, data model comparisons illustrate that propagation models may accurately predict levels of vessel noise potentially received by killer whales at a particular location. However, given the complex shallow water environment, models should be used only to complement and cannot substitute for field measurements.

Auditory Capabilities and Auditory Effects of Sound Exposure

The reports summarized in the previous section focus on describing sounds in SRKW core summer habitats. In order to assess noise impacts, it is necessary also to consider several parameters of sound that are pertinent to the auditory capabilities of the whales. Such factors include hearing sensitivity of sound based on amplitude, duration and temporal factors, frequency, and how exposure of sound affects baseline sensitivity and other auditory functions. This section reviews what is known about the hearing capabilities and auditory effects of sound exposure in killer whales. Because of the limitations associated with time and cost of conducting laboratory research on marine mammals, there is a dearth of information about some aspects of auditory capabilities in killer whales, particularly in terms of auditory effects of noise. A number of studies have investigated such effects on smaller delphinids and appropriate information will also be reviewed to supplement current knowledge in this area.

Audiograms and Basic Auditory Function

There are several experimental approaches that have been used to determine the auditory capabilities of marine mammals including killer whales. A basic assessment of hearing is to determine sensitivity as a function of frequency in order to plot an audiogram. One of the most direct approaches to measure this involves behaviorally training subjects to respond in a particular way when they hear a particular sound and plot performance as a function of the sound's amplitude (behavioral psychophysics). Another approach involves electrophysiological measurements of hearing sensitivity, which requires measuring auditory brainstem responses (ABRs) as a function of the sound's amplitude. One advantage of the latter approach is that it typically takes less time to measure a threshold, which can be desirable in some instances, for example, when hearing recovery functions are measured after noise exposure.

Both of these approaches have experimental limitations that should be noted. For example, not all listening environments are the same and ambient noise levels of the testing area should be measured and noted to avoid "noise-limited" measurements of hearing sensitivity. Additionally, the temporal summation (or temporal integration) properties of the auditory system influence sensitivity to a sound. Sounds shorter than some critical value or time constant are generally less detectable than longer signals; that is, sensitivity typically decreases nonlinearly for signals shorter than the time constant. Time constants for tonal stimuli in mammals typically vary between 30 and 800 ms and appear to be relatively consistent between several taxa (Fay 1988).

Although time constants have not been determined for killer whales, Johnson (1968) measured the dependence of thresholds on signal duration in a bottlenose dolphin (*Tursiops truncatus*) and reported time constants that ranged from 220 ms at 4 kHz to 30 ms at 45 kHz.

Johnson (1991) also measured time constants in a beluga whale (*Delphinapterus leucas*) for a 60 Hz single tone pulse and reported a time constant of 20 ms. The lower frequency results are consistent with those measured in humans (Plomp and Bouman 1959). Assuming that the temporal summation properties of killer whales follows the typical delphinid pattern, thresholds measured with signals shorter than these time constants will not be directly comparable to those measured with longer signals. For example, thresholds measured by electrophysiological methods are orders of magnitude shorter (and often below these time constants) than those measured by behavioral methods.

Hearing Sensitivity

In a behavioral psychophysical setting, hearing sensitivity is usually defined as a threshold. The concept of a threshold is a statistical probability of an animal detecting the sound at a particular level some percentage of the time (often 50 or 75% correct depending on the procedure). The physical parameters of the stimulus are not the only variables that affect threshold measurements of a subject. Nonsensory cues such as reinforcement contingencies and expectancies can significantly affect the measured threshold. For example, in most sound detection tasks, the signal level is reduced beyond some detection point and signal-present and signal-absent trials are both often used to keep the animal "honest" in reporting the presence of a signal.

If an animal experiences punishment for responding to a signal when no signal is present, then the animal might become more conservative in reporting that it heard the signal. That is, the individual might adopt a tendency to report "no" when uncertain. This would lead to a higher measured hearing threshold or perhaps an underestimate of hearing sensitivity. For these reasons, it is necessary to account for animal response biases during sensory tasks (for further discussion see Schusterman 1974, Renouf 1991). This type of response behavior is not only important to consider in terms of how animals perform in laboratory settings but also how responses of free-ranging animals might be affected by different motivational states. For example, if there is disturbance in a unique feeding habitat where food is seasonally limited, animals may tolerate exposure because they are motivated to feed and may not have suitable alternatives.

Another consideration of lab-based results is that very few individuals are typically tested. As with any population, auditory capabilities are expected to show individual variation based on factors including genetics, age, sex, and exposure to pathogens, ototoxic drugs, and noise. For example, in humans and presumably other mammals, high-frequency hearing loss or presbycusis is common with age (Yost 2000). Thus it is expected that a larger number of older animals would be less sensitive to higher frequencies than younger animals.

Two laboratory studies on the hearing sensitivity of captive killer whales have been conducted (Hall and Johnson 1972, Szymanski et al. 1999). One subadult male was tested in the earlier study and an audiogram was obtained for 8-second pure tones ranging in frequency between 500 Hz and 31 kHz. Greatest sensitivity or a lowest threshold of 30 dB re 1 μPa for this individual occurred at 15 kHz, the observed upper limit of hearing sensitivity was 32 kHz, thresholds below 10 kHz were probably noise limited, and this individual was suspected to have high-frequency hearing loss (Hall and Johnson 1972). Both behavioral and electrophysiological thresholds were obtained in two adult females in a later study (Szymanski et al. 1999). These individuals had much higher and more plausible upper frequency hearing limits of about 120 kHz compared to other delphinids than the individual in the Hall and Johnson (1972) study.

Both behavioral and electrophysiological audiograms appeared similar but behavioral thresholds were typical lower (Figure 6). However, electrophysiological thresholds were measured using much shorter signals (1 or 0.5 ms durations) than those for behavioral thresholds (2 s). Furthermore, the investigators reported electrophysiological thresholds based on dB peak-to-peak measurements and behavioral thresholds based on dB RMS measurements. If comparable dB measurements were computed, then the average differences of thresholds between the two methods would be reduced. The frequency of best sensitivity averaged for both subjects was 20 kHz and range of best sensitivity (±10 dB from lowest threshold) was 18–42 kHz (Szymanski et al. 1999).

One caveat of this study was that behavioral thresholds might be underestimates of hearing sensitivity because the subjects were reinforced after missing a signal if they waited for a recall tone and false alarm rates were reportedly so low that it is possible that the animals were trained to conservatively report the signal. Despite these experimental caveats, the audiogram based on Szymanski et al. (1999) is the best available data to determine hearing sensitivity of killer whales in quiet conditions.

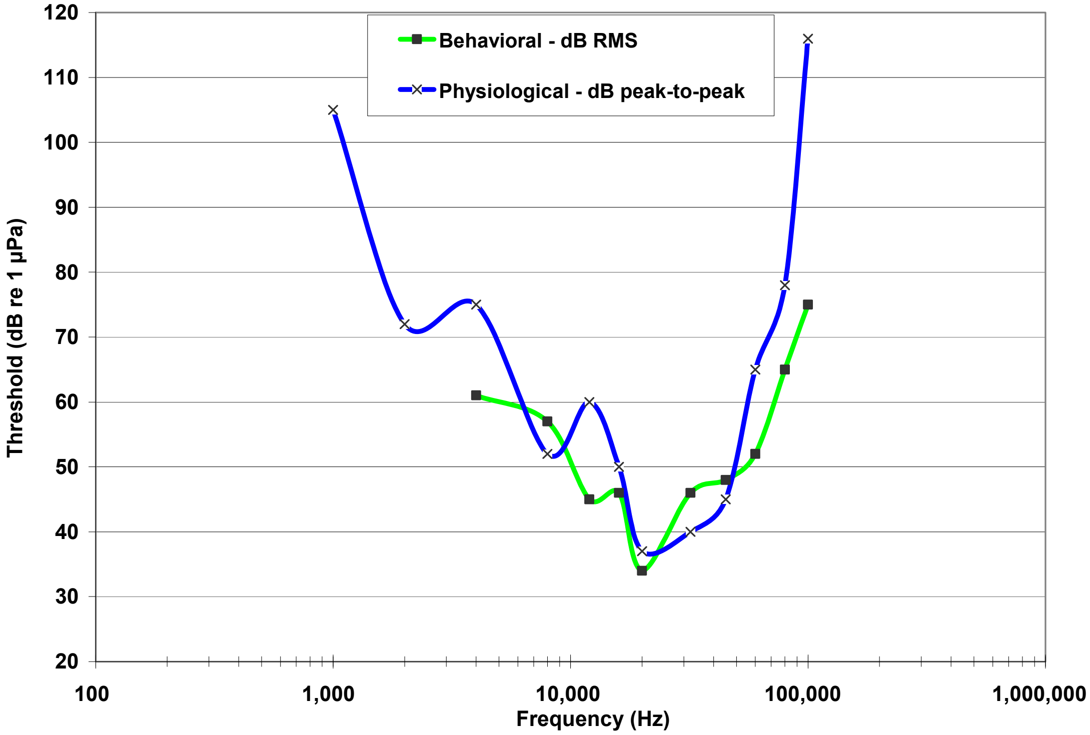

Figure 6. Behavioral and physiological (ABR) audiograms based on averaged thresholds for two female killer whales as reported by Szymanski et al. 1999. (Reprinted from Figure 5c in Szymanski et al., copyright 1999, with permission from Acoustical Society of America.)

Other Capabilities

Other auditory capabilities that are pertinent to sound exposure impacts include directional hearing and sound localization, frequency and intensity discrimination, and loudness perception. For example, animals must be able to localize the source in order to avoid a noxious sound. Unfortunately, there are no studies that have measured these auditory capabilities in killer whales, but results from studies on other delphinids are available to supplement current knowledge. Bottlenose dolphins have very good localization abilities; they can resolve angular separation of clicks within a fraction of a degree and pure tones between 6 and 100 kHz with a few degrees (≤ 5º) in both the horizontal and vertical planes (Renaud and Popper 1975). This ability is most likely related to use of echolocation for sensing objects in the underwater environment.

Frequency discrimination involves the ability to perceive two tones as being separate in frequency or a pure tone being of constant frequency versus frequency modulated. This is usually measured via the difference limen (DL), which is typically defined as the difference between a reference sound frequency and that of the sound frequency that is just perceived as different. Because this discrimination ability is dependent on the reference value, discrimination is proportional to the standard and defined as the just noticeable difference or relative DL as a percentage of the standard (Richardson et al. 1995, Yost 2000). Bottlenose dolphins seem to outperform most mammals at frequency discrimination. Behavioral tests indicate that relative DLs are 0.21–0.81% from 2 to 130 kHz (Thompson and Herman 1975). Intensity discrimination is also very good in bottlenose dolphins, who can detect level differences of 0.35 to 2 dB for pure tones and clicks (reviewed in Richardson et al. 1995).

For the most part, audiograms provide information about hearing sensitivity in quiet conditions; however, loudness perception is another important consideration of hearing and acoustic exposure impacts, particularly for predicting behavioral reactions to sounds and onset of auditory injury in mammals. The perception of loudness in humans is usually conducted through loudness-matching experiments. In these tasks, subjects are asked to match the loudness of a tone at one frequency to that of another frequency, then the results are plotted as level in dB of the comparison tone in equal loudness level contours. In people, equal loudness contours parallel the audiogram for SPLs but flatten out as SPL increases (Figure 7). Thus loudness perception can be approximated but not fully predicted by an audiogram.

In determining noise exposure for people, idealized equal loudness curves have been used to calculate weighting functions, which emphasize some frequencies (at best sensitivity) and deemphasize other frequencies. Essentially, the weighting functions are used to filter sound for calculating exposure risk because weighting functions improve dose-response functions in humans (e.g., correlation between noise exposure and annoyance response, Leatherwood et al. 2002). No equal loudness curves are currently available for delphinids or other marine mammals, so audiograms have been used to develop interim weighting function despite the fact that the effectiveness of this approach has not been tested (Miller et al. 2005, Southall et al. 2007).

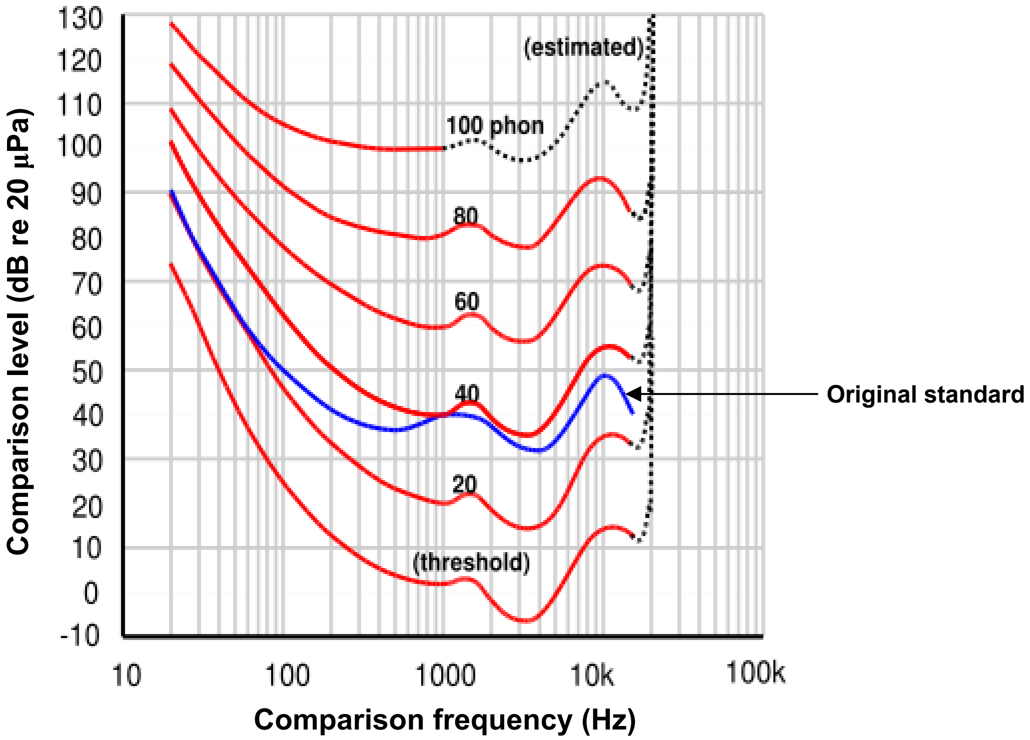

Figure 7. Equal loudness contours of human subjects depicting the level of a comparison tone required to match the perceived loudness of a 1,000 Hz tone presented at different levels (threshold, 20, 40, 60, 80, and 100 phons or dB SPL). Each of the above curves represents an equal loudness contour based on ISO Standard 226: 2003 revision. The original 40 phon standard is labeled (source: http://en.wikipedia.org/wiki/Equal-loudness_contour).

Auditory Masking, Critical Ratios, and Critical Bandwidths

While audiograms provide information about hearing sensitivity in quiet conditions, free-ranging animals usually need to detect biologically important sounds in noisy environments. There are two types of auditory effects of noise exposure that will be reviewed here. One effect is a simultaneous or masking effect while the other effect is a residual or hearing loss effect that can be temporary or permanent (Yost 2000). In most cases, background noise must sufficiently overlap in frequency, level, duration, and direction of the target signal for these auditory effects to be fully realized.

Any sound that reduces the audibility of another sound of interest has the potential to mask that sound. In practice, a target sound needs to be a certain level above the ambient noise in order for the target to be detected. This concept has been defined as the critical ratio (CR), or the difference in dB of the background ambient (masking) noise spectrum level and the amount by which a signal must exceed the background level in order to be audible. For example, a CR of 15 dB means that a signal must be 15 dB above the spectrum level of the noise in order for it to be detected. CRs across a number of mammal groups tend to be independent of the masker spectrum level and tend to increase with increasing frequency (Fay 1988). A few studies have measured CRs in the bottlenose dolphin, false killer whale (*Pseudorca crassidens*), and beluga whale (reviewed in Richardson et al. 1995). The results from these studies indicated that delphinid subjects have CRs ranging from 17 to 20 dB below 1 kHz to about 40 dB at approximately 100 kHz. Killer whales have CRs that ranged from about 20 dB at 10 kHz to 40 dB at 80 kHz (Bain et al. unpublished data, cited in Bain and Dahlheim 1994).

The amount of frequency overlap between a target signal and masker is probably one of the more important considerations of auditory masking in noisy environments. For example, pure tones are masked by sounds at frequencies near the tone frequency. Sounds at frequencies outside the bandwidth of the auditory filter typically do not affect audibility unless the noise level is very high. The limit of the frequency spread of the noise in its ability to mask a signal at a particular frequency is called the critical bandwidth. There are a number of different ways that critical bandwidths have been estimated. An indirect way, called the equal-power method, assumes that the signal power must be equal or exceed the total noise power in the masking band in order for it to be audible (Fletcher 1940). According to this assumption, the masking bandwidth (in Hz) is estimated as $10^{(CR/10)}$.

More direct ways to estimate critical bandwidths include measuring masked thresholds as a function of the bandwidth of the noise masker (band-narrowing technique) or using notched (band-reject) noise maskers and varying the bandwidth of the notch. These direct methods typically result in critical bandwidths that are usually but not necessarily wider than those estimated using the equal-power model for many mammals (Sharf 1970, Au and Moore, 1990, Southall et al. 2003a). It is now generally accepted that the indirect method using the equal-power model has limited accuracy in estimating critical bandwidths within a given species (see Southall et al. 2003b, Yost and Shofner 2005).

While there are no direct critical bandwidth measurements available for killer whales, a few studies have measured critical bandwidths in other delphinids using both the band-narrowing and notched-noise techniques. Using the band-narrowing technique, Au and Moore (1990) reported critical bandwidths of 17, 25, and 45 kHz for pure-tone frequencies of 30, 60, and 120 kHz, respectively, in a bottlenose dolphin. Using the notched noise technique, both Lemonds et al. (2000) and Finneran et al. (2002a) estimated much narrower critical bandwidths (reported as equivalent rectangular bandwidths). In the former study, critical bandwidths were 16% at 40 kHz and about 11% between 60 and 100 kHz in a bottlenose dolphin (Lemonds et al. 2000). In the latter study, critical bandwidths were 12 and 17% of the center frequency at 20 and 30 kHz, respectively, in two bottlenose dolphins and 9 and 15% at 20 and 30 kHz, respectively, in a beluga whale (Finneran et al. 2002a). These values roughly correspond to one-sixth of an octave wide and, like CRs, are frequency dependent with critical bandwidths tending to increase with increasing frequency (Yost 2000).

Spatial Overlap

Spatial overlap between a target sound and masker is another important consideration of auditory masking. The directional hearing system helps ameliorate the masking effect of noise, provided that the masker itself has a direction. That is, when a signal and masker are spatially separate, masked thresholds are lower or sensitivity is better compared to when signals and maskers are collocated. The auditory phenomenon, termed “spatial release from masking” or “spatial unmasking,” can significantly improve signal detection of both simple and complex sounds. For example, Bain and Dahlheim (1994) measured auditory thresholds for tones at 4, 8, and 20 kHz masked by low frequency noise with energy primarily between 500 Hz and 5 kHz in two captive killer whales. The signal always occurred directly in front of the subject (0, 0°) while the masker was positioned at 0 (collocated), 90, or 150° in the horizontal plane or 30° above or below the subject. Although it is unclear whether any sound energy from the noise masked the 8 and 20 kHz signal, masked thresholds were generally lower (ranging between 0 and 40 dB) when the signal and masker were spatially separate compared to when they were collocated. Masked thresholds were lowest (on average by −7 to −24 dB) when the signal and masker were separated by 90 and 150° relative to when the signal and masker were collocated. These effects were largest for the higher frequencies, but in most cases there were large interindividual differences (Bain and Dahlheim 1994).

In another investigation using a bottlenose dolphin as a subject, masked thresholds were measured with spatial separation of signal (at 30, 60, and 120 kHz) and masker (band pass of 30 and 150 kHz) in both the horizontal and vertical planes (Au and Moore 1984). As in the killer whale investigation, masked threshold differences of the dolphin were generally largest (thresholds were lowest) for the highest frequencies at both azimuth and elevation (Au and Moore 1984). CRs are generally measured with the signal and masker emanating from the same point in space. Thus masking effects in more real world situations would not be as substantial when signals and maskers have different spatial configurations relative to each other as data from critical ratio investigations suggest.

Temporal factors can also cause release from masking in some cases, particularly when two separate sources of noise (maskers) across different frequencies are amplitude modulated in a coherent way. When the signal overlaps in frequency with one of the two maskers, detection of the signal improves when the two maskers are amplitude modulated coherently compared to when they are modulated incoherently. Such an auditory phenomenon is called comodulation masking release. This effect may be related to the degree of amplitude modulation of the two maskers. For example, when only one masker is present which overlaps in frequency with the target signals, detection of the signal is easier for a masker that has a larger degree of amplitude modulation (and periods of lower energy), compared to when a masker has less amplitude modulation even when the overall sound energy level of the maskers are equivalent (Brumm and Slabberkoorn 2005). No marine mammal studies on comodulation masking release have been conducted, but the effect seems to occur across a wide range of taxa including humans and birds and is likely to occur in marine mammals as well.

While most investigations of auditory masking have measured thresholds of pure tones masked by white noise, very few studies have determined the effects of noise on the reception of natural sounds. Bain and Dahlheim (1994) reported thresholds of pure tones, killer whale clicks, and a call masked by recorded vessel noise from oil-spill clean up operations in Alaska's Prince William Sound after the 1989 *Exxon Valdez* oil spill. While masking of pure tones occurred for all frequencies below 20 kHz, very little masking of the call and clicks was observed; however, a 27 kHz pure tone was added to the call to facilitate performance transfer and it is unclear whether or not this improved detection (Bain and Dahlheim 1994). Given that the click and call plus high frequency tone were likely outside the bandwidth of the vessel noise, it is not surprising that these results were observed.

Hearing Loss Due to Sound Exposure

Another auditory effect of sound exposure is hearing loss. There are temporary and permanent forms of hearing loss. Temporary hearing loss or temporary threshold shift (TTS) involves recovery of baseline hearing over a period of time and occurs from physiological fatigue of the hair cells of the inner ear. The magnitude of the threshold shift depends on the amplitude, duration, temporal pattern, frequency, and energy content of fatiguing sound. For example, exposure to a more intense but shorter sound may result in similar TTS compared to that measured after exposure to a less intense but longer sound. With some types of sound exposure, subjects may experience large but fully recoverable shifts in threshold (≈40 dB) that become asymptotic with longer exposure durations (asymptotic TTS). The largest threshold shifts often occur at frequencies one-half to one octave above the frequency of exposure (Yost 2000).

Permanent hearing loss or permanent threshold shift (PTS) does not show recovery over time and is the manifestation of auditory injury. PTS results from damage or death of the hair cells in the inner ear. PTS can occur from repeated exposures that induce TTS or from a single intense exposure. The relationship between TTS and PTS is often quite complex despite intense research in humans and terrestrial models. For example, even very large threshold shifts (up to 40 dB) resulting from sound exposure can be fully recovered. For ethical reasons, PTS is not studied in marine mammals. TTS has been investigated to describe how temporary hearing loss is dependent on the type, frequency, amplitude, and duration of the exposure, and to model PTS from TTS data despite the complicated relationship between the two. This work is typically done in a laboratory setting in which trained animals are tested for baseline hearing and exposed to noise, then hearing is tested again to measure a threshold shift.

Both behavioral and electrophysiological methods have been used to measure TTS. While there are no TTS studies on killer whales, there are a handful of TTS investigations on bottlenose dolphins and beluga whales. These include measuring TTS after exposure to broadband noise (Nachtigall et al. 2003, 2004), tones (Schlundt et al. 2000, 2006, Finneran et al. 2005, 2007), and impulsive sounds (Finneran et al. 2000, 2002b). Since TTS is dependent on both amplitude and duration of exposure, sound exposure level (SEL in dB re 1 $\mu Pa^2 s$) is a common way to quantify exposure, because it describes both duration and amplitude in one metric. However, SEL inherently assumes that different combinations of amplitude and duration that yield equivalent SELs (i.e., equal energy) result in the same amount of TTS. Some data measured in delphinids suggest that this may be an appropriate assumption under some conditions (Finneran et al. 2005).

Using a behavioral procedure to measure threshold shifts of a 7.5 kHz tone in a bottlenose dolphin, an average TTS of 11 dB resulted after exposure to broadband noise having a flat spectrum between 4 and 11 kHz at 179 dB re 1 µPa for 41 to 54 minutes (Nachtigall et al. 2003). This corresponded to an SEL of 213.8 dB re 1 $\mu Pa^2 s$ when the average exposure duration of 50 minutes was used. Using an electrophysiological approach to measure threshold shifts at 8, 11.2, 16, 22.5, and 32 kHz in the same subject with the same fatiguing stimulus at 160 dB re 1 µPa for 30 minutes (SEL of 192.6 dB re 1 $\mu Pa^2 s$), the largest threshold shifts occurred after 5 minutes of exposure (TTS_5) at frequencies between 8 and 16 kHz. The maximum threshold shift of 8 dB occurred at 16 kHz with shifts equal to approximately 5 and 6 dB at 8 and 11.2 kHz, respectively. Recovery to baseline hearing levels was approximately 1.5 dB per doubling of time (Nachtigall et al. 2004).

TTS was also measured in bottlenose dolphins and belugas whales exposed to 1-second tones ranging between 3 and 75 kHz and at levels between 182 and 201 dB re 1 μPa. Masked temporary threshold shifts (MTTS) at frequencies equal to or higher than (by one-half or one octave) the exposure frequency ranged between 6 and 17 dB (Schlundt et al. 2000). Only threshold shifts greater than or equal to 6 dB were reported as noise induced TTS because this value seemed sufficiently above the 3–4 dB variability of baseline hearing thresholds. The beluga whale exhibited equivalent threshold shifts (between 6 and 8 dB) at higher exposure levels to that of the dolphin when the same exposure and hearing test frequencies was used between the subjects. Only masked thresholds could be measured since behavioral testing was conducted in San Diego Bay, where ambient noise levels were high and variable, so masking noise (between 3 and 100 kHz and varied in spectral levels between 63 and 95 dB re 1μPa^2/Hz) was used to create a consistent noise floor during preexposure, exposure, and postexposure conditions (Schlundt et al. 2000).

Because the presence of masking noise often results in smaller threshold shifts in terrestrial mammals, these results reported by Schlundt et al. (2000) were questioned and testing was replicated in a quiet pool again using a behavioral response procedure. In this study, two bottlenose dolphins were exposed to a 3 kHz tone at levels ranging between 100 and 200 dB re 1 μPa for 1–8 seconds, corresponding to SELs of 100 to 203 dB re 1 μPa^{2}s. In most cases, significant threshold shifts (different from control exposures) measured 4 minutes after exposure (TTS$_4$) were observed at 4.5 kHz for SELs above 190 dB re 1 μPa^{2}s. No significant TTS was observed at the same frequency as the exposure up to levels of 200 dB re 1 μPa^{2}s (Finneran et al. 2005). For SELs above 200 dB re 1 μPa^{2}s, recovery was not complete by 10 minutes, but hearing sensitivity typically returned to baseline values within a day. All data summarized for exposures to continuous noise or tones (nonimpulsive sounds) in bottlenose dolphins and beluga whales showed that, in most cases, significant threshold shifts occurred at SELs greater than or equal to 195 dB re 1 μPa^{2}s despite differences in exposure duration, sound pressure level, experimental approaches, and subjects (Finneran et al. 2005).

More recent work has extended exposure durations at the same frequency up to 128 seconds corresponding to SELs of up to 217 dB re 1 μPa^{2}s, and preliminary results indicated that TTS$_4$ of up to 23 dB occurred with full recovery within 30 minutes (Schlundt et al. 2006). Hearing sensitivity was measured both behaviorally and electrophysiologically in a bottlenose dolphin after exposure to a 20 kHz tone that ranged in SELs of 203–206 dB re 1 μPa^{2}s in a more recent study by Finneran et al. (2007). Threshold shifts up to 40–45 dB using the physiological approach were observed with frequency-dependent results. The largest shifts were observed at 30 kHz, followed by 40 and 20 kHz with no measurable TTS occurring at 10, 50, 60, and 70 kHz. Threshold shifts were smaller (19–33 dB) and recovery time was faster when the behavioral method was used even when the exposure and the time taken to measure hearing sensitivity after exposure was comparable between the two approaches (Finneran et al. 2007). Such differences are likely attributable to inherent differences between the two approaches. Further investigations are required to work out these differences and to interpret TTS results using evoked potential audiometry (Finneran et al. 2007).

A few studies have also investigated the effects of impulsive sounds (transient sounds having rapid rise times and high peak levels) on MTTS in bottlenose dolphins and beluga whales. In these studies, behavioral methods were used to measure masked hearing thresholds before and after exposure in San Diego Bay. No MTTS (defined as ≥ 6 dB) was observed in either species after exposure to sounds simulating underwater explosions with SELs up to 179 dB re 1 $\mu\text{Pa}^2\text{s}$, but behavioral responses above 156 dB re 1 $\mu\text{Pa}^2\text{s}$ were sometimes noted (Finneran et al. 2000). In another study, MTTS was measured after exposure to single underwater impulses from a seismic water gun. MTTSs of 7 and 6 dB were measured at 0.4 and 30 kHz, respectively, in a beluga whale after exposures of 186 dB re 1 $\mu\text{Pa}^2\text{s}$. Recovery occurred within 4 minutes after exposure. No MTTS greater than or equal to 6 dB was observed in the bottlenose dolphin up to SELs of 188 dB re 1 $\mu\text{Pa}^2\text{s}$ (Finneran et al. 2002b).

It is unclear what effect the masking noise had on the threshold shifts from impulsive sound exposure because these experiments were not repeated in quieter conditions. In some cases, physiological responses to sound exposure were also investigated (Romano et al. 2004) and these are reviewed in the Nonauditory Effects section below.

The TTS investigations on delphinids reviewed here involved sound exposures with constant exposure levels. However, free-ranging animals are often exposed to sounds that are intermittent in time and space. While very little work has been done on the issue of sound exposure intermittence on TTS in marine mammals, such exposures in people yield somewhat complicated results, partly because of partial recovery during the quieter periods. For very short interruptions or small fluctuations, TTS is proportional to the average exposure level. With more variability in sound exposure levels, TTS is not a consistent function of the total energy of the exposure (Ward 1997).

Another consideration is that even between similarly related species, TTS at a given SEL can vary dramatically. This might be due to differential sensitivity at the frequencies of the sound exposure. Bottlenose dolphins and beluga whales have similar audiograms (Schlundt et al. 2000) which might explain why similar sound exposure levels produced comparable amounts of TTS. Hearing sensitivity at both 4 and 20 kHz is roughly 10–20 dB better in the killer whale than in the bottlenose dolphin, as shown in Figure 8. However, these results are based on the performance of only a few individuals (n = 1 or 2). Some investigators argue that it might be more appropriate to normalize SEL relative to baseline hearing sensitivity to account for these differences (Kastak et al. 2005). Thus while TTS results of the bottlenose dolphin and beluga whale provide the best available data to assess potential hearing loss due to sound exposure in killer whales, differences in hearing sensitivity at a given frequency might produce more or less TTS as a result of sound exposure at that frequency.

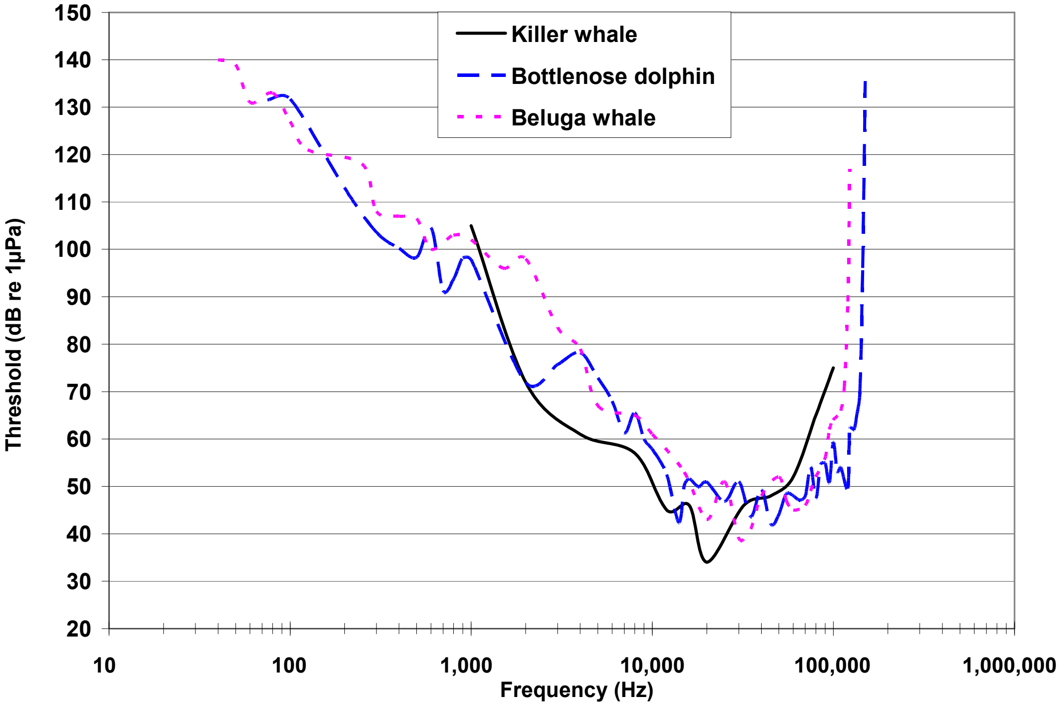

Figure 8. Behavioral audiograms plus electrophysiological thresholds below 2 kHz for the killer whale (based on Szymanski et al. 1999), bottlenose dolphin (based on Johnson 1967), and beluga whale (based on White et al. 1977, Awbry et al. 1988, Johnson et al. 1989).

Behavioral Changes in the Presence of Unwanted Sound

Other effects of sound exposure include behavioral changes that might have repercussions on life functions of affected animals. Behavioral responses to sound exposure can be highly variable and depend not only on factors related to the characteristics of the sound source, its transmission, and background ambient noise levels, but also on individual, group, population, and species-level differences arising from factors such as age, sex, dependent offspring presence, hearing sensitivity/loudness perception, activity patterns, motivational states, previous experience that may have resulted in habituation or sensitization, and noise tolerance (NRC 2003, 2005).

There have been many studies investigating the effects of sound exposure on the behavior of marine mammals, including those on killer whales. These responses may be short-term changes such as changing swimming direction, dive duration, or vocal behavior, or long-term changes such as avoiding a once popular area for foraging, breeding, or socializing (Morton and Symonds 2002, Foote et al. 2004, Bain et al. 2006, Williams and Ashe 2006, Williams et al. 2006). Most of these studies involve investigating behavioral changes in the presence of anthropogenic sources relative to some baseline behavioral measurement. However, in some cases, it is unclear whether the physical presence of the sound-producing device (such as a moving vessel) or the sound itself caused behavioral changes.

Behavioral Patterns

Behavioral activities of SRKWs were observed in the presence and absence of vessels within 1,000 m over the summers of 2003–2005 (Bain et al. 2006). Activity budgets (foraging, rest, traveling, and socializing) were reported at two sites along the west side of San Juan Island, a north site (lat 48°30.561′N, long 123°8.494′W) and a south site (lat 48°27.421′N, long 122°59.401′W). Whales spent more time traveling and less time foraging in the presence of boats within 100 and 400 m of the focal whale group than in their absence. No difference between foraging, resting, and traveling between study sites was found, but whales spent more time socializing in the north than in the south site. They were more likely to continue foraging when boats were absent compared to when they were present within 100 and 400 m, but it was possible the responses at 400 m were a result of the effect at 100 m. Whales also traveled in less direct paths and had longer average durations between breaths when vessels were present compared to when they were absent within 1,000 m. Surface active behavior was not notably more frequent in the presence of vessels than in their absence within 1,000 m, and there was no significant difference in swim course or speed due to vessel traffic (Bain et al. 2006).

Williams et al. (2006) also found that vessel presence changed the behavioral patterns of Northern Resident killer whales in Johnstone Strait, British Columbia. Whales spent significantly less time feeding and more time traveling, socializing, and resting in the presence of vessels than in their absence within eight zones, four within a reserve where no boats were allowed and four adjacent zones. While the overall energetic demand was estimated to be 3% higher in the presence of boats, the lost opportunity to feed had a larger energetic effect estimated to be about a 28% decrease in energetic gain (Williams et al. 2006).

The significance of these results as well as those of Bain et al. (2006) is contingent on the ability of researchers to accurately discriminate different behavioral states of free-ranging killer whales. Williams and Ashe (2006) also conducted controlled experiments in which adult focal male Northern Residents in Johnstone Strait were tracked when no boats were within 1,000 m, then in the presence of a few (1–3) or many (>3) boats within 1,000 m. Males swam in less direct paths in the presence of a few boats within 1,000 m, but swam in more direct paths when there were many boats present within 1,000 m. If swimming directedness had been analyzed simply in the presence or absence of boats, then this difference in behavior would not have been apparent. An investigation of Southern Resident behavior with varying number of boats would also be useful to determine boat number effects. However, unlike in the Northern Residents, there are fewer opportunities to observe Southern Resident behavior when no boats or a few boats are near them.

Active Sonar and other Anthropogenic Sounds

There was also a documented case of exposure by J pod to military active sonar by the USS *Shoup* on 5 May 2003 in the Haro Strait (U.S. Navy 2004). The USS *Shoup*, a guided missile destroyer, was equipped with a AN/SQS-53C(V)4 hull-mounted sonar with kingfisher mine avoidance system that produces frequency-modulated signals between 2.6 and 3.3 kHz at a nominal source level of 235 dB re 1 μPa at 1 m. This system's midfrequency tactical sonar is designed to detect, localize, and avoid mines and other objects. The goal of the Navy on that day was to train ship personnel to use the sonar system in a "sweep channel" exercise in which 1–2 second signals were emitted once every 28 seconds from 1123 to 1438 hours (U.S. Navy 2004). During part of the exercise, J pod was in the area and observed by several researchers familiar with SRKW behavior who described the whales as exhibiting "abnormal behavior" including bunching as a group close to shore (NMFS 2005a).

The sonar signals were recorded by Val Veirs on bottom mounted hydrophones and recordings were provided to NMFS, but the dynamic range of the system prevented accurate assessments of the highest received levels. Thus the Navy Research Laboratory estimated received levels relative to the whales. Mean received levels relative to J pod were estimated to range between 121 and 175 dB re 1 μPa and likely received SELs were 169.1–187.4 dB re 1 μPa^2s with a worse case range estimated to be 177.7–195.8 dB re 1 μPa^2s. NMFS (2005a) reported that it was likely that these received levels were audible to the whales and that reverberation from the complex underwater environment may have reduced the ability of the whales to localize the sound source.

TTS onset was estimated based on TTS studies conducted on dolphins (reviewed in the Hearing Loss Due to Sound Exposure subsection above) and except in the worse case scenario, NMFS (2005a) stated that it was not likely that the whales experienced temporary or permanent hearing loss (based on a TTS onset level of 195 dB re 1 μPa^2s), but that the received levels were high enough to induce a behavioral reaction that was consistent with observer reports.

Longer-term effects of sound exposure were observed in both resident and transient killer whales in the Broughton Archipelago, British Columbia, when acoustic harassment devices (AHDs) were in operation. The AHD signals were designed to repel harbor seals from salmon farms by producing a 10 kHz signal at a source level of 194 dB (re 1 μPa at 1m). It was estimated that the received levels of the AHD signals would reach ambient noise levels at about 50 km from the source. The number of days killer whales were sighted in the area was significantly reduced after the AHD broadcasts, compared to before (Morton and Symonds 2002).

Other anthropogenic sounds described in SRKW critical habitat are those associated with construction activity, particularly pile driving given its relatively loud and impulsive acoustic signature at close ranges. Although behavioral effects of pile driving or other marine construction noise on killer whales has not been reported, Wursig et al. (2000) found that Indo-Pacific humpback dolphins (*Sousa chinensis*) significantly increased their group swim speed during pile driving even with the deployment of a bubble curtain, but such human activity did not have a long-term overall effect on dolphin abundance in the area.

Vocal Response to Background Noise

Another response to sound exposure involves changes in vocal behavior. In humans, an unconscious response to speak louder with higher background noise levels is called the Lombard effect/response (Lombard 1911) and is often observed when people wear headphones and talk while listening to loud music. Many animals also call louder, longer, more often, or at different frequencies in the presence of masking noise (Brumm and Slabbekoorn 2005). For example, beluga whales in the St. Lawrence River responded to increases in ambient noise from boat traffic by increasing call amplitude (Schiefele et al. 2005), call repetition rate, and frequency of the call (Lesage et al. 1999). SRKWs significantly increased the duration of their primary stereotyped call in the presence of boats compared to in their absence in the more recent time periods, which correlated with a dramatic increase in the number of whale watching boats in their habitat (Foote et al. 2004). In laboratory settings, bottlenose dolphins increased the average number of clicks emitted per trial as white noise levels were increased during an echolocation target detection task (Au et al. 1982).

Biogenic noise from other animals also results in vocal compensation effects. For example, Antarctic killer whales shifted the frequency modulated points of their calls below or above the frequency range of predominant leopard seal (*Hydruga leptonyx*) calls during the season when leopard seals were calling (Mossbridge and Thomas 1999). A captive beluga whale produced louder and higher frequency echolocation clicks when it was moved from San Diego Bay, California, to Kaneohe Bay, Hawaii, which had ambient noise levels that were typically 12–17 dB higher than in San Diego due to snapping shrimp presence (Au et al. 1985). Thus when it comes to sound production, there are strategies that animals use to attempt to compensate for increased ambient sound levels. When animals exhibit such vocal compensation behaviors, it can be interpreted that auditory masking is a potential challenge, particularly when ambient sound levels reach a point where animals can no longer compensate by increasing their vocal output or other sound emission parameter.

Strandings and other Nonauditory Effects of Sound Exposure

An extreme behavioral outcome to sound exposure is stranding. In some cases, cetacean strandings have coincided in space and time with military sonar exercises. There are three well-documented cases, one in Greece in 1996, one in the Bahamas in 2000, and one in the Canary Islands in 2002, and all involved the mass stranding of 11–14 individual beaked whales (and sometimes other cetacean species). While there is a general consensus among the scientific community that military sonar was a likely causal link to the strandings, the mechanisms which caused the observed pathologies are still a mystery (NRC 2005). For example, results from necropsies performed on 10 of the 14 individuals involved in the Canary Island stranding event revealed extensive bubble formation and tissue damage in various organs. A number of hypotheses including decompression sickness and acoustically mediated bubble growth due to tissue supersaturation of nitrogen (via rectified diffusion) were proposed, but are highly debated explanations (Crum and Mao 1996, Fernandez et al. 2004, 2005, Jepson et al. 2003, Piantodosi and Thalmann 2004).

There are no documented cases of killer whale strandings that coincide with military sonar exercises; however, several harbor porpoises (*Phocoena phocoena*) stranded before, on, and after the USS *Shoup* exercise on 5 May 2003. Forensic examination revealed no evidence of acoustic trauma but inadequate preservation of the samples produced equivocal results (NMFS 2005a).

Other less extreme, but potentially serious, nonauditory effects of noise include physiological effects related to stress. Most studies investigating these effects have been conducted on terrestrial mammals and include changes in heart, respiration, and metabolic rate, and immune and reproductive function.

A few studies have investigated the production of stress hormones including catecholamines (such as norepinephrine, epinephrine, and dopamine) and aldosterone (implicated in a longer-term stress response in marine mammals) as well as decreased immune function resulting from controlled sound exposures in delphinid subjects. For instance, Thomas et al. (1990) found no difference in the levels of catecholamines in beluga whales after exposure to playbacks from an oil drilling platform. Romano et al. (2004) also measured the effects of sound exposure on nervous system and immune function in laboratory subjects of ongoing TTS studies (Finneran et al. 2002b).

Norepinephrine, epinephrine, and dopamine levels were significantly higher in the beluga whale after high-level (< 183 dB re 1 $\mu Pa^2 s$) compared to low-level (>183 dB re 1 $\mu Pa^2 s$) or control exposures of impulses from a seismic water gun. The bottlenose dolphin did not show differences in catecholamine levels but did have higher aldosterone levels and lower absolute monocyte (cells of the immune response) counts after experimental exposures compared to those measured after control exposures (Romano et al. 2004). It appears that sound exposure in some instances results in measurable differences in the production of stress hormones and cells related to immune function that could have undesirable physiological effects in free-ranging delphinids.

Zones of Influence

The previous sections have reviewed studies on the effects of sound exposure in killer whales and other odontocetes. These studies have shown that 1) marine mammals, including killer whales and closely related species, are vulnerable to auditory effects of noise exposure including masking and hearing loss effects; 2) individuals and groups of killer whales, including Southern Residents, exhibit behavioral changes that are consistent with avoidance responses in the presence of vessels and AHD signals; 3) physiological changes with undesirable effects on endocrine and immune function in smaller dolphins have resulted from sound exposure; and 4) in extreme cases, beaked whale strandings have been linked spatiotemporally with midfrequency military sonar exercises.

One objective of this review is to use results from previous investigations to assess potential impacts of sound exposure from various sources on SRKWs. An approach to this assessment is to estimate the zone where an acoustic effect is expected based on available data (Richardson et al. 1995). Four zones are defined based on different response effects. In general, these are listed in order of smaller radii of influence relative to the acoustic source as follows: 1) zone of audibility is the area where the received levels of sound exposure are high enough that a killer whale can hear it; 2) zone of responsiveness is the area where the received level is such that the whale responds behaviorally or physiologically; 3) zone of masking is the area where received levels can mask the detection of sound emissions for biosonar and vocal communication, prey sounds, or other biologically important sounds; and 4) zone of hearing loss or injury is the area where received levels (or sometimes more appropriately SELs) are high enough to cause temporary or permanent hearing loss.

Richardson et al. (1995) took the approach of describing these zones by distance, but accurate assessments of range require adequate knowledge of sound propagation characteristics. Here, zones of influence are by and large described by received levels given the complex shallow water environments inhabited by Southern Residents and the uncertainty involved in predicting propagation loss in space and time in these areas. This focus on received levels ignores the influence of temporal factors such as duration, pulse repetition rate, or duty cycle of the sound exposure that also likely contribute to potential response effects. Where information is available and appropriate, such as in the discussion of zones of hearing loss, SELs that take the duration of exposure into account will also be discussed.

Zone of Audibility

Whether or not a sound produced by human activity will be audible to a Southern Resident killer whale depends on the interplay between source, path, and receiver variables (for review, see Richardson et al. 1995). Source variables include source level, duration, and frequency of the sound; path variables include properties of transmission loss that affect received levels; and receiver variables include hearing sensitivity and responsiveness of the whales with respect to the received sound. If the received sound level falls below the detection threshold of the animal, then the signal is said to be "threshold limited," while if it is above the detection threshold but below the ambient noise level at all frequencies that killer whales are sensitive to, it is said to be "noise limited" (Richardson et al. 1995). The absolute hearing sensitivity of killer whales is generally below the typical one-third octave ambient noise level for frequencies less than 3 kHz and greater than 70 kHz (arrows in Figure 9) in low ambient noise situations such as in calm conditions (e.g., a sea state of 0) in about 200 m of water. Thus killer whale hearing will be threshold limited in these conditions.

Whales will likely be threshold rather than noise limited at lower frequencies. This prediction is based on the assumption that the critical bandwidths of the killer whale are approximated by one-third of an octave, which is a typical assumption with lack of direct measurements of critical bandwidths. More direct predictions can be made using the critical ratios of 20 dB re 1µPa at 10 kHz and 40 dB re 1µPa at 80 kHz (Bain et al. unpublished data, cited in Bain and Dahlheim 1994) and spectrum levels measured in the areas that SRKWs frequent such as those reported by Veirs and Veirs (2005). For example, the received level of a 10 kHz tonal signal would have to be approximately 78 dB in order to be detected at night in Haro Strait because ambient spectrum levels at this frequency are 58 dB re 1µPa^2/Hz and the critical ratio is 20 dB.

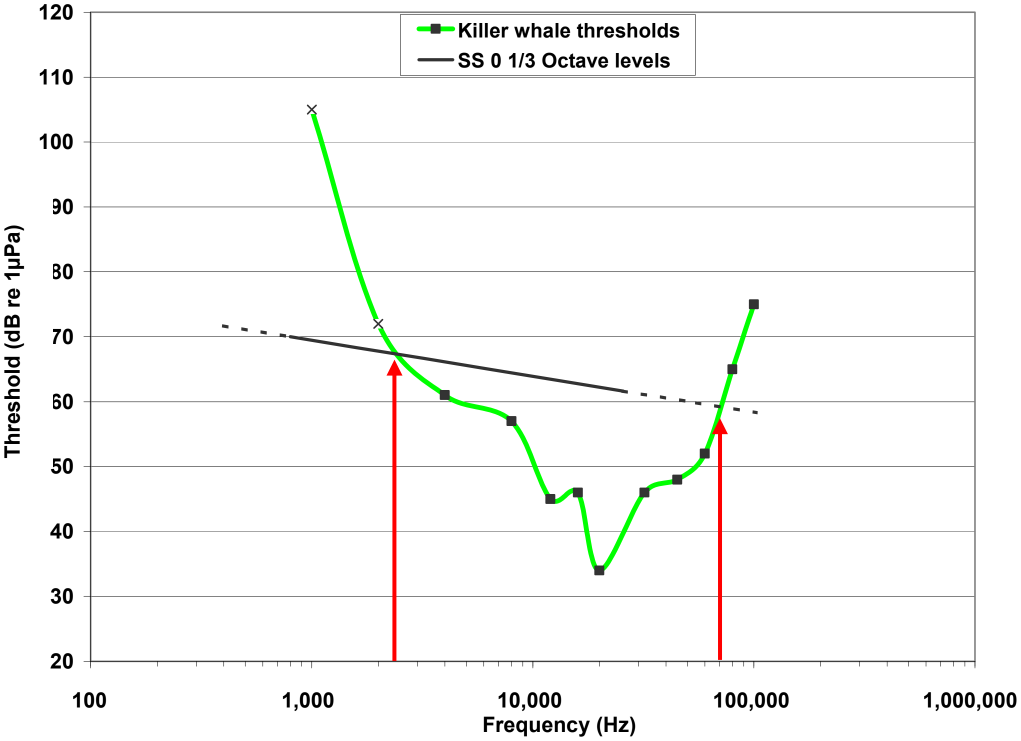

Figure 9. Killer whale behavioral audiogram based on Szymanski et al. 1999 (reprinted from Figure 5c in Szymanski et al., copyright 1999, with permission from Acoustic Society of America), and one-third octave ambient levels in a sea state of 0 from Richardson et al. 1995 (reprinted from Figure 5.1b in Richardson et al., copyright 1995, with permission from Academic Press). Dashed lines are extrapolated values and the two × marks indicate physiological thresholds to supplement information on sensitivity at the lower frequencies. Arrows indicate the frequency boundaries at which hearing would be noise versus threshold limited.

The degree to which hearing will be noise limited will depend on the ambient conditions which are quite variable. Based on the Szymanski et al. (1999) audiograms, Erbe (2002) estimated that vessel noise levels based on one-twelfth octave band analysis was sufficiently loud enough to exceed the hearing thresholds of killer whales at frequencies above 2 kHz. It was estimated that a fast inflatable operating at speed of 51 km/hour and a source level of 162 dB re 1μPa would be audible to killer whales over a range of about 16 km in a sea state of one half. A slow inflatable operating at a speed of 10 km/hour would be audible over a shorter range of 1 km in the same sea state (Erbe 2002). The model used assumed that the limit of audibility occurs when the critical band level becomes equal to or less than the ambient noise. However, one-twelfth octave critical bandwidths were assumed across all frequencies based on Fletcher's (1940) equal power assumption. The zone of audibility for this boat noise might be different if direct measurements of killer whale critical bandwidths were available.

Zone of Responsiveness

In this section, the zone of responsiveness is defined as the area where the received level is such that killer whales would respond behaviorally. Most studies that illustrate statistically significant effects on the behavior of killer whales in the presence of anthropogenic sound provide insufficient information related to the source or received sound levels, frequencies, and duration of exposure, making it extremely challenging to predict the zone of responsiveness. The level of ambient noise, which can fluctuate in space and time and the resulting effective signal-to-noise ratio, also adds complexity to this issue. Furthermore, reactions based on other sensory information such as visual cues cannot be ruled out in the absence of controlled studies (such as those involving playbacks).

Based on the little information available, it appears that killer whales will avoid an area within about 4 km of an AHD producing a 10 kHz signal at a source level of 194 dB re 1 μPa at 1 m (Morton and Symonds 2002, NMFS 2005a). The transmission loss characteristics of the waters surrounding the AHD broadcasts are unknown; but, if we assume that at the most it would take the form of spherical spreading loss and at the least cylindrical spreading loss, the likely range for received levels at 4 km can be determined. The validity of these spreading loss assumptions is tentative. Using the following equation to calculate received levels,

$$RL = SL - TL \qquad (6)$$

where RL is the received level, SL is the source level of the signal, and TL is the transmission loss, where $TL = 20 \log R + \alpha R$ for spherical spreading or $TL = 10 \log R + \alpha R$ for cylindrical spreading, and where α is the absorption loss at 10 kHz and is approximately 0.00118 dB/m (Urick 1983), this would correspond to a AHD received level ranging from 117 to 153 dB re 1 μPa.

Based on the response of J pod to the sonar exercise by USS *Shoup* in May 2003, Southern Residents are predicted to behaviorally respond to frequency modulated signals between 2.6 and 3.3 kHz at received levels estimated between 121 and 175 dB re 1 μPa and received SELs of 169.1–187.4 dB re 1 $\mu Pa^2 s$.

In the absence of data on the effects of vessel noise levels on killer whale behavior, Erbe (2002) assumed that a received broadband sound pressure level of 120 dB re 1µPa based on Richardson et al. (1995) would produce a behavioral reaction by killer whales. A behavioral response was predicted to occur when the vessel was within a 200 m and a 50 m range for an inflatable operating at 51 km/hour and 10 km/hour, respectively. Effects on the activity budgets of Southern Residents in the presence of vessels within 400 m as reported by Bain et al. (2006) agree with these predictions by Erbe (2002). In addition, the lower limits of the estimated received levels of both AHDs and midfrequency sonar signals that produced behavioral responses in Northern and Southern Resident killer whales fall within this range.

Furthermore, if it is assumed that received levels above 120 dB would produce a significant behavioral response, then this would occur 50% of the time during summer days at the position of the recording hydrophones based on 2-second sound pressure level averages reported in the Veirs and Veirs (2005) study. This type of criterion based on one broadband sound pressure level measurement does not take into account frequencies of best hearing sensitivity of killer whales. For example, most energy of the received levels measured by Veirs and Veirs (2005) falls below 1 kHz, which is outside the range of best hearing sensitivity of killer whales (Szymanski et al. 1999). Weighting functions that emphasize frequencies of best hearing sensitivity while deemphasizing those of least hearing sensitivity based on the killer whale audiogram (or equal loudness level contours if available) would be useful to measure exposure levels in order to address these frequency effects.

Zone of Masking and Effects on the Active Space of Sound Emissions

The active space of a signal is the range over which it can be detected. Any sound that masks the reception of a communicative signal, such as conspecific call or echolocation click, will reduce the active space of that signal. Detection of natural signals a few dB below ambient noise levels has been demonstrated in humans and some marine mammals (Richardson et al. 1995). Captive killer whales have demonstrated that detection thresholds of clicks and calls are not impeded by boat noise, although in some cases, frequency overlap was not a likely contributing factor (Bain and Dahlheim 1994). The sonar equation provides a useful way to describe the active space of an acoustic signal. To determine the range at which a sound might be heard by a listener, the sonar equation may be defined as follows:

$$RL = SL - TL - NL + DI \qquad (7)$$

where NL is the noise level and DI is the directivity index of the auditory system (Urick 1983).

In most cases where background noise exists, the received level would have to exceed the CR in order for detection to take place. In the case of auditory masking, NL increases, reducing the received level of the signal. If the source level of a signal is not below the hearing threshold, the excess level (in dB) can be used to calculate the distance over which it might be heard after transmission loss has been taken into account. Assumptions about transmission loss vary considerably as well as the bandwidth appropriate for analyzing background noise. Based on a one-twelfth octave bandwidth analysis, Erbe (2002) estimated an inflatable operating at a speed of 51 km/hour would mask the reception of a killer whale call to the point of being unrecognizable (when spectral components at 4.7 and 5.8 kHz were inaudible) within a range of 14 km. An inflatable operating at a speed of 10 km/hour was predicted to mask killer whale calls within a 1 km range. However, the killer whale call level incorporated into the model was based on received levels (up to 124 dB re 1µPa) and not source levels which are known to be considerable higher (135–168 dB re 1 µPa at 1 m, Miller 2006). Thus the range of masking effects is likely overestimated in this instance.

Miller (2006) reported that the mean active space of stereotyped calls with high-frequency components was 10–16 km in sea state 0 while the mean active space of stereotyped calls without high-frequency components, whistles, and variable calls was 5–9 km. In these cases, a killer whale call was assumed to be detectable by another whale when the received level in at least 1 one-third octave band exceeded the hearing threshold or was 6 dB below the background noise level, whichever was greater. Relative to the sea state of 0 in which noise spectral levels are about 26 dB re 1µPa^2/Hz lower, estimated active space of calls in a sea state of 6 were 75%, 83%, and 91% lower for stereotyped calls, variable calls, and whistles, respectively, using Fletcher's (1940) equal power assumption (Miller 2006). The frequency and time structure of signals relative to that of the noise as well as the directional hearing system might result in increased sensitivity to these signals, but these considerations were not incorporated in the models for simplicity (Erbe and Farmer 2000, Erbe, 2002, Miller 2006).

For echolocation signals, the sonar equation is modified to include two-way transmission loss as well as the target strength (TS) of the ensonified object as follows:

$$\mathrm{RL} = \mathrm{SL} - 2\mathrm{TL} + \mathrm{TS} - \mathrm{NL} + \mathrm{DI} \qquad (8)$$

Au et al. (2004) used these equations to estimate the horizontal range at which a foraging killer whale would be able to perceive biosonar echoes off prey in quiet conditions. This was based on a model of Chinook salmon (*Oncorhynchus tshawytscha*) target strength and an estimated echo level (EL) received by a whale using the following equation:

$$\mathrm{EL} = \mathrm{SL} - 2\mathrm{TL} + \mathrm{TS} \qquad (9)$$

Based on these assumptions and using dB peak-to-peak levels, it was estimated that an echolocating killer whale would receive echoes of a Chinook salmon at 65 m depth between 29 and 33 dB above threshold at a horizontal range of 100 m. In noisier conditions, the noise level along with the DI must also be considered. The DI of a killer whale was estimated to be 21 dB at 50 kHz based on a scaling factor to convert the DI of a bottlenose dolphin, since no direct measurements of DI in killer whales were available (Au et al. 2004). For moderately heavy rain, the ambient noise spectral density level was estimated to be 52 dB re 1µPa^2/Hz and it was predicted that the echo level would not extend above the noise level until the whale was within a horizontal range of 40 m from the prey item (Au et al. 2004). Thus the active space of the echolocation signal was reduced considerably from quiet to noisier conditions created by heavy rain. It is important to note that the quiet condition assumed a complete lack of ambient noise, which is not realistic. Free ranging animals will never be in an environment that is completely lacking of ambient noise.

The analysis of active space for echolocation signals can be extended for conditions when vessels are operating in the vicinity of killer whales. Based on data from Hildebrand et al. (2006), spectral levels of vessel and Haro Strait ambient noise were used to estimate the horizontal detection range of killer whales echolocating on Chinook salmon. For this analysis, source spectral density levels at 50 kHz for cruise and power up operating speeds were available to predict received levels of a whale at 100, 200, and 400 m from two whale watching vessels. Received levels for these conditions are based on the following equation:

$$\mathrm{RL} = \mathrm{SL} - \mathrm{TL} = \mathrm{SL} - 20 \log \mathrm{R} + \alpha \mathrm{R} \tag{10}$$

where SL is the source level at 50 kHz for a given vessel at a given speed, R is the range in meters, and α is the absorption loss at 50 kHz and is approximately 0.016 dB/m (Urick 1983) as shown in Table 1.

Table 1. Source levels for whale watching vessels at different speeds, calculated received levels assuming spherical spreading loss for three distances, and measured received level of a container ship as reported by Hildebrand et al. (2006).

Vessel	**Speed**	**SL at 50 kHz dB re 1µPa^2/Hz**	**Range (m)**	**Calculated RL at 50 kHz in dB re 1µPa^2/Hz**	**Measured RL at 50 kHz in dB re 1µPa^2/Hz**
Boat B	Cruise	101	100	59.4	
		101	200	51.8	
		101	400	42.6	
	Power	92	100	50.4	
		92	200	42.8	
		92	400	33.6	
Boat C	Cruise	93	100	51.4	
		93	200	43.8	
		93	400	34.6	
	Power	95	100	53.4	
		95	200	45.8	
		95	400	36.6	
Boat D	Cruise	111	100	69.4	
		111	200	61.8	
		111	400	52.6	
	Power	104	100	62.4	
		104	200	54.8	
		104	400	45.6	
Ship (*Hanjin Marseilles*)	21 knots	107	100	65.4	
		107	200	57.8	
			442		48
Ambient (measured on day after MV *Hanjin Marseilles* pass)					26

Transmission loss was assumed to be spherical which is likely only accurate in deep water. The range of distances between whales and vessel noise is pertinent based on the voluntary “Be Whale Wise” guidelines, which included maintaining cruising speeds of less than 7 knots (13 km/hour) within 400 m of the whales and a “no go” zone within 100 m of the whales (http://www.nwr.noaa.gov/Marine-Mammals/upload/BeWhaleWise.pdf). Additionally, the received spectral density level at 50 kHz of a container ship at 442 m was used. The NL that a killer whale auditory system would receive was calculated as

$$NL = N_0 + BW - DI \tag{11}$$

where N_0 is the noise spectral density of the noise, BW is the received bandwidth, and DI is the directivity index used in the previous investigation (Au et al. 2004).

This NL was based on RMS measurements, so 9 dB were added to convert to peak-to-peak levels as in the previous analysis by Au et al. (2004). The maximum horizontal range (m) was taken as the horizontal distance between a whale at 1 m and a salmon at 65 m of depth in which the echo levels off the fish would extend into the noise floor. Echo levels were determined by the same methods as in Au et al. (2004). Ambient levels in the Haro Strait were those reported by Hildebrand et al. (2006) and are comparable to ambient levels between a sea state of 1 in deep water (Urick 1983). It was assumed that only the noise from one vessel was present and other sources, including natural ones, were not included in the calculation of NL. The results of this analysis are shown in Table 2 and Figure 10.

Compared to ambient levels in the Haro Strait in which the maximum horizontal detection range was predicted to be 400 m, boat noise generated by cruise and power up speeds up to 400 m relative to the whales was predicted to significantly reduce the active space of an echolocation click at 50 kHz. At a distance of 400 m between vessel and whale, Boat B at power up speed and the Boat C at cruise speed were predicted to have the least impact (maximum horizontal detection range was 250 m, resulting in a 150 m reduction in active space compared to Haro Strait ambient levels), while Boat D at cruise was predicted to have the greatest impact that was similar to the masking effects of heavy rain (maximum detection range was 40 m, thus the active space was effectively reduced by 360 m).

The predicted distances at which noise produced by cruising (> 20 knots) whale watching vessels would approach ambient levels in Haro Strait are 650, 950, and 1,400 m for Boat C, Boat B, and Boat D, respectively. The zones of masking for these vessel types are defined within these ranges. Even the large container ship (MV *Hanjin Marseilles*) passing at 442 m relative to a whale was predicted to reduce the active space of the biosonar signal by 340 m relative to Haro Strait ambient conditions.

The above analysis is oversimplified because it only considers one frequency and echolocation clicks are broadband. Whales most likely use acoustic information across a range of frequencies to detect prey through biosonar. Additionally, as stated by Au et al. (2004), the DI assumes that the noise is isotropic (independent of direction) and vessel noise at this range is likely anisotropic. For example, the whales might be able to swim to some depth to avoid vessel noise at the surface, but such behavior might also have energetic costs. The target strength of a

Table 2. Predicted maximum horizontal detection ranges at 50 kHz for a killer whale at the surface echolocating on a Chinook salmon at 65 m depth, reduction in range relative to ambient Haro Strait measurement, and percent of reduction in range relative to ambient Haro Strait measurement.

Noise condition	**Horizontal echo detection range (m)**	**Reduction in range re ambient Haro Strait (m)**	**% reduction in range re ambient Haro Strait**
No noise	650	–	–
Sea state of 1	450	–	–
Ambient Haro Strait	400	–	–
Heavy rain	40	–	–
Boat B cruise at 100 m	20	380	95
Boat B cruise at 200 m	40	360	90
Boat B cruise at 400 m	100	300	75
Boat B power at 100 m	50	350	88
Boat B power at 200 m	100	300	75
Boat B power at 400 m	250	150	38
Boat C cruise at 100 m	40	360	90
Boat C cruise at 200 m	100	300	75
Boat C cruise at 400 m	250	150	38
Boat C power at 100 m	40	360	90
Boat C power at 200 m	80	320	80
Boat C power at 400 m	200	200	50
Boat D cruise at 100 m	0	400	100
Boat D cruise at 200 m	20	380	95
Boat D cruise at 400 m	40	360	90
Boat D power at 100 m	10	390	98
Boat D power at 200 m	30	370	93
Boat D power at 400 m	80	320	80
Hanjin (ship) at 100 m	0	400	100
Hanjin (ship) at 200 m	20	380	95
Hanjin (ship) at 442 m	60	340	85

Chinook salmon was modeled using tonal sonar signals and a model of a fish (Au et al. 2004) and not with killer whale-like echolocation signals and real fish to determine target strength values (see Au et al. 2007). Furthermore, information on target detection thresholds in killer whales in quiet conditions is lacking, so interpreting appropriate detection ranges predicted from this analysis is unknown.

The range at which killer whales would detect, discriminate, and consider pursuing prey is debatable as discussed in Au et al. (2004). For example, even if the detection of prey is predicted at 400 m in quiet ambient conditions (Figure 10), it does not necessarily mean that whales will choose to pursue the prey. For these reasons, the data from this analysis should be interpreted with caution, given the lack of information about predator-prey interactions in free-ranging killer whales.

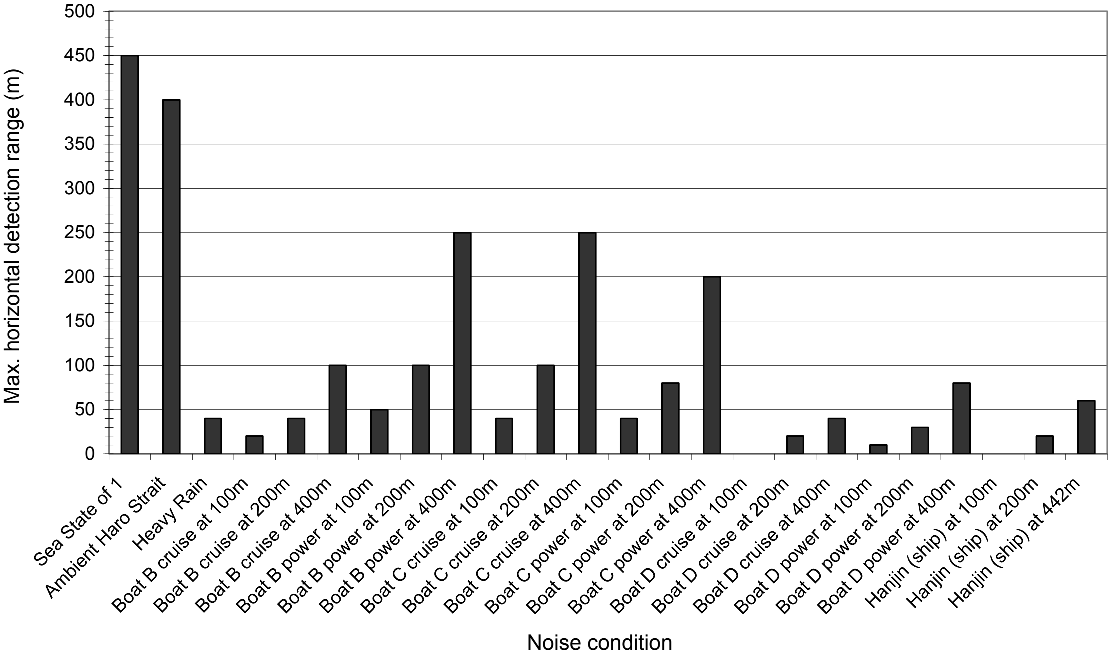

Figure 10. Predicted maximum horizontal detection ranges at 50 kHz for a killer whale at the surface echolocating on a Chinook salmon at 65 m depth for various noise conditions.

Zone of Hearing Loss or Injury

The zone of hearing loss or injury is the area where received levels or sound exposure levels are high enough to cause temporary or permanent hearing loss. Studies on TTS in various marine mammal groups are needed (NRC 2003) because of their utility to predict auditory injury or PTS. Hearing loss effectively reduces the active space of important signals such as conspecific calls or biosonar signals. That is, whales that experience temporary or permanently elevated hearing thresholds must be closer to a source to detect it. TTS data from studies conducted on bottlenose dolphins and beluga whales have been used to predict the zones of hearing loss (TTS) and injury (PTS) in killer whales. In particular, the TTS studies that have used white noise as fatiguing stimuli (Nachtigall et al. 2003, 2004) are most appropriate for extrapolating potential hearing loss in killer whales due to vessel noise exposure, those using pure tone fatiguing stimuli (Finneran et al. 2005, 2007) are most appropriate for sonar exposure, and those using impulsive sounds (Finneran et al. 2000, 2002b) are most appropriate for pile driving activity.

Erbe (2002) used preliminary data from the Nachtigall et al. (2003) study as cited in Au et al. (1999) in which a TTS of 12–18 dB at 7.5 kHz occurred in a bottlenose dolphin after exposure to an octave band of noise at 179 dB re 1µPa for approximately 50 minutes (resulting in an SEL of 213 dB re 1µPa^{2}s). This corresponded to a sensation level of 96 dB (amount above the pure tone threshold of the subject at 7.5 kHz). Within a few meters, octave band noise levels of whale watching boats were usually less than 96 dB above the threshold of killer whales in this bandwidth. Therefore, Erbe (2002) used data from humans and other terrestrial animals to scale down the amount of TTS and assumed that exposure levels of 68–74 dB above threshold for the same exposure period would result in a 5 dB threshold shift. At that time, the accuracy of this assumption was questionable because no TTS was observed in the bottlenose dolphin for exposure levels of 87 dB above threshold (Au et al. 1999). It was predicted that a TTS of 12–18 dB would result from exposure of a fast inflatable (traveling at 51 km/hour) within a 10 m range after 30–50 minutes. A TTS of 5 dB was predicted from exposure of the same inflatable over the same duration within 450 m range. Exposure to a slower moving inflatable (10 km/hour) would result in TTS of 5 dB after 30–50 minutes within a 20 m range (Erbe 2002).

Based on more current work by Nachtigall et al. (2003), a TTS of 11 dB at 7.5 kHz would result from exposure to a fast inflatable within a 10 m range after 50 minutes of continuous exposure. In another study using the same octave band of noise for exposure, an electrophysiological approach was used to measure postexposure hearing thresholds more rapidly and over a wider range of frequency. Lower amplitude and shorter duration exposures produced measurable TTS (Nachtigall et al. 2004) compared to a lack of TTS measured through behavioral techniques (Nachtigall et al. 2003), but differences in results from these two approaches might be related to methodological issues that are still being resolved (Finneran et al. 2007). Based on the current data, a killer whale would experience a TTS of 5 dB at approximately the same frequency (8 kHz) and a TTS of 8 dB at 16 kHz after 30 minutes of continuous exposure within a range from the vessel corresponding to 70 dB above hearing sensitivity.

Based on TTS results in which a 3 kHz pure tone was used for exposure, onset TTS one-half octave higher than the frequency of exposure (4.5 kHz) was predicted at sound exposure levels of 195 dB re 1μPa^{2}s (Finneran et al. 2005). Killer whales, however, are about 10–15 dB more sensitive than bottlenose dolphins within this frequency range, but these behavioral results are only based on a one or two individuals (Johnson 1967, Szymanski et al. 1999). When differences in hearing sensitivity are taken into account, sound exposure levels that are predicted to result in killer whale TTS onset should be considered to be lower by approximately 10–15 dB within this frequency range. This is a conservative estimate given the lack of direct measurements of TTS in killer whales.

Pile driving activity in Friday Harbor had an average SEL that ranged between 171 and 187 dB re 1 μPa^{2}s (Lauglin 2005). Exposure to a seismic water gun resulted in masked TTS of 7 and 6 dB at 0.4 and 30 kHz, respectively but no TTS greater than or equal to 6 dB at 4 kHz occurred in a beluga whale after a comparable SEL of 186 dB re 1 μPa^{2}s (peak-to-peak pressures of 226 dB re 1 μPa). No MTTS greater than or equal to 6 dB was observed in the bottlenose dolphin at 0.4, 4, or 30 kHz up to SELs of 188 dB re 1 μPa^{2}s (peak-to-peak pressures of 228 dB re 1 μPa, Finneran et al. 2002b). While the absolute thresholds of the bottlenose dolphin and beluga whale are very similar at 0.4 and 4 kHz, the beluga whales hearing is better by about 10 dB at 30 kHz (Johnson 1967, White et al. 1977, Awbry et al. 1988, Johnson et al. 1989). The killer whale's threshold at 4 kHz is about 18 dB better while at 30 kHz is 5 dB worse than the beluga whale's thresholds (Syzmanski et al. 1999). No absolute threshold measurements at 0.4 kHz are available in the killer whale. With these sensation level differences in mind, killer whales exposed to the same stimulus might not exhibit a threshold shift greater than or equal to 6 dB at 30 kHz but might experience TTS at 4 kHz. The magnitude of TTS at this frequency cannot be predicted with accuracy without further data.

The range of auditory injury or PTS in killer whales exposed to boat noise has been predicted by Erbe (2002). An approach based on those used for humans was taken in which a weighting function used to predict PTS from broadband exposures was modeled. The killer whale weighting function was taken as the killer whale audiogram subtracted from the critical band levels of noise and the energy was integrated across all frequencies. A PTS of 2–5 dB was predicted for whales exposed within 1 km of a fast inflatable or within 50 m of a slow inflatable, continuously for an 8 hour day, 5 days a week, over 50 years (Erbe 2002). Another approach to predict PTS onset from TTS data is to extrapolate curves of TTS as a function of weighted and unweighted levels (depending on the exposure type) up to a value that would likely be associated with PTS in terrestrial mammals (i.e., TTS ≥ 40 dB). This approach is being considered for a number of groups based on hearing function in order to define noise exposure criteria for free-ranging marine mammals (Southall et al. 2007).

Likelihood of Acoustic Impacts on the SRKW Population

Determining whether sound exposure results in measurable differences in hearing or behavior at the individual or group level has been explored in a few species (Richardson et al. 1995, Nowacek et al. 2007, Finneran et al. 2007). Determining whether the effect is biologically significant, that is, having overall deleterious effects at the level of the population, is difficult (NRC 2005). Currently available information is sparse enough to preclude an accurate assessment of the deleterious effects of sound exposure and related cumulative effects at the individual as well as the population level for Southern Resident killer whales. However, the following discussion, based on the framework provided in NRC (2005), may serve as an outline for recommendations for future work (discussed in the next section) while suggesting some provisional approaches for acoustic impact assessment in the interim period.

Risk Assessment

One approach is that based on risk assessment. Risk assessment requires that the problems are clearly identified and the probability of exposure and the types of biological effects are defined, leading to an estimate of risk. There are many types of potential biological effects, but the focus should be on those that have repercussions on life functions that ultimately affect population status as outlined in the Population Consequences of Acoustic Disturbance model (NRC 2005).

Both the risk of behavioral disturbance and auditory injury at the population level require specific information regarding the type, amplitude, and duration of the exposure; the range over which the exposure might affect free-ranging animals including sound propagation properties; the fraction of the population affected (e.g., through dose-response curves); the intensity of the reaction; and the ease with which animals of the population might recovery from the exposure (such as finding alternative habitats or recovering hearing function).

In many cases, it is very difficult to interpret the biological significance of behavioral disturbances. For example, does the disturbance result from the acoustic signal resembling the sounds of a predator, prey, or conspecific? Or is the response just indicative of annoyance that scales to exposure level? It is often impossible to determine differences in the reasons why animals respond in a particular way because of a lack of power to determine causal relationships between exposure and behavioral changes (NRC 2005). For healthy populations, recovery from sound exposure disturbance might be relatively easy, having no long-term effects on that population. However, for small populations that are constrained by other factors that affect survival, such as food availability, recovery from sound exposure disturbance that affects foraging success, for example, might not be as easy.

Related to these considerations is the area affected versus the habitat available (Richardson et al. 1995). The core habitat of Southern Residents is particularly concentrated on the west side of San Juan Island during the summer (Hauser 2006), and it is considered a central area for feeding (Baird and Hanson 2004, Hanson et al. in prep.). This is also an area where few or no parts of the range are absent of human-produced noise. Broadband received levels (0.1–15 kHz) were more than 120 dB re 1 μPa approximately 50% of the time during summer days in 2005 in this area due to vessel traffic (Veirs and Veirs 2005). This broadband level has been used to estimate zones of behavioral responses in resident killer whales without direct empirical evidence (Erbe 2002).

Assuming these measurements are representative of summer exposure levels on a regular basis and that Southern Residents would show a behavioral response to these exposure levels, then it is likely that behavioral effects of noise exposure would be manifested on a regular basis. However, these broadband SPLs reported by Veirs and Veirs (2005) do not include the frequency range of best hearing sensitivity of killer whales (18–42 kHz, Syzmanski et al. 1999).

Extent of Masking Effects

The masking effects of sound exposure on communication signals (pulsed calls) are likely to extend up to several kilometers away in some conditions. While whale watching vessels may shut down when within a few hundred meters of the whales, they commonly motor around the area adjacent to the whales and within the zone of masking when arriving, repositioning, or exiting the area. Other vessels, such as larger ships and fishing vessels, as well as sound from natural sources contribute to particularly high levels of ambient noise in SRKW summer habitat. Thus masking of communication signals is probably a consistent challenge for Southern Residents. Pulsed calls are most often heard while killer whales are foraging and traveling (Ford 1989, Miller 2002). Some individuals, particularly females, share prey items (Ford and Ellis 2006) and calls used for communication during cooperative foraging could be masked by a variety of sound sources.

The levels of many anthropogenic sounds, such as vessel traffic, are strongest at frequencies below 10 kHz (Figure 5). A common assumption is that masking of echolocation signals is not much of a concern compared to communication signals because echolocation signals are strongest above 20 kHz (Au et al. 2004). However, killer whale hearing is tuned to higher frequencies and whales probably need to hear faint echoes from ensonified objects. Thus any sounds that increase background levels at higher frequencies above ambient levels have the potential to decrease the range at which echolocation activities are effective, including those associated with foraging.

The analyses of active space of echolocation signals in the presence of both natural and anthropogenic sources clearly illustrate potential impacts in terms of the reductions in the range at which killer whales can detect prey items. The fact that whale watching vessels idling at 200 m produced source spectral levels that were comparable to ambient levels (Hildebrand et al. 2006) suggests that an idling vessel at this distance alone would not reduce the active space of echolocation signals. However, this assumption needs to be investigated further. According to the “Be Whale Wise” guidelines, 400 m is the shortest distance where vessels can power up to normal cruising speeds relative to whales. At this range, vessel noise reduced the horizontal detection range of a 50 kHz echolocation signal by as much as 360 m (from 400 to 40 m) relative to ambient conditions. The distance at which noise from vessel operations would be reduced to ambient is considerably larger than 400 m for the vessel measured by Hildebrand et al. (2006). Additional noise level measurements of vessels of different propulsion types operating at slower speeds (less than 7 knots) in SRKW habitat are needed to evaluate potential masking effects further, particularly those that include high frequency measurements.

Most studies that aim to identify biological effects of noise exposure do not consider cumulative acoustic effects. Even if only a very small number of individuals are exposed to sounds that have the potential for auditory injury, other individuals in the area might still be affected by masking or behavioral disturbance, given the larger range at which a sound source would have an impact. Thus the assessment of sound exposure impacts should not ignore less severe events. For example, if a severity score is calculated based on models used for determining the Potential for Biological Removal, the scores must reflect the cumulative effects to determine total number of individuals affects and the effects on the population should be determined from this total number (NRC 2005).

Furthermore, there are other auditory processes besides detection that are affected by noise exposure such as sound localization, discrimination, and recognition. For example, if a killer whale detects the call of a conspecific but cannot localize or recognize the call, then the function of the call for social cohesion may be impaired. There are very few studies on the effects of noise on these types of auditory processes in marine mammals. Nonetheless, these types of impacts should also be considered.

Interaction with Nonacoustic Variables

Sound exposure effects might also interact with nonacoustic variables that have been identified as possible factors related to the population decline of SRKWs. For example, when they are within 100–400 m of vessels, there is evidence that they switch from foraging behaviors to other behaviors such as traveling (Bain et al. 2006). It is unknown whether this behavioral change is in response to the presence of the vessels, to the noise produced by vessels, to other variables, or some combination of these factors. In addition, most vessels, such as those used for whale watching, operate during the day and there is evidence that foraging behavior is also diurnally dependent (Baird et al. 2005). Thus Southern Residents might be temporally and spatially restricted for activities related to foraging. There are cases in which individuals including those with dependent offspring have physical signs of poor body condition before they disappear and are assumed to have died. These observations indicate that starvation is a threat.

Many marine mammals go through periods when food is less abundant. Entering such periods with insufficient body reserves can have significant biological effects at the level of the individual as well as for an endangered population. Acoustic variables that affect caloric intake in SRKWs would affect population recovery. Other anthropogenic effects might also interact with the metabolic effects of periods of low caloric intake such as the mobilization of lipophilic contaminants. For example, fasting northern elephant seals (*Mirounga angustirostris*) showed higher levels of mobilized polychlorinated biphenyls (PCBs) into the bloodstream during the end of their postweaning fast compared to before the fast. Additionally, PCB concentrations in the blood were higher in leaner animals, suggesting that they might be more at risk to potential toxic effects (Debier et al. 2006). The interaction of acoustic effects with other threats on the SRKW population is of significant concern. However, it is extremely difficult to link potentially small but consistent changes in response to noise exposure to population level effects.

Recommendations for Future Work

Future studies that would increase confidence in predicting potential acoustic impacts on SRKWs include those that accurately measure source characteristics and sound propagation effects, particularly of anthropogenic sounds in core habitat. For example, knowing the source level, frequency content, and typical duration of exposure can enable better predictions of what the received levels might be at the whales' location. Because the propagation of vessel sounds is not equal in all directions, a study that measured propagation from the bow, sides, and stern of the vessel would be able to determine if received levels are higher for a particular orientation and how this might interact with directional hearing and the ability to detect signals in noise. Such information could be used to mitigate killer whale and whale watching vessel interactions, particularly with respect to maneuvering around the whales.

Killer whale hearing sensitivity has been described for frequencies between 1 and 100 kHz. The whales will likely hear a sound if received levels are above the hearing threshold of the animal and if background levels are sufficiently low enough to allow it. The assessment of the zones of audibility and masking are contingent on knowledge of the auditory bandwidths of killer whales. As a first approach, one-third octave levels could be used but direct measurements of critical bandwidths across a wide range of frequencies within the hearing range is necessary to assess the validity of using one-third octave band analysis to describe sound sources and assess their impacts. In the interim, it may be useful to bracket the potential effects by using an upper and lower range of probable critical bandwidths of killer whales, such as one-third and one-twelfth octave analyses (e.g., see Southall 2003b). For these reasons, studies that include noise level measurements in SRKW habitat should report sound pressure levels in broadband, one-third octave band, one-twelfth octave band, and spectral density levels whenever possible.

Masking Effect Assessment

Further assessment of masking effects on echolocation signals would be useful, particularly for more types of whale watching vessels as well as other vessels in core habitat. Because the energy of most noise produced by vessels is concentrated at the lower frequencies, it is tempting to assume that the amount of noise generated at the higher frequencies will not affect killer whales. This assumption might be based in part on an anthropomorphic bias, since humans do not hear functionally above 20 kHz yet laboratory evidence shows that killer whales are most sensitive at and above this frequency. It is necessary to consider the whales' subjective perceptual world (the "Umwelt") that they experience.

Currently, there are very few available measurements of ambient noise levels above 20 kHz (see Hildebrand et al. 2006). The existing data suggests that maintaining slow cruise speeds within 400 m of the whales and shutting vessel engines down completely when within 100 m of a whale is sufficient to reduce the effects of masking of echolocation signals used for foraging. Predicting zones of masking will depend on the vessel type, operating speed, and propulsion type as well as the number of vessels actively motoring. Whale watching vessels operating at normal cruise speeds even beyond 400 m are predicted to reduce the active space of the 50 kHz echolocation signal, but there is difficulty in interpreting appropriate detection ranges. Further information about noise levels above 20 kHz for vessels of different sizes, propulsion types, and operating speeds, particularly for speeds less than 7 knots, would be useful to further assess the impacts of anthropogenic sounds on killer whales and how regulatory guidelines such as those based on "Be Whale Wise" may or may not be effective in reducing such potential impacts on animals using higher frequency biosonar for prey detection.

Information about the target strength of key prey items such as various Pacific salmon species using more realistic signals as well as investigations on predator-prey interactions in free-ranging killer whales are needed to predict the potential masking impacts of anthropogenic sources on echolocation and foraging behavior. Studies that focus on foraging efficiency and sound production associated with foraging in the presence and absence of anthropogenic noise would provide additional information that could be used for this assessment.

Behavioral Response

Predicting the zone of responsiveness is dependent on knowing the quantitative aspects of the exposure relative to the whales. As is the case for most cetacean species (see Nowacek et al. 2007), there is either a lack of information altogether or too broad a range of received levels that are potentially correlated with behavioral responses to provide confidence regarding what future exposure might result in behavioral avoidance or displacement in SRKWs. A broadband received level of 120 dB re 1 μPa without respect to frequency content and the sensitivity of the animals has sometimes been used as a generalized rule of thumb to define the onset of a behavioral response to noise (e.g., Erbe 2002). Two-second SPL averages were greater than or equal to 120 dB (re 1 μPa measured from 0.1 to 15 kHz) 50% of the time during summer days in Haro Strait (Veirs and Veirs 2005).

One interpretation is that killer whales would potentially behaviorally or physiologically respond to sounds in the areas associated with summer foraging at least 50% of the time. This assumption requires further empirical evaluation, particularly since killer whales are most sensitive at frequencies between 18 and 42 kHz. Studies that investigate behavioral responses to vessel traffic should at the least report received sound levels, if not source levels, and preferably sound exposure levels to address the temporal aspects of the exposure. Moreover, it is unknown whether behavioral responses to vessel effects result from the presence of the vessel, the sounds it produces, or some combination of these factors. Future studies that tease apart these effects, such as controlled exposure experiments in which recordings of the vessel sounds are played back in the absence of the vessel to determine behavioral changes, are also needed to make more accurate predictions of the zone of responsiveness.

Likelihood of Auditory Injury

Determining the likelihood of auditory injury is probably one of the more imperative assessments of sound exposure in SRKWs. Even if the potential for such exposure is low relative to other possible acoustic risk factors, the uncertainty in predicting this type of auditory effect in an ESA-listed species warrants further research efforts. Auditory injury is usually estimated from studies on TTS because inflicting permanent hearing loss in marine mammals is a significant violation of the Marine Mammal Protection Act of 1972. No studies have measured TTS in killer whales but a few well documented studies have measured TTS in bottlenose dolphins and beluga whales. From these studies, TTS onset has been estimated at a sound exposure level of 195 dB re 1 $\mu Pa^2 s$ for midrange exposure and test frequencies. Given the sensation level differences between killer whales, bottlenose dolphins, and beluga whales at these frequencies, this onset SEL criterion might not be conservative enough to prevent auditory injury in killer whales. TTS studies in killer whales would provide direct evidence to support or refute these assumptions.

Research focused on assessing potential population effects of sound exposure is probably the most important (albeit most challenging) research priority. Dose-response curves, in which the percentage of individuals that show responses to sound exposure such as behavioral disturbance or onset-TTS as a function of sound exposure level, are needed to make accurate assessments at the level of the population (NRC 2005). Additionally, studies that measure equal loudness contours in killer whales would be extremely useful to determine weighting functions that could be used when measuring ambient noise in order to better predict behavioral and auditory disturbance. An approximation to this would be to use a weighting function based on the killer whale audiogram.

Finally, there is a need for more information in published, peer-reviewed formats. For example, a working group is in the process of publishing noise exposure criteria for marine mammals that includes the development of interim weighting functions for different groups. These groups are categorized based on hearing abilities given the variety of auditory sensitivities at different frequencies across the different marine mammal taxa (Southall et al. 2007). Such documents would provide tools for policy makers and natural resource managers to make more confident decisions about potential acoustic impacts from human activity on SRKWs and other marine mammal populations.

Conclusions

This report offers the following conclusions on sound exposure and SRKWs:

- Given the chronic nature of vessel traffic in areas designated as critical habitat of SRKWs and the potential for vessel noise to mask communication and echolocation signals of SRKWs, more measurements of these sounds within the relevant frequency range of killer whale hearing (i.e., 1–100 kHz) are needed. Such studies should further investigate the effects of vessel size, propulsion type, operating speed, and vessel orientation on the sound levels emitted.
- Some vessel noise, depending on the vessel size, propulsion type, and operating speed, was predicted to significantly reduce the range at which echolocating killer whales could detect salmon in the water column. Interpretations of these effects are limited by a dearth of information on predator-prey interactions, such as the typical ranges at which killer whales detect and pursue salmon in quiet conditions. Studies that elucidate this information would enable a more accurate assessment of the impacts of vessel noise as well as other anthropogenic sources on the foraging success of echolocating killer whales.
- The ability to assess zones of audibility and masking are dependent on knowing the appropriate bandwidth to quantify anthropogenic sounds. The effective filter bandwidth of the killer whale auditory system is unknown and has been estimated through indirect methods. However, these methods are often not accurate compared to direct measurements. Direct measurements of killer whale auditory filter bandwidths are needed to provide more confidence in assessing impacts of anthropogenic sounds.
- Assessment of auditory injury in killer whales is estimated through surrogate species such as the bottlenose dolphin. However, differences in hearing sensitivity between the two species may result in inaccurate assessments of TTS in killer whales. Investigations that measure TTS directly in killer whales would provide very valuable data to assess the potential of auditory injury from sound exposure.
- Predicting the zone of responsiveness is dependent on knowing the quantitative aspects of the exposure relative to the whales. There is either a lack of or too broad a range of received levels that are potentially correlated with behavioral responses to provide confidence regarding what future exposure might result in behavioral effects in SRKWs. Investigations that quantify both the received levels of the sound exposure (using appropriate metrics and within the appropriate frequency range) and the behavioral response of free-ranging killer whales are needed to assess zones of responsiveness. These studies should ideally measure the proportion of animals affected from controlled exposures.

References

Ainslie, M. A., and J. G. McColm. 1998. A simplified formula for viscous and chemical absorption in sea water. J. Acoust. Soc. Am. 103:1671–1672.

Au, W. W. L., and P. W. B. Moore. 1984. Receiving beam patterns and directivity indices of the Atlantic bottlenose dolphin, *Tursiops truncatus*. J. Acoust. Soc. Am. 75:255–258.

Au, W. W. L., and P. W. B. Moore. 1990. Critical ratio and critical bandwidth for the Atlantic bottlenose dolphin. J. Acoust. Soc. Am. 88:1635–1638.

Au, W. W. L., D. A. Carder, R. H. Penner, and B. L. Scronce. 1985. Demonstration of adaptation in beluga whales echolocation signals. J. Acoust. Soc. Am. 77:726–730.

Au, W. W. L., R. H. Penner, and J. Kadane. 1982. Acoustic behavior of echolocating Atlantic bottlenose dolphins. J. Acoust. Soc. Am. 71:1269–1275.

Au, W. W. L., P. E. Nachtigall, and J. L. Pawlowski. 1999. Temporary threshold shift in hearing induced by an octave band of continuous noise in the bottlenose dolphin. J. Acoust. Soc. Am. 106:2251(A).

Au, W. W. L., J. K. B. Ford, J. K. Horne, and K. A. Newman Allman. 2004. Echolocation signals of free-ranging killer whales (*Orcinus orca*) and modeling of foraging for Chinook salmon (*Oncorhynchus tshawytscha*). J. Acoust. Soc. Am. 115:901–909.

Au, W. W. L., K. J. Benoit-Bird, and R. A. Kastelein. 2007. Modeling the detection range of fish by echolocating bottlenose dolphins and harbor porpoises. J. Acoust. Soc. Am. 121:3954–3962.

Awbry, F. T., J. A. Thomas, and R. A. Kastelein. 1988. Low frequency underwater hearing sensitivity in belugas (*Delphinapterus leucas*). J. Acoust. Soc. Am. 84:2273–2275.

Bain, D. E., and M. E. Dahlheim. 1994. Effects of masking noise on detection thresholds of killer whales. *In* T. R. Loughlin (ed.), Marine mammals and the *Exxon Valdez*, p. 243–257. Academic Press, San Diego, CA.

Bain, D. E., R. Williams, J. C. Smith, and D. Lusseau. 2006. Effects of vessels on behavior of Southern Resident killer whales (*Orcinus* spp.) 2003–2005. NMFS Contract AB133F05SE3965. (Available from D. E. Bain, Friday Harbor Laboratories, Univ. Washington, 620 University Road, Friday Harbor, WA 98250.)

Baird, R. W., M. B. Hanson, and L. M. Dill. 2005. Factors influencing the diving behaviour of fish-eating killer whales: Sex differences and diel and interannual variation in diving rates. Can. J. Zool. 83:257–267.

Baird R. W., and M. B. Hanson. 2004. Diet studies of Southern Resident killer whales: Prey sampling and behavioral cues of predation. NMFS Contract AB133F-03-SE-1070. (Available from Cascadia Research Collective, 218½ W. Fourth Ave., Olympia, WA 98501.)

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

Brumm, H., and H. Slabbekoorn. 2005. Acoustic communication in noise. *In* P. J. B. Slater, C. T. Snowden, H. J. Borckmann, T. J. Roper, and M. Naguib (eds.), Advances in the study of behavior, Vol. 35, p. 151–207. Academic Press, San Diego, CA.

Crum, L. A., and Y. Mao. 1996. Acoustically enhanced bubble growth at low frequencies and its implications for human diver and marine mammal safety. J. Acoust. Soc. Am. 99:2898–2907.

Debier, C., C. Chalon, B. J. Le Boeuf, T. deTillesse, Y. Larondelle, and J. Thome. 2006. Mobilization of PCBs from blubber to blood in northern elephant seals (*Mirounga angustirostris*) during the post-weaning fast. Aquat. Toxicol. 80:149–157.

Deecke, V. B., J. K. B. Ford, and P. J. B. Slater. 2004. The vocal behaviour of mammal-eating killer whales: Communicating with costly calls. Anim. Behav. 69:395–405.

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.

Erbe, C. 2002. Underwater noise of whale-watching boats and potential effects on killer whales (*Orcinus orca*), based on an acoustic impact model. Mar. Mamm. 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. Am. 108:1327–1331.

Fay, R. R. 1988. Hearing in vertebrates: A psychophysical databook, Hill-Fay Associates, Winnetka, IL.

Fernández, A., M. Arbelo, R. Deaville, I. A. P. Patterson, P. Castro, J. R. Baker, E. Degollada, H. M. Ross, P. Herráez, A. M. Pocknell, E. Rodríguez, 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). Nature 428 (15 April 2004).

Fernandez, A., J. F. Edwards, F. Rodriguez, A. Espinosa de los Monteros, P. Herraez, P. Castro, J. R. Jaber, V. Martin, and M. Arbelo. 2005. "Gas and fat embolic syndrome" involving a mass stranding of beaked whales (family Ziphiidae) exposed to anthropogenic sonar signals. Vet. Pathol. 42:446–457.

Finneran, J. J., C. E. Schlundt, D. A. Carder, J. A. Clark, J.A. Young, J.B. Gaspin, and S. H. Ridgway. 2000. Auditory and behavioral responses of bottlenose dolphins (*Tursiops truncatus)* and a beluga (*Delphinapterus leucas*) to impulsive sounds resembling distant signatures of underwater explosions. J. Acoust. Soc. Am. 108:417–431.

Finneran, J. J., C. E. Schlundt, D. A. Carder, and S. H. Ridgway. 2002a. Auditory filter shapes for the bottlenose dolphin (*Tursiops truncatus)* and the white whale (*Delphinapterus leucas*) derived with notched noise. J. Acoust. Soc. Am. 112:322–328.

Finneran, J. J., C. E. Schlundt, C.E. Dear, D. A. Carder, and S. H. Ridgway. 2002b. Temporary shift in masked hearing thresholds in odontocetes after exposure to single underwater impulses from a seismic water gun. J. Acoust. Soc. Am. 111:2929–2940.

Finneran, J. J., D. A. Carder, C. E. Schlundt, and S. H. Ridgway. 2005. Temporary threshold shift in bottlenose dolphins (*Tursiops truncatus)* exposed to mid-frequency tones. J. Acoust. Soc. Am. 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 truncates*) using multiple simultaneous auditory evoked potentials. J. Acoust. Soc. Am. 122:1249–1264.

Fletcher, H. 1940. Auditory patterns. Rev. Mod. Phys. 12:47–65.

Foote, A. D. 2005. Correlates of variability in killer whale stereotyped call repertoires. Master's thesis. University of Durham, Durham, UK.

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

Ford, J. K. B. 1989. Acoustic behavior of resident killer whales (*Orcinus orca*) off Vancouver Island, British Columbia. Can. J. Zool. 67:727–745.

Ford, J. K. B. 1991. Vocal traditions among resident killer whales (*Orcinus orca*) in coastal waters of British Columbia. Can. J. Zool. 69:1454–1483.

Ford, J. K. B., and G. M. Ellis. 1999. Transients: Mammal-hunting killer whales of British Columbia, Washington, and southeastern Alaska. University of British Columbia Press, Vancouver.

Ford, J. K. B., and G. M. Ellis. 2006. Selective foraging by fishing-eating killer whales (*Orcinus orca*) in British Columbia. Mar. Ecol. Prog. Ser. 316:185–199.

Ford, J. K. B., G. M. Ellis, and K. C. Balcomb. 2000. Killer Whales, 2nd edition. University of British Columbia Press, Vancouver.

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

Hanson, M. B., R. W. Baird, C. Emmons, J. Hempelmann, G. S. Schorr, J. Sneva, D. Van Doornik, K. Ayres, S. K. Wasser, K. C. Balcomb, K. Balcomb-Baratok, and M. J. Ford. In prep. Species and stock identification of prey selected by endangered "Southern Resident" killer whales in their summer range. (Available from M. B. Hanson, Northwest Fisheries Science Center, 2725 Montlake Blvd. E., Seattle, WA 98112.)

Hauser, D. D. W. 2006. Summer space use of Southern Resident killer whales (*Orcinus orca*) within Washington and British Columbia inshore waters. Master's thesis. Univ. Washington, Seattle.

Hildebrand, J., M. McDonald, J. Calambokidis, and K. Balcomb. 2006. Whale watch vessel ambient noise in the Haro Strait. NMFS Contract NA17RJ1231. (Available from Univ. California San Diego, Scripps Institution of Oceanography, Joint Institute for Marine Observations, 9500 Gilman Dr., La Jolla, CA 92093.)

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. Pocknell, F. Rodriguez, F. E. Howie, 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, C. S. 1967. Sound detection thresholds in marine mammals. *In* W. N. Tavolga (ed.), Marine bioacoustics, Vol. 2, p. 247–260. Pergamon, Oxford.

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

Johnson, C. S. 1991. Hearing thresholds for a 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.

Jones, C. D., and M. A. Wolfson. 2006. Acoustic environment of the Haro Strait: Preliminary propagation modeling and data analysis. APL-UW Tech. Memo. 03-06. Univ. Washington, Applied Physics Laboratory, Seattle.

Kastak, D., B. L. Southall, R. J. Schusterman, and C. Reichmuth Kastak. 2005. Underwater temporary threshold shift in pinnipeds: Effects of noise level and duration. J. Acoust. Soc. Am. 118:3154–3163.

Krahn, M. M., M. J. Ford, W. F. Perrin, P. R. Wade, R. P. Angliss, M. B. Hanson, B. L. Taylor, G. M. Ylitalo, M. E. Dahlheim, J. E. Stein, and R. S. Waples. 2004. Status review of Southern Resident killer whales (*Orcinus orca*) under the Endangered Species Act. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-62.

Lauglin, J. 2005. Underwater sound levels associated with restoration of the Friday Harbor ferry terminal. Washington Dept. Transportation, Seattle.

Leatherwood, J. D., B. M. Sullivan, K. P. Shepherd, D. A. McCurdy, and S. A. Brown. 2002. Summary of recent NASA studies of human response to sonic boom. J. Acoust. Soc. Am. 111:586–598.

Lemonds, D. W., W. W. L. Au, P. E. Nachtigall, and H. L. Roitblat. 2000. High-frequency auditory filter shapes in an Atlantic bottlenose dolphin. J. Acoust. Soc. Am. 108:2614(A).

Lesage, V., C. Barrette, M. C. Kingsley, and B. Sjare. 1999. The effect of vessel noise on the vocal behavior of belugas in the St. Lawrence River estuary, Canada. Mar. Mamm. Sci. 15:65–84.

Lombard, E. 1911. Le signe de l'elevation de la voix. Ann. Mal. Oreille Larynx 37:101–119.

Madsen, P. T. 2005. Marine mammals and noise: Problems with root mean square sound pressure levels for transients. J. Acoust. Soc. Am. 117:3952–3957.

Miller, J. H., A. E. Bowles, B. L. Southall, R. L. Gentry, W. T. Ellison, J. J. Finneran, C. R. Greene Jr., D. Kastak, D. R. Ketten, P. L. Tyack, P. E. Nachtigall, W. J. Richardson, and J. A. Thomas. 2005. Strategies for weighting exposure in the development of acoustic criteria for marine mammals. J. Acoust. Soc. Am. 118:2019(A).

Miller, P. J. O. 2002. Mixed-directionality of killer whale stereotyped calls: A direction of movement cue? Behav. Ecol. Sociobiol. 52:262–270.

Miller, P. J. O. 2006. Diversity in sound pressure levels and estimated active space of resident killer whale vocalizations. J. Comp. Physiol. A. 192:449–459.

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

Mossbridge, J. A., and J. A. Thomas. 1999. An "acoustic niche" for Antarctic killer whale and leopard seal sounds. Mar. Mamm. Sci. 15:1351–1357.

Nachtigall, P. E., J. L. Pawloski, and W. W. L. Au. 2003. Temporary threshold shifts and recovery following noise exposure in the Atlantic bottlenosed dolphin (*Tursiops truncatus*). J. Acoust. Soc. Am. 113:3425–3429.

Nachtigall, P.E., A. Y. Supin, J. L. Pawloski, and W. W. L. Au. 2004. Temporary threshold shifts after noise exposure in the bottlenose dolphin (*Tursiops truncatus*) measured using evoked auditory potentials. Mar. Mamm. Sci. 20:673–687.

NMFS (National Marine Fisheries Services). 2005a. Assessment of acoustic exposures on marine mammals in conjunction with USS *Shoup* active sonar transmissions in the eastern Strait of Juan de Fuca and Haro Strait, Washington, 5 May 2003. NMFS Office of Protected Resources, Silver Spring, MD.

NMFS (National Marine Fisheries Services). 2005b. Endangered and threatened wildlife and plants: Endangered status for Southern Resident killer whales. Federal Register [Docket No. 041213348-5285-02, 18 November 2005] 70(217):69903–69912.

NMFS (National Marine Fisheries Services). 2006. Endangered and threatened species; designation of critical habitat for Southern Resident killer whales. Federal Register [Docket No. 060228057-6283-02, 29 November 2006] 71(229):69054–69070.

NRC (National Research Council). 2003. Ocean noise and marine mammals. The National Academies Press, Washington, DC.

NRC (National Research Council). 2005. Marine mammals populations and ocean noise. The National Academies Press, Washington, DC.

Nowacek, D. P., L. H. Thorbe, D. W. Johnstone, and P. L. Tyack. 2007. Response of cetaceans to anthropogenic noise. Mamm. Rev. 37:81–115.

Nystuen, J. A. 2006. Marine mammals monitoring for NW Fisheries. NMFS Contract N00024-02-D-6602. Univ. Washington, Applied Physics Laboratory, Seattle.

Nystuen, J. A., C. C. McGlothin, and M. S. Cook. 1993. The underwater sound generated by heavy rainfall. J. Acoust. Soc. Am. 93:3169–3177.

Olesiuk, P. F., M. A. Bigg, and G. M. Ellis. 1990. Life history and population dynamics of resident killer whales (*Orcinus orca*) in the coastal waters of British Columbia and Washington State. Rep. Int. Whaling Comm., Spec. Issue 12:209–243.

Piantadosi, C. A., and E. D. Thalmann. 2004. Pathology: Whales, sonar, and decompression sickness. Nature 428 (15 Apr 2004).

Plomp, R., and M. A. Bouman. 1959. Relation between hearing threshold and duration of tone pulses. J. Acoust. Soc. Am. 31:749–758.

Renaud, D. L., and A. Popper. 1975. Sound localization by the bottlenose porpoise (*Tursiops truncatus*). J. Exp. Biol. 63:569–585.

Renouf, D. 1991. Sensory reception and processing in Phocidae and Otariidae. *In* D. Renouf (ed.), The behaviour of pinnipeds, p. 345–394. Chapman and Hall, London.

Richardson W. J., C. R. Greene, C. I. Malme, and D. H. Thomson. 1995. Marine mammals and noise. Academic Press, San Diego, CA.

Romano, T. A., M. J. Keogh, C. Kelly, P. Feng, L. Berk, C. E. Schlundt, D. A. Carder, and J. J. Finneran. 2004. Anthropogenic sound and marine mammal health: Measures of the nervous and immune systems before and after intense sound exposure. Can. J. Fish. Aquat. Sci. 61:1124–1134.

Scheifele, P. M., S. Andrew, R. A. Cooper, M. Darre, F. E. Musiek, and L. Max. 2005. Indication of a Lombard vocal response in the St. Lawrence River beluga. J. Acoust. Soc. Am. 117:1486–1246.

Schevill, W. E., and W. A. Watkins. 1966. Sound structure and directionality in *Orcinus* (killer whale). Zoologica 51:70–76.

Schlundt, C. E., R. L. Dear, D. A. Carder, and J. J. Finneran. 2006. Growth and recovery of temporary threshold shifts in a dolphin exposed to mid-frequency tones with durations up to 128 s. J. Acoust. Soc. Am. 120:3227(A).

Schlundt, C. E., J. J. Finneran, D. A. Carder, and S. H. Ridgway. 2000. Temporary shift in masked hearing thresholds of bottlenose dolphins, *Tursiops truncatus*, and white whales, *Delphinapterus leucas*, after exposure to intense tones. J. Acoust. Soc. Am. 107:3496–3508.

Schusterman, R. J. 1974. Low false-alarm rates in signal detection by marine mammals. J. Acoust. Soc. Am. 55:845–848.

Sharf, B. 1970. Critical bands, *In* J. V. Tobias (ed.), Foundations of modern auditory theory, Vol. I, p. 159–202. Academic Press, New York.

Simon, M., M. Wahlberg, and L. A. Miller. 2007. Echolocation clicks from killer whales (*Orcinus orca*) feeding on herring (*Clupea harengus*). J. Acoust. Soc. Am. 121:749–752.

Southall, B. L., R. J. Schusterman, and D. Kastak. 2003a. Auditory masking in three pinnipeds: Aerial critical ratios and direct critical bandwidth measurements. J. Acoust. Soc. Am. 114:1660–1666.

Southall, B. L., R. J. Schusterman, and D. Kastak. 2003b. Acoustic communication ranges for northern elephant seals (*Mirounga angustirostris*). Aquat. Mamm. 29:202–213.

Southall, B. L., A. E. Bowles, W. T. Ellison, J. J. Finneran, R. L. Gentry, C. R. Green 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. 33:414-521.

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. Acoust. Soc. Am. 106:1134–1141.

Thomas, J. A., R. A. Kastelein, and F. T. Awbrey. 1990. Behavior and blood catecholamines of captive belugas during playbacks of noise from an oil drilling platform. Zoo Biol. 9:393–402.

Thomsen, F., D. Franck, and J. K. B. Ford. 2001. Characteristics of whistles from the acoustic repertoire of resident killer whales (*Orcinus orca*) off Vancouver Island, British Columbia. J. Acoust. Soc. Am. 109:1240–1246.

Thomsen, F., D. Franck, and J. K. B. Ford. 2002. On the communicative significance of whistles in wild killer whales (*Orcinus orca*). Naturwissenschaften 89:404–407.

Thompson, R. K. R., and L. M. Herman. 1975. Underwater frequency discrimination in the bottlenosed dolphin (1–140 kHz) and the human (1–8 kHz). J. Acoust. Soc. Am. 57:943–948.

Urick, R. J. 1983. Principles of underwater sound, 3rd edition. McGraw-Hill, New York.

U.S. Navy, Pacific Fleet. 2004. Report on the results of the inquiry into allegations of marine mammal impacts surrounding the use of active sonar by USS *Shoup* (DDG 86) in the Haro Strait on or about 5 May 2003. Online at: http://www.acousticecology.org/docs/SHOUPNavyReport0204.pdf [accessed 24 January 2008].

Veirs, V., and S. Veirs. 2005. Average levels and power spectra of ambient sound in the habitat of Southern Resident orcas. Report to NOAA/NMFS/NWFSC. Online at http://www.coloradocollege.edu/dept/ev/Research/Faculty/OVALItems/pdf_Papers/051204noaa-haro_noise_final.pdf [accessed 23 January 2008].

Ward, W. D. 1997. Effects of high-intensity sound. *In* M. J. Crocker (ed.), Encyclopedia of acoustics, p. 1497–1507. John Wiley and Sons, New York.

Wenz, G. M. 1962. Acoustic ambient noise in the ocean: Spectra and sources. J. Acoust. Soc. Am. 34:1936–1956.

White Jr., M. J., J. Norris, D. Ljungblad, K. Baron, and G. di Sciara. 1977. Auditory thresholds of two beluga whales (*Delphinapterus leucas*). Tech. Rep. 78-108. Hubbs Sea World Research Institute, San Diego, CA.

Williams, R., and E. Ashe. 2006. Northern Resident killer whale response to vessels varied with number of boats. NMFS Contract AB133F04SE0736. (Available from R. Williams, Pearse Island, Box 193, Alert Bay, BC V0N1A0, or E. Ashe, 2103 N. 54th St., Seattle, WA 98103.)

Williams, R., D. Lusseau, and P. S. Hammond. 2006. Estimating relative energetic costs of humans disturbance to killer whales (*Orcinus orca*). Biol. Conserv. 133:301–311.

Wursig, B., C. R. Greene Jr., and T. A. Jefferson. 2000. Development of an air bubble curtain to reduce underwater noise of percussive piling. Mar. Environ. Res. 49:79–93.

Yost, W. A. 2000. Fundamentals of hearing, 4th edition. Academic Press, San Diego, CA.

Yost, W. A., and W. Shofner. 2005. The critical band and the critical ratio. J. Acoust. Soc. Am. 117:2582(A).

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