APPENDIX D
ACOUSTIC AND EXPLOSIVE IMPACTS SUPPORTING INFORMATION

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Draft

Supplemental Environmental Impact Statement/ Overseas Environmental Impact Statement Atlantic Fleet Training and Testing

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D ACOUSTIC AND EXPLOSIVE IMPACTS SUPPORTING INFORMATION

D.1 ACOUSTIC AND EXPLOSIVE CONCEPTS/PRIMER

This appendix briefly explains the transmission of sound and explosive energy underwater and in air; introduces some of the basic mathematical formulas used to describe propagation; and defines acoustical terms, abbreviations, and units of measurement. Methods used to analyze hearing are also described.

For a more extensive background on acoustics, explosives, and marine bioacoustics the following resources are recommended:

- *Marine Mammals and Noise* (Richardson et al., 1995b)
- *Principles of Underwater Sound* (Urick, 1983)
- *Fundamentals of Acoustical Oceanography* (Medwin & Clay, 1998)
- *Principles of Marine Bioacoustics* (Au & Hastings, 2008)
- *Exploring Animal Behavior Through Sound: Volume 1 Methods* (Erbe & Thomas, 2022)
- *Discovery of Sound in the Sea* (https://dosits.org/)

D.1.1 TERMINOLOGY

The following terms are used in this document when discussing sound and the attributes of a sound source.

D.1.1.1 Sound

Sound is produced when an elastic medium (such as air or water) is set into motion, typically by a vibrating object within the medium. As the object vibrates, its motion is transmitted to adjacent "particles" of the medium. The motion of these particles is transmitted to adjacent particles, and so on. The result is a mechanical disturbance (the "sound wave") that moves away from the source and propagates at a medium-dependent speed (the "sound speed"). As the sound wave travels through the medium, the individual particles of the medium oscillate about their original positions but do not actually move with the sound wave. This particle movement creates small changes in the medium's density, pressure, and temperature.

Sound may be described by both physical and subjective attributes. Physical attributes, such as sound amplitude and frequency, may be directly measured. Subjective (or sensory) attributes like loudness depend on an animal's perception of sound, and can vary between species and individuals.

D.1.1.2 Signal Versus Noise

When sound is purposely created to convey information, communicate, or obtain information about the environment, it is often referred to as a signal. Examples of signals include sonar pings, marine mammal vocalizations and echolocation clicks, tones used in hearing experiments, and small sonobuoy explosions used for submarine detection. Typically, signals have some type of known characteristics, for example, they could use a limited set of frequencies, have a specific set of harmonics, or be used such that the pulse context provides information to a receiver.

Noise is defined as any undesired sound (American National Standards Institute, 2013) that typically lacks the clear characteristics previously described. Sounds produced by naval aircraft and vessel propulsion are considered noise because they represent possible inefficiencies within the system and increased detectability by adversaries. Whether a sound is perceived as noise depends on the receiver (i.e., the animal or system that detects the sound). For example, small explosions and sonar pings used to generate sounds to locate enemy submarines produce signals that are useful to sailors engaged in anti-submarine warfare, but are assumed to be noise when detected by marine species.

The combination of all sounds (including signals and noise) at a particular location, whether these sources are located near or far, is defined as ambient noise (American National Standards Institute, 2013). Ambient noise includes natural sources such as sound from crashing waves, rain, and animals (e.g., snapping shrimp), and anthropogenic sources such as seismic surveys and vessel noise. Every location in the marine environment contains some ambient noise, but how much depends on a multitude of factors. Characterizing the ambient noise level of a location is imperative to understanding potential impacts to marine life from anthropogenic sound.

D.1.1.3 Frequency and Wavelength

Frequency is the physical attribute associated with the subjective attribute “pitch”, the higher the frequency, the higher the pitch. Frequency is defined by the number of oscillations (i.e., cycles) in the sound pressure or particle motion per second. One hertz (Hz) is equal to one oscillation per second, and one kilohertz (kHz) is equal to 1,000 oscillations per second. “Bandwidth” refers to the range between the minimum and maximum frequency of a sound source or receiver.

Pure tones have energy at a constant, single frequency. Complex tones contain energy at multiple, discrete frequencies, rather than a single frequency. A harmonic of a sound at a particular frequency is a multiple of that frequency. For example, harmonic frequencies of a 2 kHz fundamental frequency tone (i.e., the lowest and most intense frequency of a complex tone) are 4 kHz, 6 kHz, 8 kHz. A source operating at a nominal frequency may emit several harmonic frequencies, but at lower amplitudes and higher frequencies. Some sources may also emit subharmonics which are lower in frequency than the fundamental frequency; however, these are typically many orders of magnitude less powerful than the fundamental frequency. Sounds with large bandwidths (“broadband” sounds) have energy spread across many frequencies.

In this document, sounds are generally described as either low- (less than 1 kHz), mid- (1 kHz to 10 kHz), high- (10 kHz to 100 kHz), or very high- (greater than 100 kHz) frequencies. Hearing ranges of marine animals (e.g., fishes, birds, sea turtles, and marine mammals) are variable and species dependent. For example, some fishes can detect sounds below 100 Hz and some species of marine mammals have hearing capabilities that extend above 100 kHz. Therefore, acoustic impact analyses must focus on the sound amplitude (i.e., pressure or particle motion, see Section D.1.1.4, Sound Amplitude), in addition to the sound frequency and animal sensory capabilities.

The wavelength of a sound is the distance between wave peaks. Wavelength decreases as frequency increases. The frequency multiplied by the wavelength equals the speed of sound in a medium, as shown in this equation:

$$\text{sound speed (m/s)} = \text{frequency} \left(\frac{1}{s}\right) \text{x wavelength (m)}$$

The approximate speed of sound in sea water is 1,500 meters per second (m/s) and in air is 340 m/s, although speed varies depending on environmental conditions (e.g., pressure, temperature, and, in the case of sea water, salinity; see Section D.1.3.1, Speed of Sound).

D.1.1.4 Sound Amplitude

Sound amplitude is the physical attribute associated with the subjective attribute loudness. Amplitude is related to the amount that the medium particles oscillate about their original positions and can be thought of as the “strength” of a sound (as the amplitude increases, the loudness also increases). As the sound wave travels, the particles of the medium oscillate and transfer energy from one particle to another but do not actually travel with the wave. The result is a mechanical disturbance (i.e., the sound wave) that propagates energy away from the sound source. Sound amplitude is typically characterized by measuring the acoustic pressure or particle motion.

D.1.1.5 Impulsive Versus Non-Impulsive Sounds

Although no standard definitions exist, sounds may be broadly categorized as impulsive or non-impulsive. Impulsive sounds have short durations, rapid rise-times, broad frequency content, and high peak pressures. Impulsive sounds are often produced by processes involving a rapid release of energy or mechanical impacts (Hamernik & Hsueh, 1991). Explosions and weapons firing are examples of impulsive sound sources analyzed in this document. In contrast, sonar, vessel operation, and underwater transducers lack the characteristics of impulsive sound sources and are thus examples of non-impulsive sound sources. Non-impulsive sounds can be essentially continuous, such as machinery noise, or intermittent, such as sonar pings. Impulsive signals, particularly at close range, are characterized as brief and broadband with rapid rise time and higher instantaneous peak pressure than other signal types. However, because of propagation effects, an impulsive signal can lose those characteristics, and at a variable distance it could be characterized as a non-impulsive signal (Hastie et al., 2019; Martin et al., 2020).

D.1.1.6 Acoustic Impedance

Acoustic impedance is a property of the propagation medium (air, water, sediment, or tissue) that can be simply described as the opposition to the flow of a pressure wave. Acoustic impedance is a function of the density and speed of sound in a medium. Sound transmits more readily through materials of similar acoustic impedance, such as water and animal tissue, since soft tissue is mainly comprised of water. When sound waves encounter a medium with different acoustic impedance (for example, an air-water interface), they reflect and refract (see Sections D.1.3.3.3, Refraction, and D.1.3.3.4, Reflection and Multipath Propagation), creating more complex propagation conditions. For example, sound traveling in air (low impedance) encountering the water surface (high impedance) will be largely reflected, preventing most sound energy in the air from being transmitted into the water. The impedance difference at the tissue-air interface in animals with gas-containing organs also makes these areas susceptible to damage when exposed to the shock wave near an explosion. Transmission from high-impedance to low-impedance can result in large motion at the boundary.

D.1.1.7 Duty Cycle

Duty cycle describes the portion of time that a source generates sound. It is defined as the ratio of time that a signal or system is on compared to the time it is off during an operational period. For example, if a sonar source produces a one-second ping once every 10 seconds, the duty cycle is 10 percent. Duty cycles vary within and between different acoustic sources; in general, a duty cycle of 20 percent or less is considered low, and a duty cycle of 80 percent or higher is considered high.

D.1.1.8 Resonance

Resonance occurs when an object is vibrated at a frequency near its “natural frequency” or resonant frequency. The resonant frequency can be considered the preferred frequency at which an object will oscillate at a greater magnitude than when exposed to other frequencies. In this document, resonance is considered in relation to the size of an air bubble or air cavity (e.g., lungs). Biological life exposed to high pressure waves from an outside source can lead to potential injury. Due to an inverse relationship, the smaller the bubble, the higher the resonant frequency. The natural frequency of biological life would vary based on the size of the bubbles trapped within them. For example, large whale lungs would have a lower resonant frequency than dolphin lungs. The natural frequencies of dolphin and beluga lungs near the surface are about 36 Hz and 30 Hz, respectively (Finneran, 2003). As an animal dives deep within the water column, there is a corresponding increase in pressure. Hence, any air bubbles trapped within the animal would likely shrink as a result of the pressure change (Bostrom et al., 2008). Because of the change in bubble size, the resonant frequencies would tend to increase as an animal dives.

D.1.2 SOUND METRICS

The sound metrics described here are used to quantify exposure to a sound or explosion.

D.1.2.1 Pressure

Sound pressure is the incremental variation in a medium's static pressure (i.e., the ambient pressure without the added sound) as a sound wave travels through it. Sound pressure is typically expressed in units of micropascals (µPa), although explosive overpressure may also be described in pounds per square inch (psi).

Various sound pressure metrics are illustrated in Figure D.1-1 for (a) a non-impulsive sound (a pure tone in this illustration) and (b) an impulsive sound. As shown in Figure D.1-1, the non-impulsive sound has a relatively gradual rise in pressure from static pressure, while the impulsive sound has a near-instantaneous rise to a high peak pressure. The peak pressure shown on both illustrations is the maximum absolute value of the instantaneous sound pressure during a specified time interval ("zero-to-peak" or "peak"). "Peak-to-peak" pressure is the difference between the maximum and minimum sound pressures.

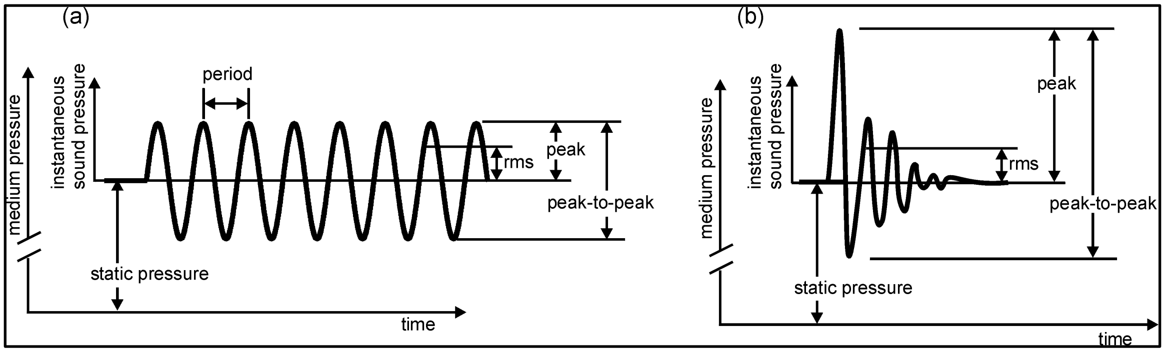


Figure D.1-1: Various Sound Pressure Metrics for a Hypothetical (a) Pure Tone (Non-Impulsive) and (b) Impulsive Sound

The root-mean-square (rms) value is often used to describe the average sound pressure level (SPL). SPLs provided in this Supplemental Environmental Impact Statement (EIS)/Overseas EIS (OEIS) are root-mean-square values unless otherwise specified. As the name suggests, this method takes the square root of the average squared sound pressure values over a time interval. The duration of this time interval can have a strong effect on the measured rms sound pressure for a given sound, especially where pressure levels vary significantly, as during an impulsive sound exposure. If the analysis duration includes a large portion of the waveform after the sound pressure has returned to zero, the rms pressure would be relatively low. If the analysis duration includes only the highest pressures of the impulsive exposure, the rms value would be comparatively high. For this reason, it is important to specify the duration used to calculate the rms pressure for impulsive sounds.

D.1.2.2 Sound Pressure Level

The most common sound level metric is SPL. Because many animals can detect very large pressure ranges and judge the relative loudness of sounds by the ratio of the sound pressures (a logarithmic behavior), SPL is described by taking the logarithm of the ratio of the sound pressure to a reference pressure. Use of a logarithmic scale compresses the wide range of measured pressure values into a more useful scale.

SPLs are normally expressed in decibels. A decibel is 1/10 of a bel, a unit of level when the logarithm is to the base ten and the quantities concerned are proportional to power (American National Standards Institute, 2013). SPL in decibels is calculated as follows:

$$SPL = 20\log_{10}\left(\frac{P}{P_{ref}}\right)$$

where P is the sound pressure and P_{ref} is the reference pressure. Unless stated otherwise, the pressure (P) is the rms value of the pressure (American National Standards Institute, 2013). In some situations, SPL is calculated for the peak pressure rather than the rms pressure. On the occasions when rms pressure is not used, the pressure metric will be stated (e.g., peak SPL means an SPL calculated using the peak pressure rather than the rms pressure).

When a value is presented in decibels, it is important to also specify the value and units of the reference quantity. Normally the numeric value is given, followed by the text "re," meaning "with reference to," and the numeric value and unit of the reference quantity. For example, a pressure of 1 Pa, expressed in decibels with a reference of 1 micropascal (µPa), is written 120 dB re 1 µPa. The standard reference pressures are 1 µPa for water and 20 µPa for air. The reference pressure for air, 20 µPa, is the approximate lowest threshold of human hearing. It is important to note that because of the differences in reference units, the same sound pressures would result in different SPL values for each medium (the same sound pressure measured in water and in air would result in a higher SPL in water than in air, since the in-air reference is larger). Therefore, SPLs in air and in water cannot be directly compared.

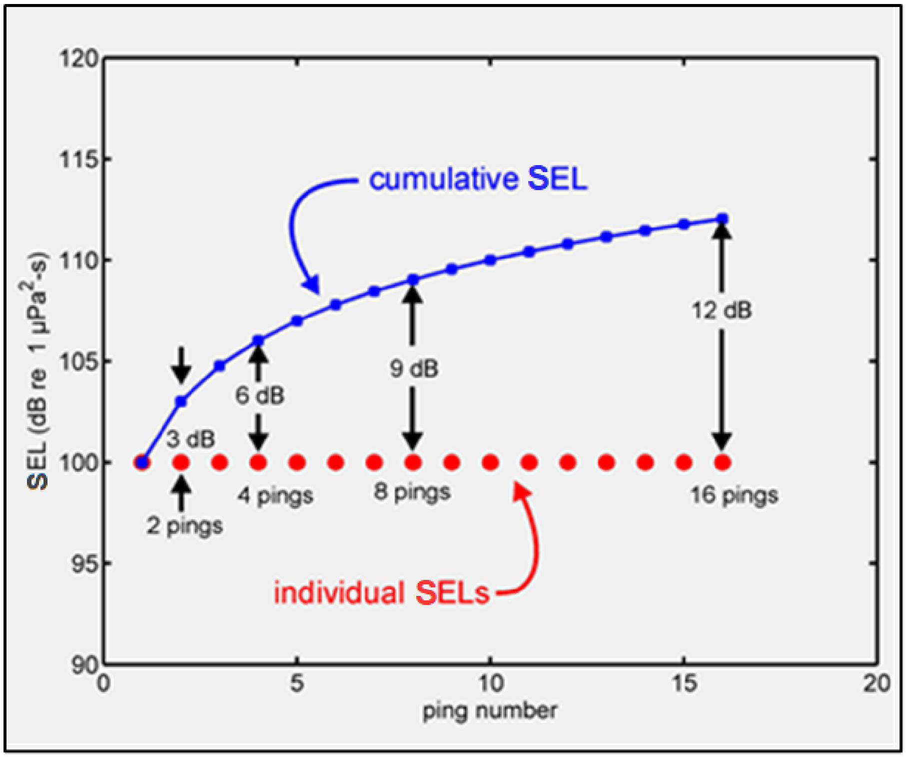
D.1.2.3 Sound Exposure Level

Sound exposure level (SEL) can be thought of as a composite metric that represents both the SPL of a sound and its duration. Individual time-varying noise events (e.g., a series of sonar pings or an impulsive sound) have two main characteristics: (1) a sound pressure that changes throughout the event and (2) a period during which a receiver is exposed to the sound. SEL can be provided for a single exposure (i.e., a single sonar ping or single explosive detonation) or for an entire acoustic event (i.e., multiple sonar pings or multiple explosive detonations). Cumulative SEL provides a measure of the net exposure of the entire acoustic event, but it does not directly represent the sound level at a given time. SEL is determined by calculating the decibel level of the cumulative sum-of-squared pressures over the duration of a sound, with units of dB re 1 micropascal squared seconds (re 1 $\mu Pa^2 s$) for sounds in water, and dB re 20 micropascal squared seconds (re 20 $\mu Pa^2 s$) for sounds in air.

Guidelines for SEL are as follows:

- The numeric value of SEL is equal to the SPL of a one-second sound that has the same total energy as the exposure event. If the sound duration is one second, SPL and SEL have the same numeric value (but not the same reference quantities). For example, a one-second sound with an SPL of 100 dB re 1 µPa has a SEL of 100 dB re 1 $\mu Pa^2 s$.
- If the sound duration is constant but the SPL changes, SEL will change by the same number of decibels as the SPL.
- If the SPL is held constant and the duration (T) changes, SEL will change as a function of $10\log_{10}(T)$:
    - $10 \log_{10}(10) = 10$, so increasing duration by a factor of 10 raises SEL by 10 dB.
    - $10 \log_{10}(0.1) = -10$, so decreasing duration by a factor of 10 lowers SEL by 10 dB.
    - $10 \log_{10}(2) \approx 3$, so doubling the duration increases SEL by 3 dB.
    - $10 \log_{10}(1/2) \approx -3$, so halving the duration lowers SEL by 3 dB.

Figure D.1-2 illustrates the summation of energy for a succession of sonar pings. In this hypothetical case, each ping has the same duration and SPL. The SEL at a particular location from each individual ping is 100 dB re 1 µPa^{2}s (red circles). The upper, blue curve shows the running total or cumulative SEL.

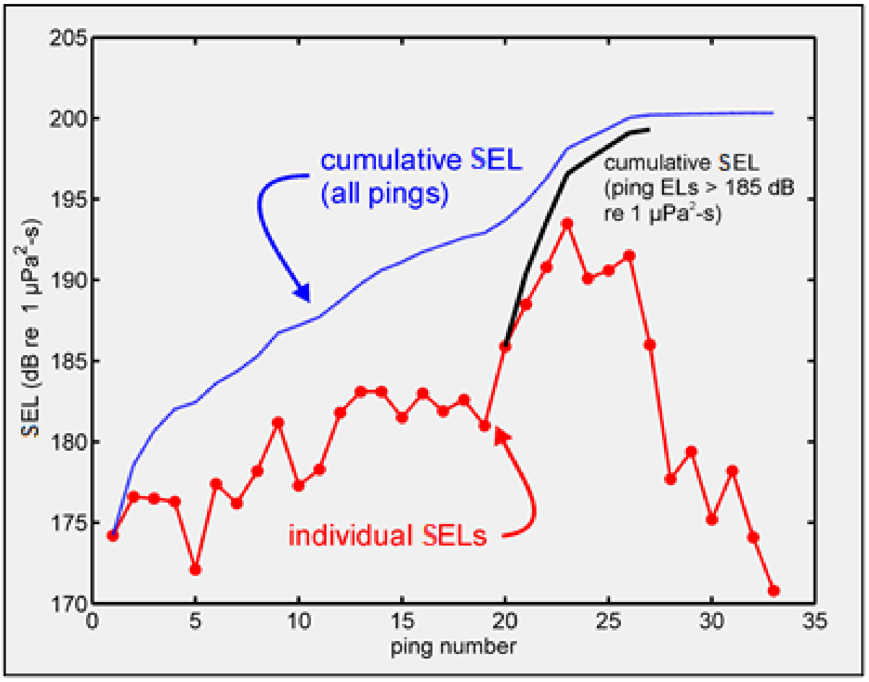


Note: dB = decibels; SEL = sound exposure level; dB re 1 µPa^2-s = decibels with a reference of 1 micropascal (µPa) squared per second

Figure D.1-2: Summation of Acoustic Energy from a Hypothetical, Intermittently Pinging, Stationary Sound Source

After the first ping, the cumulative SEL is 100 dB re 1 µPa^{2}s. Because each ping has the same duration and SPL, receiving two pings is the same as receiving a single ping with twice the duration. The cumulative SEL from two pings is therefore 103 dB re 1 µPa^{2}s. The cumulative SEL from four pings is 3 dB higher than the cumulative SEL from two pings, or 106 dB re 1 µPa^{2}s. Each doubling of the number of pings increases the cumulative SEL by 3 dB.

Figure D.1-3 shows a more realistic example where the individual pings do not have the same SEL. These data were recorded from a stationary hydrophone as a sound source approached, passed, and moved away from the hydrophone. As the source approached the hydrophone, the received SEL of each ping increased. After the source passed the hydrophone, the received SEL from each ping decreased as the source moved farther away (downward trend of red line), although the cumulative SEL increased with each additional ping received (slight upward trend of blue line). The main contributions are from those pings with the highest individual SELs. Individual pings with SELs 10 dB or more below the ping with the highest level contribute little (less than 0.5 dB) to the total cumulative SEL. This is shown in Figure D.1-3, where only a small error is introduced by summing the energy from the eight individual pings with SEL greater than 185 dB re 1 µPa^{2}s (black line), as opposed to including all pings (blue line).



Note: dB = decibels; ELs = exposure levels; SEL = sound exposure level; dB re 1 µPa^2-s = decibels with a reference of 1 micropascal (µPa) squared per second

Figure D.1-3: Cumulative SEL under Realistic Conditions with a Moving, Intermittently Pinging Sound Source

D.1.2.4 Particle Motion

The particles of a medium (e.g., water or air) oscillate around their original position as a sound wave passes through. Particle motion comprises particle displacement (m or dB re 1 pm), particle velocity (m/s or dB re 1 nm/s^2), and particle acceleration (m/s^2 or dB re 1 µm/s^2) (Nedelec et al., 2016a). Note that particle velocity is not the same as sound speed, which is how fast a sound wave moves through a medium. Particle motion is also directional, whereas sound pressure measurements are not (Nedelec et al., 2016a).

Near acoustic boundaries (e.g., the sea floor and sea surface) and in the shallow waters, the relationship between sound pressure and particle motion is complex and it is necessary to measure particle motion directly (Pierce, 1989). At distances far from a sound source (i.e., in the far field) and without boundary interactions that could cause wave interference, particle velocity is directly proportional to sound pressure. However, closer to a sound source (i.e., in the near field), the particle velocity component of the field contains more energy than the sound pressure component of the field. The rate of decline of particle velocity in the near field depends on the nature of the sound source and its movement pattern (Harris & van Bergeijk, 1962). The distance from a source at which the near field transitions to the far field is related to the wavelength of the signal, with a greater distance for lower frequencies.

D.1.2.5 Intensity

The intensity of a sound wave (I) is defined as the amount of energy per second (power in units Watts) propagating through 1 square meter of a medium (e.g., seawater). A propagating sound wave carries both kinetic energy of a medium’s particles in motion (particle velocity [u]) and potential energy due to the acoustic impedance of the medium (sound pressure [p]) and is calculated as follows:

$$I = pu$$

Intensity and velocity are both vector quantities with a magnitude and direction. The motion of particles in a sound wave are generally oriented in the direction of propagation at a velocity equal to the velocity of sound (c). In a plane wave, the sound pressure is related to the particle velocity by:

$$p = \rho c u, \text{ or } u = \frac{p}{\rho c}$$

Where the fluid density (ρ) and velocity of sound (c) are known as the specific acoustic impedance of the medium. Therefore, for a plane wave, the instantaneous intensity is related to the instantaneous sound pressure by:

$$I = \frac{p^2}{\rho c}$$

D.1.2.6 Impulse

Impulse is a metric used to describe the pressure and time component of a pressure wave. Impulse is typically only considered for high-energy exposures to impulsive sources, such as exposures of marine species close to explosives. Specifically, pressure impulse is the time integral of the pressure with units of Pascal-seconds (Pa-s). Impulse is a measured quantity that is distinct from the term “impulsive,” which is not a measurement term, but rather describes a type of sound (see Section D.1.1.5, Impulsive Versus Non-Impulsive Sounds).

D.1.3 PREDICTING HOW SOUND TRAVELS IN WATER

While the concept of a sound wave traveling from its source to a receiver is straightforward, sound propagation is complex because of the simultaneous presence of numerous sound waves of different frequencies and source levels (i.e., the sound radiated by a projector). Waves undergo changes in direction (i.e., reflection, refraction, and diffraction) that can cause interferences (waves adding together or cancelling one another out). Ocean bottom types, water density, and surface conditions also affect sound propagation. While simple examples are provided here for illustration, the Navy Acoustic Effects Model used to quantify acoustic exposures to marine mammals and sea turtles considers the influence of multiple factors to predict acoustic propagation [see technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase IV Training and Testing* (U.S. Department of the Navy, 2024b).

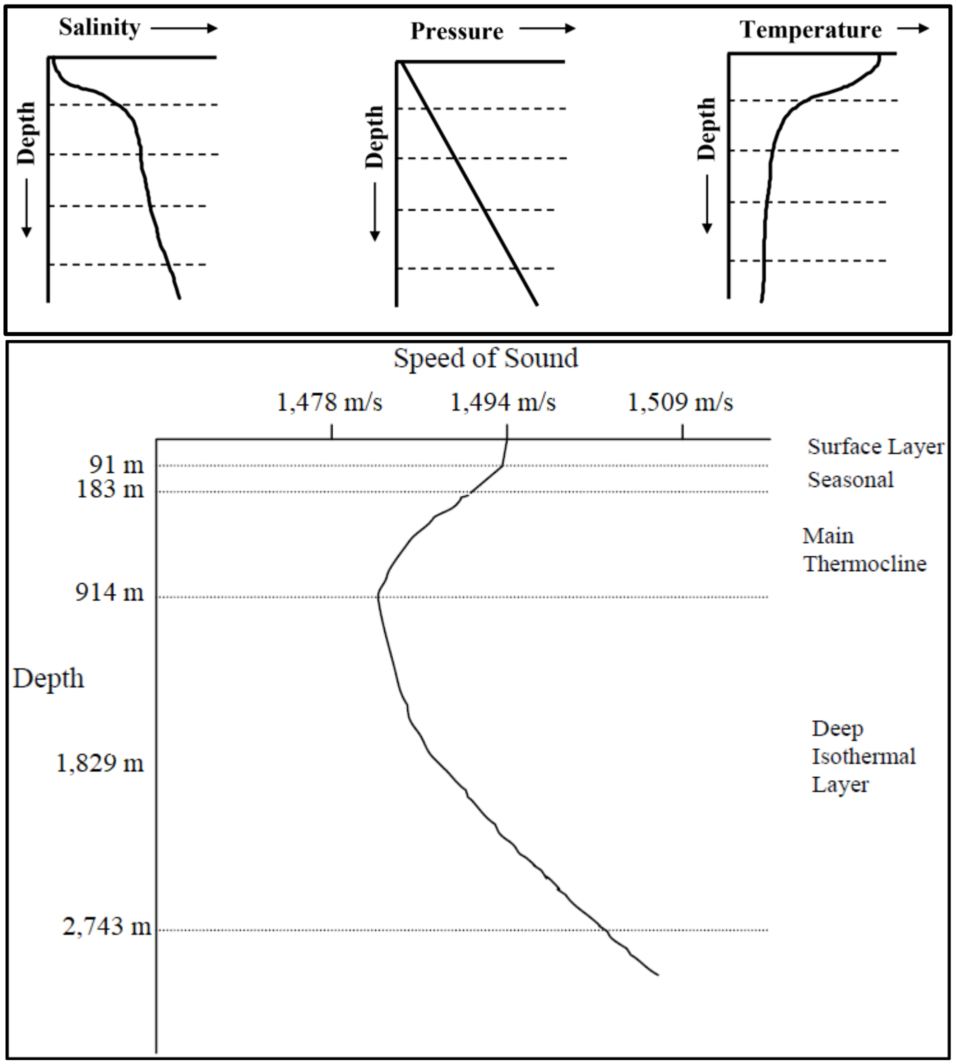
D.1.3.1 Speed of Sound

The speed of sound is not affected by the SPL or frequency of the sound, but depends wholly on characteristics of the medium through which it is passing. The speed of sound (c) is calculated using the bulk modulus (B), which describes resistance to compression, and density (ρ) of seawater, which are influenced by the pressure and temperature of the medium.

$$c = \sqrt{\frac{B}{\rho}}$$

Sound travels faster through a medium that is harder to compress. For example, water is more difficult to compress than air, and sound travels approximately 340 m/s in air and 1,500 m/s in seawater. The density of air is primarily influenced by temperature, relative humidity, and pressure, because these attributes affect the density and compressibility of air. Generally, the speed of sound in air increases as air temperature increases. The density of seawater is primarily influenced by temperature, pressure, and salinity. In general, the density is higher for colder temperatures, higher hydrostatic pressure, and higher salinity. The speed of sound in seawater also increases with increasing temperature and, to a lesser degree, with increasing hydrostatic pressure and salinity.

The combination of effects from temperature, pressure, and salinity creates a sound velocity profile. Figure D.1-4 shows the independent relationship each of these three attributes have with depth. For most areas of the ocean, temperature decreases from the surface to the bottom, although there are many local variations. Shallow layers see the most variation with time and depth (e.g., surface mixing, solar heating, currents, seasonal variations), and at deeper layers the temperature becomes relatively constant at 4°C. Hydrostatic pressure makes the speed of sound increase with depth because of variations in the bulk modulus. Below 1,500 meters (m), the increasing hydrostatic pressure is the dominant factor on sound speed. The change in the mix of pure water and dissolved salts affects the speed of sound. Salinity has minimal variation with depth, but there can be stronger variations near areas with freshwater inputs such as river estuaries and melting ice. Inhomogeneities in seawater can also affect the speed of sound and include bubble layers close to the surface, mineral particles in suspension, and living organisms.



Note: m = meters; m/s = meters per second

Figure D.1-4: Sound Velocity Profile (Sound Speed) Is Related to Temperature, Salinity, and Hydrostatic Pressure of Seawater

Figure D.1-4 also shows an example of a standard sound velocity profile and its four distinctive layers. The surface layer tends to be irregular and is influenced by diurnal (i.e., daily) heating and cooling; mixing from currents, local wind action, and storms; and changes in salinity due to evaporation, precipitation, freezing, ice melt, and river runoff. The surface layer may contain a mixed layer of isothermal (i.e., nearly constant temperature) water that traps sound. Under prolonged calm and sunny conditions, the mixed layer does not exist and water temperature decreases with depth. The seasonal thermocline (i.e., temperature gradient) is influenced by seasonal heating and cooling and mixing from wind action and storms. The seasonal thermocline is characterized by temperature decreasing with depth. During the summer and fall when waters are warm, the seasonal thermocline is well defined. However, during winter and spring or in cold waters, the seasonal thermocline can be indistinguishable from the surface layer. The main, or permanent thermocline, is independent of the surface layer, is only slightly affected by seasonal changes within a localized area and is where the major temperature difference between the cold depths of the sea occurs. The main thermocline extends to about 300 m and marks the limit where temperature has the most influence on sound velocity due to less mixing at greater depths. The deep isothermal layer is defined by a nearly constant temperature and sound velocity is mainly influenced by pressure. At the inflection point where sound velocity decreases with depth in the main thermocline, and where sound velocity begins to increase in the deep isothermal layer, is where a sound velocity minimum occurs and sound at depth is focused by refraction.

D.1.3.2 Source Directivity

Most sonar and other active acoustic sources do not radiate sound in all directions, unlike noise from vessels and explosions for example. Rather, they emit sounds over a limited range of angles to focus sound energy on a specific area or object of interest. The specific angles are sometimes given as horizontal or vertical beam width. Some sources can be described qualitatively as “forward-looking,” when sound energy is radiated in a limited direction in front of the source, or “downward-looking,” when sound energy is directed toward the bottom.

D.1.3.3 Transmission Loss

As a sound wave passes through a medium, the sound level decreases with distance from the sound source. This phenomenon is known as transmission loss (TL). The transmission loss is used to relate the source SPL (SL), defined as the SPL produced by a sound source at 1 m, and the received SPL (RL) at a particular location, as follows:

$$RL = SL - TL$$

The main contributors to transmission loss are as follows (Urick, 1983) and are discussed in detail below:

- Geometric spreading of the sound wave as it propagates away from the source
- Sound absorption (conversion of sound energy into heat)
- Scattering, diffraction, multipath interference, and boundary effects

D.1.3.3.1 Spreading Loss

Spreading loss is a geometric effect representing the regular weakening of a sound wave as it spreads out from a source. Spreading describes the reduction in sound pressure caused by the increase in surface area as the distance from a sound source increases. Spherical and cylindrical spreading are the simplest forms of spreading loss.

In the simple case of sound propagating from a point source without obstruction or reflection, the sound waves take on the shape of an expanding sphere. An example of spherical spreading loss is shown in Figure D.1-5. As spherical propagation continues, the sound energy is distributed over an ever-larger area following the inverse square law: the pressure of a sound wave decreases inversely with the square of the distance between the source and the receptor. For example, doubling the distance between the receptor and a sound source results in a reduction in the pressure of the sound to one-fourth of its initial value, tripling the distance results in one-ninth of the original pressure, and so on. Because the surface area of a sphere is $4\pi r^2$, where r is the sphere radius, the change in SPL with distance r from the source is proportional to the radius squared. This relationship is known as the spherical spreading law. The TL for spherical spreading between two locations is:

$$TL = 20log_{10}(r)$$

- 2 x distance, 6 dB loss
- 3 x distance, 10 dB loss
- 10 x distance, 20 dB loss

In cylindrical spreading, spherical waves expanding from the source are constrained by the water surface and the seafloor and take on a cylindrical shape. In this case the sound wave expands in the shape of a cylinder rather than a sphere, and the transmission loss is:

$$TL = 10log_{10}(r)$$

- 2 x distance, 3 dB loss
- 3 x distance, 5 dB loss
- 10 x distance, 10 dB loss

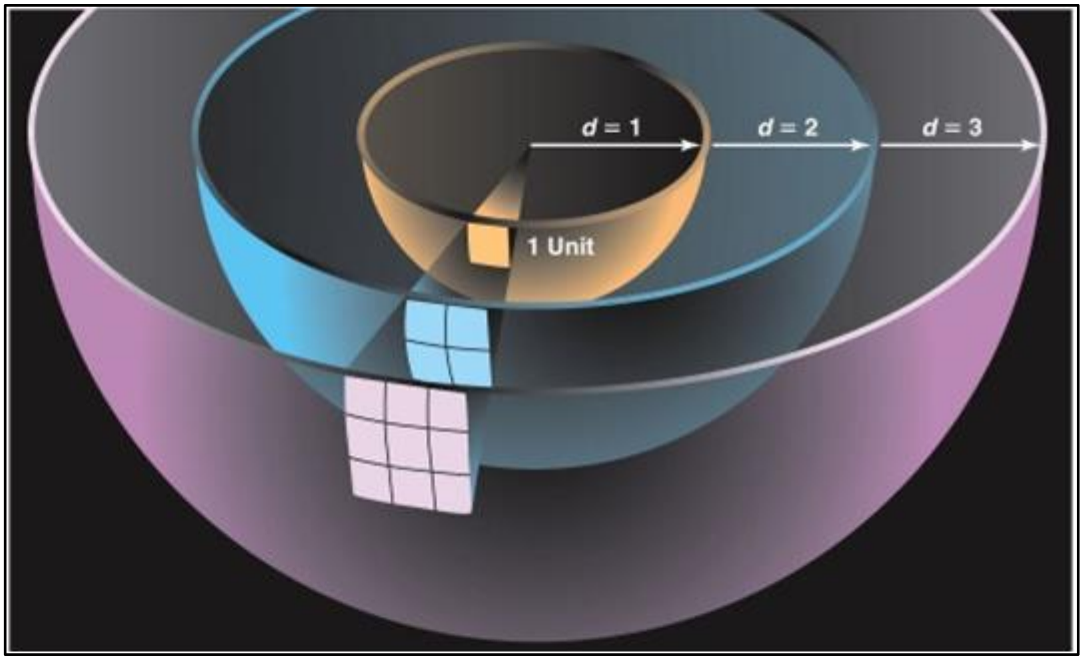


Figure D.1-5: Graphical Representation of the Inverse Square Relationship in Spherical Spreading with Increasing Distance from the Source (d)

The cylindrical and spherical spreading equations above represent two simple hypothetical cases. In reality, geometric spreading loss is more spherical near a source and more cylindrical with distance, and is better predicted using more complex models that account for environmental variables, such as the Navy Acoustic Effects Model [see technical report *Quantifying Acoustic Impacts on Marine Mammals and Sea Turtles: Methods and Analytical Approach for Phase IV Training and Testing* (U.S. Department of the Navy, 2024b).

D.1.3.3.2 Absorption

Absorption loss is the conversion of acoustic energy to heat and kinetic energy and occurs when sound propagates through a medium(Urick, 1983). Absorption is directly related to sound frequency, with higher frequencies (>10 kHz) having higher rates of absorption. The main cause of absorption in sea water occurs below 100 kHz and is due to ionic relaxation of dissolved salts (primarily magnesium sulfate). Therefore, absorption is the cause of an appreciable amount of attenuation for high- and very high-frequency sound sources, reducing the distance over which these sources may be perceived compared to mid- and low-frequency sound sources with the same source level.

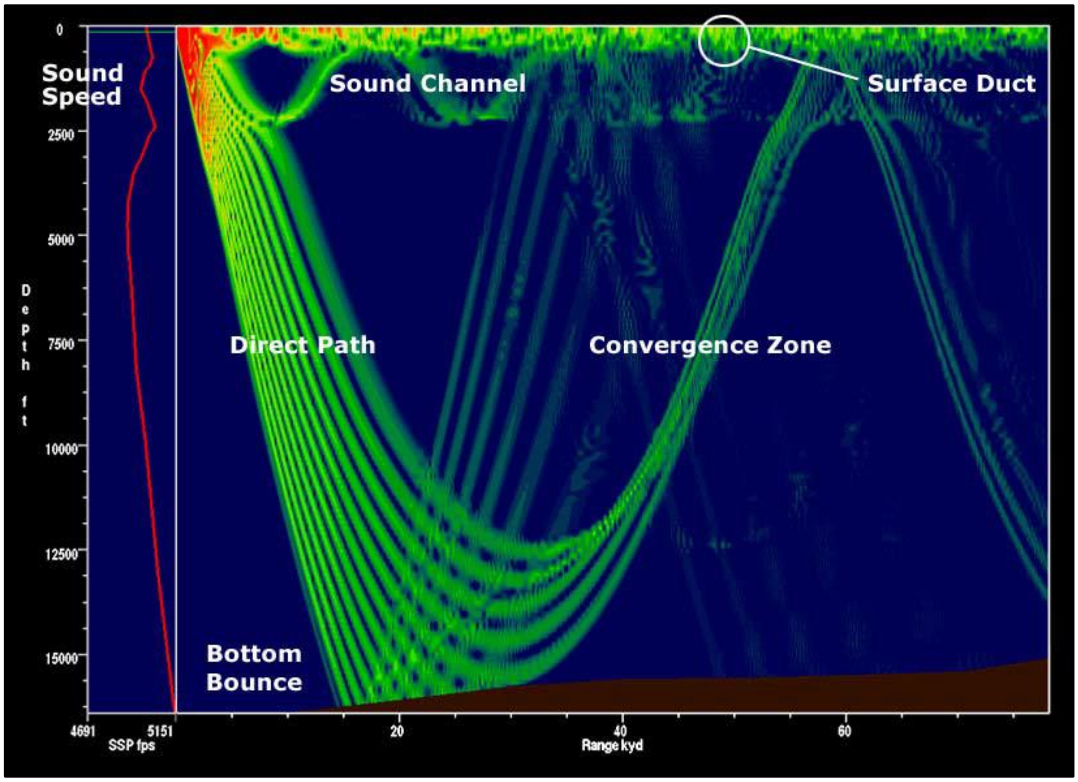
D.1.3.3.3 Refraction

When a sound wave propagating in a medium encounters a second medium with a different density (e.g., the air-water boundary), part of the incident sound will be reflected back into the first medium and part will be transmitted into the second medium (Kinsler et al., 1982). The propagation direction will change as the sound wave enters the second medium; this phenomenon is called refraction. Refraction may also occur within a single medium if the properties (e.g., temperature) of the medium change enough to cause a variation in the sound speed.

As discussed in Section D.1.3.1, Speed of Sound, the sound speed in the ocean primarily depends on hydrostatic pressure (i.e., depth) and temperature. Although the actual variations in sound speed are small, the existence of sound speed gradients in the ocean has an appreciable effect on the propagation of sound in the ocean. If one pictures sound as rays emanating from an underwater source, the propagation of these rays changes as a function of the sound speed profile in the water column. Specifically, the directions of the rays bend toward regions of slower sound speed. This phenomenon creates ducts in which sound becomes “trapped,” allowing it to propagate with high efficiency for large distances within certain depth boundaries. During winter months, the reduced sound speed at the surface due to cooling can create a surface duct that efficiently propagates sound such as commercial shipping noise (Figure D.1-6).

Sources located within this surface duct can have their sounds trapped, but sources located below this layer would have their sounds refracted downward. The deep sound channel, or sound frequency and ranging (SOFAR) channel is between 600 and 1,200 m deep at mid-latitudes and is where the slowest sound speed (i.e., sound speed minimum) occurs. The sound speed minimum creates a waveguide where sound waves are continually bent, or refracted, towards the region of lower sound speed which allows sound to travel long distances with minimal attenuation.

Similarly, the path of sound will bend toward regions of lower sound speed in air. Air temperature typically decreases with altitude. Since the speed of sound decreases in cooler temperatures, sounds produced in air tend to bend skyward. When an atmospheric temperature inversion is present, air is cooler near the earth’s surface than at altitude. In inversion conditions, sound waves near the earth’s surface will tend to refract downward.



Note: 1 kiloyard (kyd) = 0.9 km

Figure D.1-6: Sound Propagation Showing Multipath Propagation and Conditions for Surface Duct

D.1.3.3.4 Reflection and Multipath Propagation

In multipath propagation, sound may not only travel a direct path (with no reflection) from a source to a receiver, but also be reflected from the surface or bottom multiple times before reaching the receiver (Urick, 1983). Reflection is shown in Figure D.1-6 at the seafloor (bottom bounce) and at the water surface. At some distances, the reflected wave will be in phase with the direct wave (their waveforms add together and create a convergence zone), and at other distances the two waves will be out of phase (their waveforms cancel). The existence of multiple sound paths, or rays, arriving at a single point can result in multipath interference, a condition that permits the addition and cancellation between sound waves, resulting in the fluctuation of sound levels over short distances.

Reflection plays an important role in the pressures observed at different locations in the water column. Near the bottom, the direct path pressure wave may sum with the bottom-reflected pressure wave, increasing the exposure. Near the surface, however, the surface-reflected pressure wave may destructively interfere with the direct path pressure wave, by “cutting off” the wave and reducing exposure (called the Lloyd mirror effect). This can cause the sound level to decrease dramatically within the top few meters of the water column.

D.1.3.3.5 Diffraction, Scattering, and Reverberation

Diffraction, scattering, and reverberation are examples of what happens when sound waves interact with obstacles in the propagation path.

Diffraction may be thought as the change of direction of a sound wave as it passes around an obstacle. Diffraction depends on the size of the obstacle and the sound frequency. The wavelength of the sound must be larger than the obstacle for notable diffraction to occur. If the obstacle is larger than the wavelength of sound, an acoustic shadow zone will exist behind the obstacle where the sound is unlikely to be detected. Common examples of diffraction include sound heard from a source around the corner of a building and sound propagating through a small gap in an otherwise closed door or window.

An obstacle or inhomogeneity (e.g., smoke, suspended particles, gas bubbles due to waves, and marine life) in the path of a sound wave causes scattering as these inhomogeneities reradiate incident sound in a variety of directions (Urick, 1983). Reverberation refers to the prolongation of a sound, after the source has stopped emitting, caused by multiple reflections at water boundaries (surface and bottom) and scattering.

D.1.3.3.6 Surface and Bottom Effects

Because the sea surface reflects and scatters sound, it has a major effect on the propagation of underwater sound in applications where either the source or receiver is at a shallow depth (Urick, 1983). If the sea surface is smooth, the energy from a reflected sound wave is nearly equal to the energy of an incident (i.e., incoming) sound wave; however, if the sea surface is rough, the amplitude of the reflected sound wave will be reduced. Sound waves in water reflected from a boundary with air (i.e., the sea surface) experience a phase reversal (i.e., a 180° change). When the surface-reflected waves interact with the direct path waves near the surface, a destructive interference pattern is created in which the two waves are out of phase by half a cycle and cancel each other out when added together. As a result, the amplitude of the two waves and the sound pressure become zero.

The sea bottom is also a reflecting and scattering surface, like the sea surface. Sound interaction with the sea bottom is more complex, primarily because the acoustic properties of the sea bottom are more variable, and the bottom is often layered into regions of differing density. As sound travels into the seafloor it reflects off these different density layers in complex ways. For sources in contact with the bottom, such as bottom-placed explosives, a ground wave is produced that travels through the bottom sediment and may refract back into the water column.

Sediment grain size, composition, and the measure of pore space (i.e., porosity) affect sound propagation and attenuation at the sea floor. In addition, sediments contain free or trapped gas and/or organic content which can affect the bulk properties of the sediment. For a hard bottom such as rock, the reflected wave will be approximately in phase with the incident wave. Thus, near the ocean bottom, the incident and reflected sound pressures may add together (constructive interference), resulting in increased sound pressure near the sea bottom. Soft bottoms such as mud or sediment absorb sound waves and reduce the level in the water column overall.

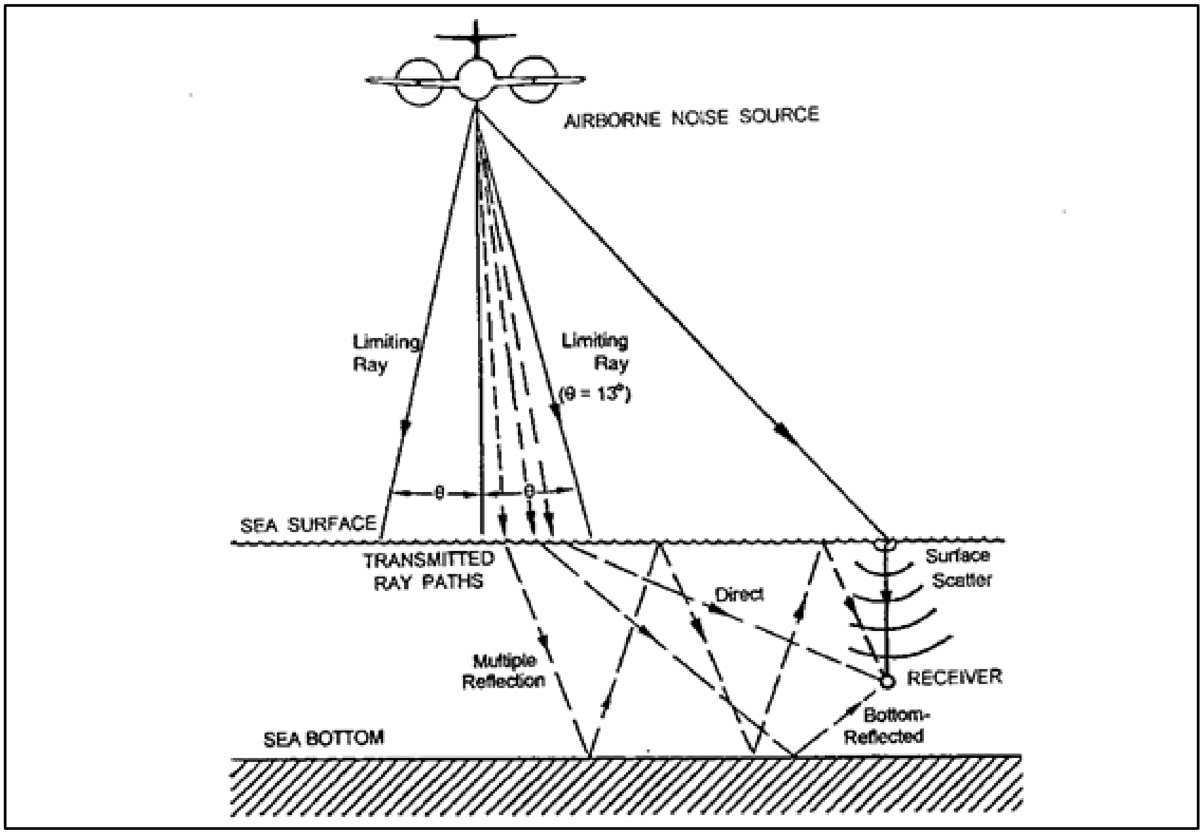
D.1.3.4 Air-Water Interface

Sound from aerial sources such as aircraft and weapons firing may be transmitted into the water under certain conditions. The most studied of these sources are fixed-wing aircraft and helicopters, which create noise with most energy below 500 Hz. Underwater noise levels are highest at the surface and are highly dependent on the altitude of the aircraft, the angle at which the aerial sound encounters the water surface, and the amount of wave action and surface roughness. Transmission of the sound once it is in the water is identical to any other sound as described in the sections above.

Transmission of sound from a moving, airborne source to a receptor underwater is influenced by numerous factors and has been addressed by Young (1973), Urick (1983), Richardson et al. (1995b), Eller and Cavanagh (2000), U.S. Department of the Air Force (2000), and others. Sound is transmitted from an airborne source to a receptor underwater by four principal means: (1) a direct path, refracted upon passing through the air-water interface; (2) direct-refracted paths reflected from the bottom in shallow water; (3) evanescent transmission in which sound travels laterally close to the water surface; and (4) scattering from interface roughness due to wave motion.

At the air-water interface, sound can either be transmitted across the air-water boundary or reflected off the water surface. When sound waves meet the water at a perpendicular angle (e.g., straight down from an in-air source to a flat-water surface), the sound waves are both transmitted directly across the water surface in the same direction of travel and reflected 180 degrees back toward the original direction of travel. This can create a localized condition at the water surface where the incident and reflected waves sum, resulting in constructive interference, and doubling the in-air overpressure (+ 6 dB). As the incident angle of the in-air sound wave changes from perpendicular, this phenomenon is reduced, ultimately reaching the angle where sound waves are parallel to the water surface and there is no surface reflection.

The sound that enters the water is refracted due to the difference in sound velocity between air and water, as shown in Figure D.1-7. As the angle of the in-air incident wave moves away from perpendicular, the direction of travel of the underwater refracted waves becomes closer to parallel to the water surface. When the incident angle is reached, the underwater refracted sound wave is parallel to the water surface and all the sound is reflected into the air and no sound enters the water. This occurs at an angle of about 13 to 14 degrees. As a result, most of the acoustic energy is transmitted into the water through a relatively narrow cone extending vertically downward from the in-air source. The width of the footprint would be a function of the source altitude. Lesser amounts of sound may enter the water outside of this cone due to surface scattering (e.g., from water surface waves that can vary the angle of incidence over an area) and as evanescent waves that are only present very near the surface.



Source: (Richardson et al., 1995b)

Figure D.1-7: Characteristics of Sound Transmission Through the Air–Water Interface

If a sound wave is ideally transmitted into water (that is, with no surface transmission loss, due to foamy, wave conditions that could decrease sound entering the water), the SPL underwater is calculated by changing the pressure reference unit from 20 µPa in air to 1 µPa in water. For a sound with the same pressure in air and water, this calculation results in a +26 dB SPL in water compared to air. Sounds of equal intensity, however, will be 62 dB higher in water than in air. This is due not only to the difference in reference pressures, but also differences in impedance. For this reason, sound measurements in water and in air cannot be directly compared.

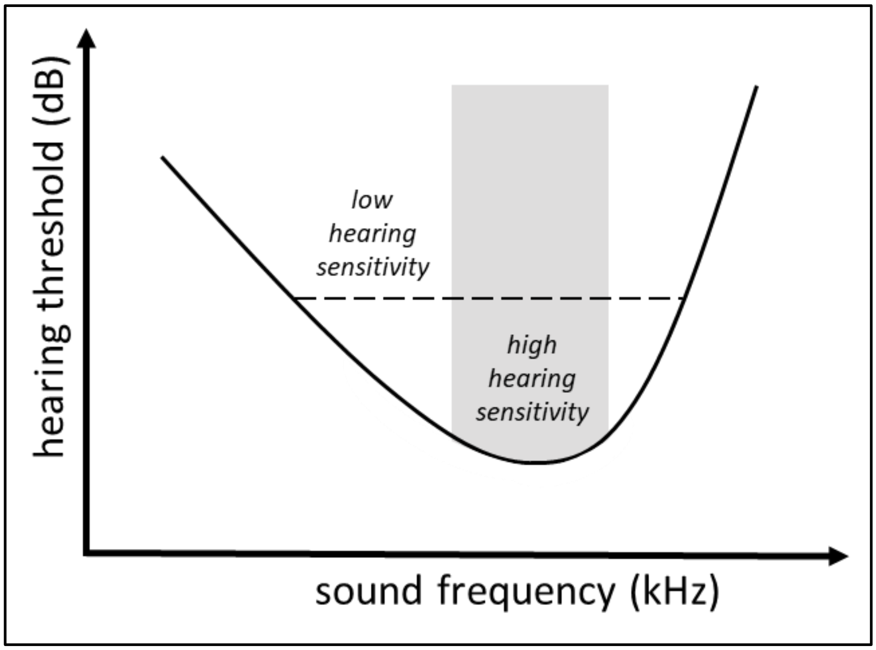
D.1.4 AUDITORY PERCEPTION

Animals with an eardrum or similar structure, including mammals, birds, and reptiles, detect the pressure component of sound. Some marine fishes also have specializations to detect pressure changes, although most invertebrates and many marine fishes do not have anatomical structures that enable them to detect the pressure component of sound and are only sensitive to the particle motion component of sound. This difference in acoustic energy sensing mechanisms limits the range at which fishes and invertebrates can detect most sound sources.

Because mammalian ears can detect large pressure ranges and humans judge the relative loudness of sounds by the ratio of the sound pressures (a logarithmic behavior), sound amplitude is described by the SPL, calculated by taking the logarithm of the ratio of the sound pressure to a reference pressure (see Section D.1.2.2, Sound Pressure Level). Use of a logarithmic scale compresses the wide range of pressure values into a more usable numerical scale. On the decibel scale, the smallest audible sound in air (near total silence) to a human is 0 dB re 20 µPa. If the sound intensity increases by a factor of 10, the SPL would increase to 10 dB re 20 µPa. If the sound intensity increases by a factor of 100, the SPL would increase to 20 dB re 20 µPa, and if the sound intensity increases by a factor of 1000, the SPL would be 30 dB re 20 µPa. A quiet conversation has an SPL of about 50 dB re 20 µPa, while a jet engine taking off 200 ft away is about 130 dB re 20 µPa (Cavanaugh & Tocci, 1998).

While sound pressure and frequency are physical measures of the sound, loudness is a subjective attribute that varies not only with sound pressure but also other attributes of the sound, such as frequency. For example, a human listener would perceive a 60 dB re 20 µPa sound at 2 kHz to be louder than a 60 dB re 20 µPa sound at 50 Hz, even though the SPLs are identical. This effect is most noticeable at lower SPLs; however, at very high SPLs, the difference in perceived loudness at different frequencies becomes smaller. This difference in perception for sounds having the same SPLs but different frequencies is related to the hearing capabilities of the individual or species.

The most accurate tests for determining the hearing capabilities of animals are direct measurements of auditory sensitivity. The two standard types of hearing tests are: 1) behavioral, where an animal is trained to provide a response to sound, and 2) physiological, where – without any training – the brain's responses to sound are measured (auditory-evoked potentials, or AEPs) (Finneran, 2015). During these tests, the sound is played at progressively lower levels until the animal can no longer hear it or until the brain's responses are no longer detected, and the hearing threshold in dB SPL is determined. The hearing threshold is the quietest audible sound, so a low hearing threshold indicates more sensitive hearing. When multiple frequencies are tested across the hearing range of an animal, a plot called an audiogram illustrates how hearing threshold changes as a function of sound frequency. An example of an audiogram is shown in Figure D.1-8.



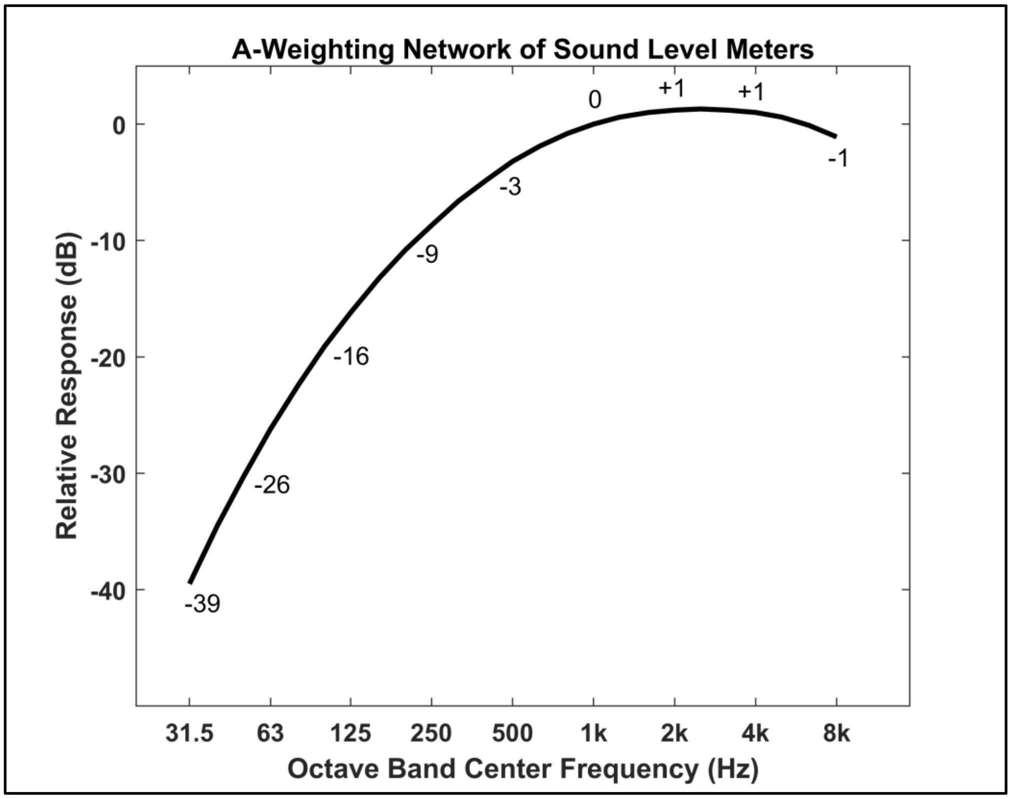
Notes: (dB = decibels; kHz = kilohertz) The area within the solid curve represents audible sounds. The dotted line illustrates that the listener is not as sensitive to frequencies on the tail ends of the curve as the frequencies that align with the bottom of the "U." The shaded area is the frequency range with the lowest thresholds and highest hearing sensitivity, also called the region of best hearing. Marine mammal auditory sensitivity typically decreases more slowly at frequencies lower than the best frequency and decreases more quickly for frequencies higher than the best frequency.

Figure D.1-8: Example of an Audiogram

To account for differences in hearing sensitivity at various frequencies, acoustic risk analyses commonly use auditory weighting functions—mathematical functions that adjust (or "weight") received sound levels with frequency based on how the listener's sensitivity or susceptibility to sound changes at different frequencies. For humans, the most common weighting function is called "A-weighting" (see Figure D.1-9). A-weighted sound levels are specified in units of "dBA" (A-weighted decibels). For example, if the unweighted received level of a 500 Hz tone at a human receiver was 90 dB re 20 µPa, the A-weighted sound level would be 90 dB – 3 dB = 87 dBA because the A-weighting function amplitude at 500 Hz is -3 dB (Figure D.1-9. Many measurements of sound in air appear as A-weighted decibels in the literature because the intent of the authors is to assess noise impacts on humans.

The auditory weighting concept can be applied to other species. When used in analyzing the impacts of sound on an animal, auditory weighting functions adjust received sound levels to emphasize ranges of best hearing and de-emphasize ranges of less or no sensitivity. Auditory weighting functions were developed for marine mammals and sea turtles and are used to assess acoustic impacts. Additional information on auditory weighting functions and their derivation for this analysis are described in the *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase IV) technical report* (U.S. Department of the Navy, 2024a).

Masking occurs when noise interferes with the detection, discrimination, or recognition of the relevant sound or signal (Erbe et al., 2016). Auditory masking is defined as the amount in dB by which the threshold of hearing for one sound is raised by the presence of a masking sound (Acoustical Society of America, 2015). Masking occurs only in the presence of the masking noise and does not persist after the cessation of the noise.



Notes: (dB = decibels; Hz = hertz) The numbers along the curve indicate how a received sound level would be adjusted at that frequency.

Figure D.1-9: A-Weighting for Human Hearing of Sounds in Air (adapted from OSHA)

D.1.5 ACOUSTIC PROPAGATION IN SMALL TANKS

Although it is common to conduct bioacoustic research in small tanks with fishes, invertebrates, and other taxa, results from such experiments should be considered with caution due to the complicated acoustic fields that exist within small tank environments (Akamatsu et al., 2002). In a natural environment such as the open ocean, the particle velocity component of a signal contains more energy closer to the source (i.e., in the near field) compared to sound pressure. As sound propagates away from the source, this relationship shifts into a linear one as the two decay at the same rate in the far field. In a small tank, the acoustic field is complicated by boundaries, specifically the air-water interface at the walls and floor of the tank, and at the water surface (Akamatsu et al., 2002). These boundaries cause multiple overlapping reflections that alter the relationship between particle motion and sound pressure in the near field, attenuate the low-frequency components of the sound, and distort the directionality of the signal. As described in Section D.1.1.8, Resonance, it is known that small containers have resonant frequencies depending on their physical dimensions. When the acoustic signal used in an experiment overlaps that of the tank's resonant frequency, the sound is further distorted. Additionally, the physical dimensions of small tanks can be shorter than the wavelength of the signal used in bioacoustic experiments, further complicating the potential received signal. The placement of the sound source is also an important consideration as there is evidence that the source characteristics may vary at the receiver depending on whether the transducer is located in-water (within the tank) or in-air (adjacent to the tank) (Rogers et al., 2016). It is important for laboratory tests in small tanks to properly measure and characterize the sound field considering reverberations and refractions off the boundaries of the tank (Takahashi & Akamatsu, 2018), as well as the test subject itself (especially when using animals that contain air filled organs). In the absence of such considerations, experiments conducted in small tanks may overestimate or mischaracterize the results.

D.1.6 EXPLOSIVES

Explosive materials used in Navy military readiness activities are either (1) high explosives (HE) material has a fast rate of detonation (exceeding the speed of sound), or (2) low explosives, which exhibit a relatively slow burn, or deflagration, such as black powder. Because low explosives are typically used in small quantities and have less destructive power, the below discussion focuses on high explosives.

The rate of detonation of a high explosive is supersonic and instantaneous, producing a steep, high-pressure shock wave that travels forward through explosive material. This shock front is produced by the supersonic expansion of the explosive products, but as the shock front travels away from the immediate area of the detonation, it begins to behave as an acoustic wave front travelling at the speed of sound.

The near-instantaneous rise from ambient to an extremely high peak pressure is what makes the explosive shock wave potentially damaging. Explosive exposures are usually characterized by the metrics of impulse and peak pressure. The positive impulse is calculated by integrating the positive pressure over the duration of the positive phase. The positive pressure produced by an explosion is also referred to as overpressure. As the shock front passes a location, the positive pressure exponentially decays, as shown in Figure D.1-10. As the shock front travels away from the detonation, the waveform is stretched—the peak pressure decreases while the positive duration increases. Both the reduction in peak pressure and stretching of the positive impulse reduce the potential for injury. In addition, absorption losses of higher frequencies over distance results in a softening of the shock front, such that the rise to peak pressure is no longer near instantaneous.

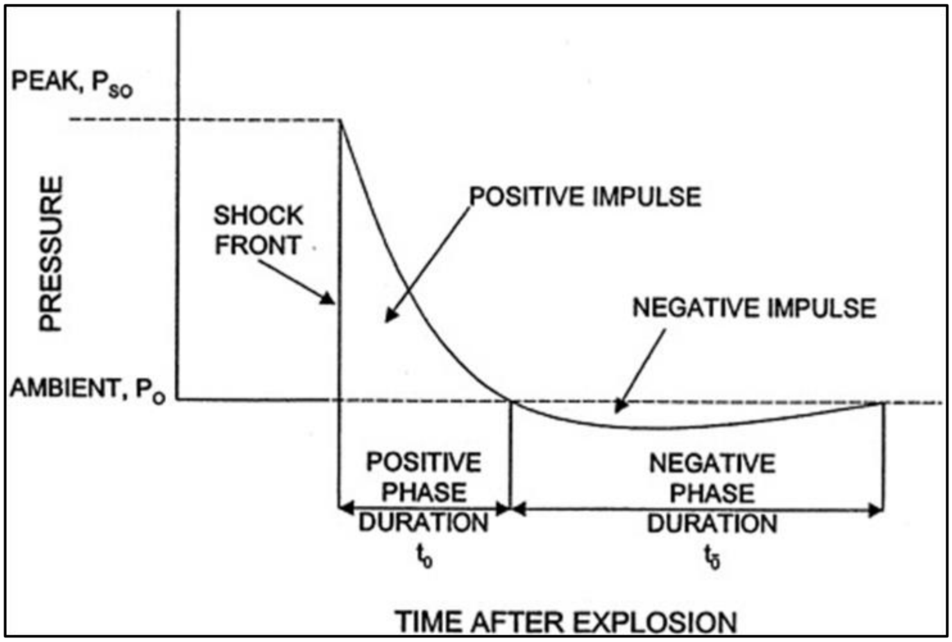


Figure D.1-10: Impulse Shown as a Function of Pressure over Duration at a Specific Location

The peak pressure experienced by a receptor (i.e., an animal) is a function of the explosive material, the net explosive weight (NEW), and the distance from the charge. NEW is a way to classify and compare quantities of different explosive compounds. The NEW for a given charge is the energetic equivalent weight of trinitrotoluene (TNT). In general, shock wave effects near an explosive charge increase in proportion to the cube root of the explosive weight (Young, 1991). For example, shock wave impacts will double when the explosive charge weight is increased by a factor of eight (i.e., cube root of eight equals two). This cube root scaling relationship is known as the similarity principle and allows for a simple prediction of peak pressure in a uniform free field environment to estimate explosive effects in air and in water. However, at longer distances or in more complex environments with boundaries and variations in the propagation medium, explosive propagation modeling is preferred.

D.1.6.1 Explosions in Air

Pressure waves from explosions in air interact with the air-water boundary as previously described under Section D.1.3.4, Air-Water Interface. In certain explosive geometries, depending on the size of the explosive and its height of detonation, a combined shock wave, called a Mach stem, can be created when direct and reflected shock waves merge and form a single wave (Kinney & Graham, 1985). In instances where this specific geometry does not occur, only the direct path wave is experienced because there is no surface reflection (waves are parallel to or angled away from the water surface, such as would occur when an explosive is detonated at the water surface), or separate direct and reflected pressure waves may be experienced.

D.1.6.2 Explosions in Water

At the instant of explosion underwater, gas byproducts are generated at high pressure and temperature, creating a bubble. The heat causes a certain amount of water to vaporize, adding to the volume of the bubble. This action immediately begins to force the water in contact with the blast front in an outward direction, creating an intense, supersonic-pressure shock wave. As the high-pressure wave travels away from the source, it slows to the speed of sound and acts like an acoustic wave like other impulsive sources that lack a strong shock wave. Explosions have the greatest amount of energy at frequencies below 500 Hz, although energy is present at frequencies exceeding 10 kHz (Urick, 1983). The higher frequency components exhibit more attenuation with distance due to absorption (see Section D.1.3.3.2, Absorption).

The shock wave caused by an explosion in deeper water may be followed by several bubble pulses in which the explosive byproduct gases expand and contract, with correlated high- and low-pressure oscillations. These bubble pulses lack the steep pressure front of the initial explosive pulse, but the first bubble pulse may still contribute to the total energy released at frequencies below 100 Hz (Urick, 1983). Subsequent bubble pulses contribute little to the total energy released during the explosion (Urick, 1983). If the detonation occurs at or just below the water surface, a portion of the explosive power is released into the air and a pulsating gas bubble is not formed.

The pressure waves from an explosive can constructively add or destructively cancel each other in ocean environments with multi-path propagation, as described for acoustic waves in Sections D.1.3.3.3, Refraction, and D.1.3.3.4, Reflection and Multipath Propagation. The received impulse is affected by the depth of the charge and the depth of the receiving animal. Pressure waves from the detonation may travel directly to the receiver or interact with the water surface or sea floor before arriving at the receiver. If a charge is detonated closer to the surface or if an animal is closer to the surface, the time between the initial direct path arrival and the following surface-reflected tension wave arrival is reduced, resulting in a steep negative pressure cut-off of the initial direct path positive impulse exposure. Two animals at similar distances from a charge, therefore, may experience the same peak pressure but different levels of impulse at different depths.

D.2 ACOUSTIC HABITAT

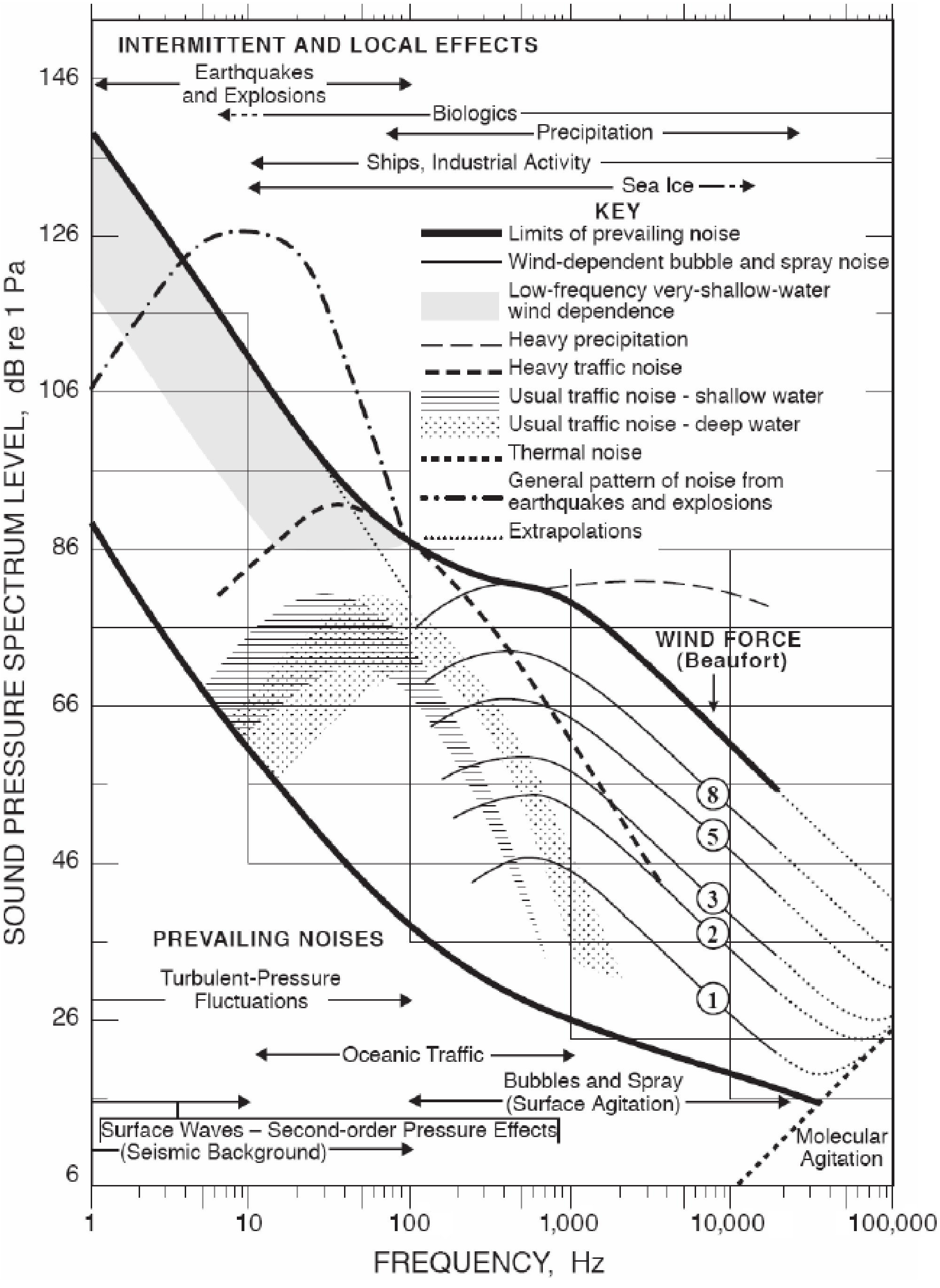
Ambient noise is defined as encompassing all noise at a specific location and time in the absence of a specified sound (International Organization for Standardization, 2017). Ambient noise is continuous and has considerable variation across time and space, varying by as much as 10 to 20 dB from day to day (Richardson et al., 1995b). The first systematic investigation of ambient noise was performed by Knudsen et al. (1948) and examined the relationship between noise level, wind speed, and sea state. Wenz (1962) expanded on the work by Knudsen et al. (1948) and described the spectra of natural and anthropogenic sources that contribute to noise in the ocean (Figure D.2-1). In general, the ambient noise spectrum can be broadly categorized into three frequency bands (Wenz, 1962). The low-frequency band (10 to 500 Hz) is dominated by shipping noise, the mid-frequency band (500 Hz to 25 kHz) is governed by surface agitation from wind and weather, and the high-frequency band (greater than 25 kHz) is influenced by thermal noise from molecular agitation of water molecules (particularly greater than 50 kHz). Despite changes in the ocean environment, the Knudsen Curves and Wenz Curves are still applicable and useful for understanding and estimating noise levels.

D.2.1 NATURAL NOISE

In underwater soundscape ecology, naturally occurring noise is categorized as geophony, which includes natural sounds of the earth (e.g., wind, waves, and earthquakes), and biophony, which includes sounds from living organisms (e.g., whales, fish, and snapping shrimp). Anthropophony (human generated signals) are not considered part of natural environmental noise. In the absence of distant shipping noise, natural sources dominate the long-term, time-averaged ocean noise across all frequencies. When distant shipping noise is present, natural sources continue to dominate time-averaged ocean noise spectra below 5 Hz and from around 500 Hz to over 200 kHz (National Research Council, 2003; Wenz, 1962). Prevalent sources of naturally occurring noise discussed in this section are generated by processes including wind, waves, rain, earthquakes, volcanoes, thermal noise, and biological sources.

D.2.1.1 Surface Interactions

Prevailing ambient noise associated with wind, waves, and rain has multiple contributing factors across a broad frequency range from below 1 Hz to at least 50 kHz (Figure D.2-1). Between 500 Hz and 25 kHz, ambient noise is governed by wind speed, sea state, and resulting surface agitation including air bubble cavitation and spray. At frequencies lower than 500 Hz, ambient noise is less correlated with wind speed and sea state, and as low as 50 to 100 Hz no relationship exists (Wenz, 1962). Noise from shipping and other anthropogenic activities become the prevalent sources of ambient noise at frequencies lower than 500 Hz and it is difficult to discern the impact of wind related noise at lower frequencies (Wenz, 1962). The wind-generated noise spectra for a given sea state (i.e., Beaufort 1, 2, 3, 5, and 8 in Figure D.2-1) have a slope of -5 dB/octave (e.g., a loss of 5 dB of sound energy for each doubled frequency range) or -18 dB/decade (e.g., a loss of 18 dB of sound energy for each tenfold frequency range) and a -29 dB in the spectra from 500 Hz to 25 kHz (Knudsen et al., 1948). Cavitating air bubbles that form near the surface and grow due to a process called rectified diffusion from pressure changes caused by waves, contribute to overall noise levels when bubbles collapse. Whitecaps and spray at the surface can increase estimated noise levels for a given Beaufort sea state in Figure D.2-1 by 4 to 5 dB when conditions are unusually windy, such as during a large storm (Knudsen et al., 1948). In contrast, estimated noise levels for a given Beaufort sea state may be lower than those in Figure D.2-1 when there is reduced spray and calm conditions.



Source: Wenz (1962)
Note: Hz = hertz; dB re 1 µPa = decibels with a reference of 1 micropascal (µPa)

Figure D.2-1: Wenz Curves Describing the Spectra of Ambient Ocean Noise

At frequencies below 10 Hz, surface gravity wave interactions create pressure fluctuations. First order pressure effects are due to the elevation and movement of water at the surface and causes subsurface pressure fluctuations below 0.3 Hz at less than 100 m depth (Wenz, 1962). Second order pressure effects occur when two surface waves with the same wavelength travel in opposite directions (e.g., from being reflected offshore). This magnifies the crests and troughs and form a standing wave with consistent pressure across depth, and a frequency twice that of the two surface waves. The noise spectrum of a standing wave has a slope of -8 to -10 dB/octave in the frequency range from 1 to 10 Hz (Wenz, 1962).

Intermittent ambient noise from rain is affected by the rate of rainfall, droplet size, wind speed, and area covered. Together, these factors contribute to noise levels primarily above 500 Hz, however, noise levels can extend to lower frequencies (e.g., if heavy rainfall occurs with low wind speeds) (Wenz, 1962). Underwater noise from rainfall is generated by the impact of droplets on the water surface, and by trapping a bubble underwater during a splash (Nystuen, 2001). Rain droplet size affects the underwater sound spectrum. Small droplets (0.8 to 1.2-millimieter [mm] diameter) have a strong signal in the spectrum from 13 to 25 kHz; medium droplets (1.2 to 2.0 mm diameter) have a signal from 1 to 30 kHz; large droplets (2.0 to 3.5-mm diameter) have a signal from 1-35 kHz with a peak in the spectrum at 2 to 5 kHz, and very large droplets (greater than 3.5-mm diameter) have a signal from 1-50 kHz with a peak in the spectrum from 1-2 kHz (Nystuen, 2001). During light rainfall, the ambient noise level can increase by 10 to 20 dB around 15 kHz (Nystuen & Farmer, 1987). In the 1 to 50 kHz range, heavy rainfall can increase the noise level up to 35 dB, and during extreme rainfall events (rate greater than 100 mm/hour) the noise level can increase up to 50 dB (Nystuen, 2001).

D.2.1.2 Biological Sources

Biological sources with an appreciable contribution to underwater ambient noise levels are briefly summarized here. Additional details on sounds from biological sources are provided in the sections below.

Marine mammal vocalizations cover a wide frequency range from less than 10 Hz to around 200 kHz. Broadband clicks and burst pulse signals produced by odontocetes can be used for echolocation, navigation, prey capture, and communication and have peak energy between approximately 10 and 150 kHz. Odontocetes also produce whistles for communication with fundamental frequencies between approximately 1 and 50 kHz. Vocalizations from mysticetes are lower frequency, from tens of Hertz to typically less than 10 kHz, and have the potential to be detected over long distances. For example, low-frequency blue whale calls can be heard by other whales up to 1,600 km away. An exception are humpback whales which can produce calls over 10 kHz (Zoidis et al., 2008) with harmonics up to 24 kHz (Au et al., 2006). Calls from mysticetes are diverse and complex in composition and are used for breeding, feeding, navigation, and communication. Depending on the timing and location, marine mammal vocalizations can be the dominant source of underwater noise in a region. For example, vocalizations produced by migrating mysticetes can seasonally increase ambient noise levels an average of 2 to 9 dB and up to 25 dB in the 15 to 22 Hz band (Curtis et al., 1999).

Many species of fish produce pulsed signals with most energy below 1 kHz for communication, courtship, mating, aggressive interactions, and when in distress (National Research Council, 2003). The occurrence of fish sounds can also exhibit diurnal, lunar, seasonal, and annual temporal variability. Sounds are produced by individuals, and collectively, many individuals produce choruses which can cause a sustained increase of 10-30 dB in ambient noise levels under 3 kHz (Cato, 1978; D'Spain & Batchelor, 2006).

Sounds from marine invertebrates are prolific in bays, harbors, estuaries, and coastal areas, and can be a major source of biological noise. Snapping shrimp produce high intensity, broadband impulses to communicate, deter predators, and stun prey. Sounds they produce have peak energy from 2 to 5 kHz with spectral components up to 250 kHz (Au & Banks, 1998) and can increase ambient noise levels up to 20 dB (Hildebrand, 2009). They occur in large aggregations in shrimp beds and are prevalent year-round in shallow and warm waters between +/- 40 degrees latitude (Knudsen et al., 1948). Snap rates are positively correlated with water temperature, and noise levels can vary up to 15 dB in the 1.5 to 20 kHz frequency band between winter and summer (Bohnenstiehl et al., 2015). Although sounds from snapping shrimp are the most prevalent, other marine invertebrates generate sounds as well. For example, sea urchins generate a scraping sound during feeding from 800 to 2,800 Hz (Radford et al., 2008), and spiny lobsters generate broadband pulses called "antennal rasps," potentially for intra-specific communication, with most energy below 1 kHz (Jezequel et al., 2022).

D.2.1.3 Geologic Activity

Geologic activity primarily contributes to ocean noise at frequencies less than 100 Hz. Earthquake generated acoustic waves in the ocean are called T-waves (tertiary waves) and produce intermittent sound at low frequencies. Earthquakes can occur under the ocean floor, or originate on land, and propagate between the land and ocean interface. Small earthquakes are more frequent and almost continuous in seismically active regions (e.g., the Mid-Atlantic Ridge and the East Pacific Rise). Recordings of earthquakes at the Mid-Atlantic Ridge have an estimated average source level between 199 and 234 dB re 1 μPa (Williams et al., 2006a), and a 20 dB increase in the ambient noise level has been observed in the 5 to 32 Hz band (McGrath, 1976). Active underwater volcanoes also generate low-frequency noise with most energy in the octave band centered near 10 Hz (Northrop, 1974).

D.2.1.4 Thermal Noise

Thermal noise is generated by pressure fluctuations from the thermal agitation (the movement of molecules due to energy transference) of water molecules. It is the remaining noise when all other sources are removed and provides a threshold on the minimum observable noise levels in the ocean. Thermal noise dictates the shape and level of ambient noise spectra above 50 to 100 kHz and causes an increase in ambient noise levels at rate of 6 dB/octave (Urick, 1983).

D.2.2 ANTHROPOGENIC NOISE

Marine species have existed, evolved, and adapted in the presence of naturally occurring noise for millions of years whereas the presence of anthropogenic noise is relatively recent, has intensified in the past century, and caused widespread alterations to the acoustic habitat (Duarte et al., 2021). Noise from human activities is often dynamic and few sources (e.g., shipping) have consistent inputs to the acoustic habitat. Anthropogenic noise varies widely in terms of frequency range, duration, and loudness and can have short-term and localized effects on acoustic habitats, as well as long-term effects over large areas. These characteristics strongly influence any potential impacts on marine species and their acoustic habitats. Prevalent sources of anthropogenic noise discussed in this section include vessel noise, sonar, explosions, and industrial activities.

D.2.2.1 Vessel Noise

Vessel noise is a major contributor to noise in the ocean. Radiated noise from ships varies depending on the size, hull design, type of propulsion, and speed. Ship-radiated noise increases with speed and primarily includes propeller blade tip and sheet cavitation (i.e., low pressure vortices shed by blade tips, and a sheet of bubbles on the back of the blade respectively), and broadband noise from water flowing across the hull (Richardson et al., 1995b; Urick, 1983). Based on these factors, vessel noise can contribute to ocean noise from 10 Hz to 10 kHz (Wenz, 1962). Different classes of vessels have unique acoustic signatures characterized by variances in dominant frequencies. Bulk carrier noise is predominantly near 100 Hz while container ship and tanker noise are predominantly below 40 Hz (McKenna et al., 2012). In comparison, small craft emit higher-frequency noise between 1 kHz and 5 kHz (Hildebrand, 2009).

Globally, commercial shipping is not uniformly distributed. Major shipping lanes typically follow great circle routes or coastlines and go to and from dozens of major ports, and hundreds of small harbors and ports. Most recreational boating occurs in shallow coastal waters whereas military, fishing, and scientific research vessels can be widely distributed (National Research Council, 2003).

Within the Study Area, commercial traffic is heaviest along the U.S. East Coast and the northern coast of the Gulf of Mexico, and follows distinct routes overseas and across the Gulf of Mexico (Mintz, 2012).

Figure D.2-2 highlights commercial routes along the East Coast of the U.S., the Bahamas, and great circle routes in the Atlantic Ocean. Navy vessel traffic in the Study Area (Figure D.2-3) is concentrated along the U.S. East Coast near port facilities, especially around and between the ports of Norfolk, Virginia and Jacksonville, Florida (Mintz, 2012; Mintz, 2016). Commercial traffic (and, therefore, broadband noise generated by it) is relatively steady throughout the year whereas Navy traffic occurs intermittently and is variable in duration. Within the Study Area, Navy vessels represent one percent of overall vessel traffic (Table D.2-1), with the other 99 percent of overall vessel traffic broken down by non-military vessel class in Table D.2-2. In terms of anthropogenic noise, Navy ships are engineered to be as quiet as possible given ship class limitations, and would contribute a correspondingly smaller amount of shipping noise compared to more common commercial shipping and boating (Mintz, 2012; Mintz & Filadelfo, 2011).

Table D.2-1: Overall Vessel Traffic in the Study Area

Vessel Class	*% of Traffic*
Non-Military	99%
U.S. Navy	1%
U.S. Coast Guard	< 1%
Foreign Military	< 1%

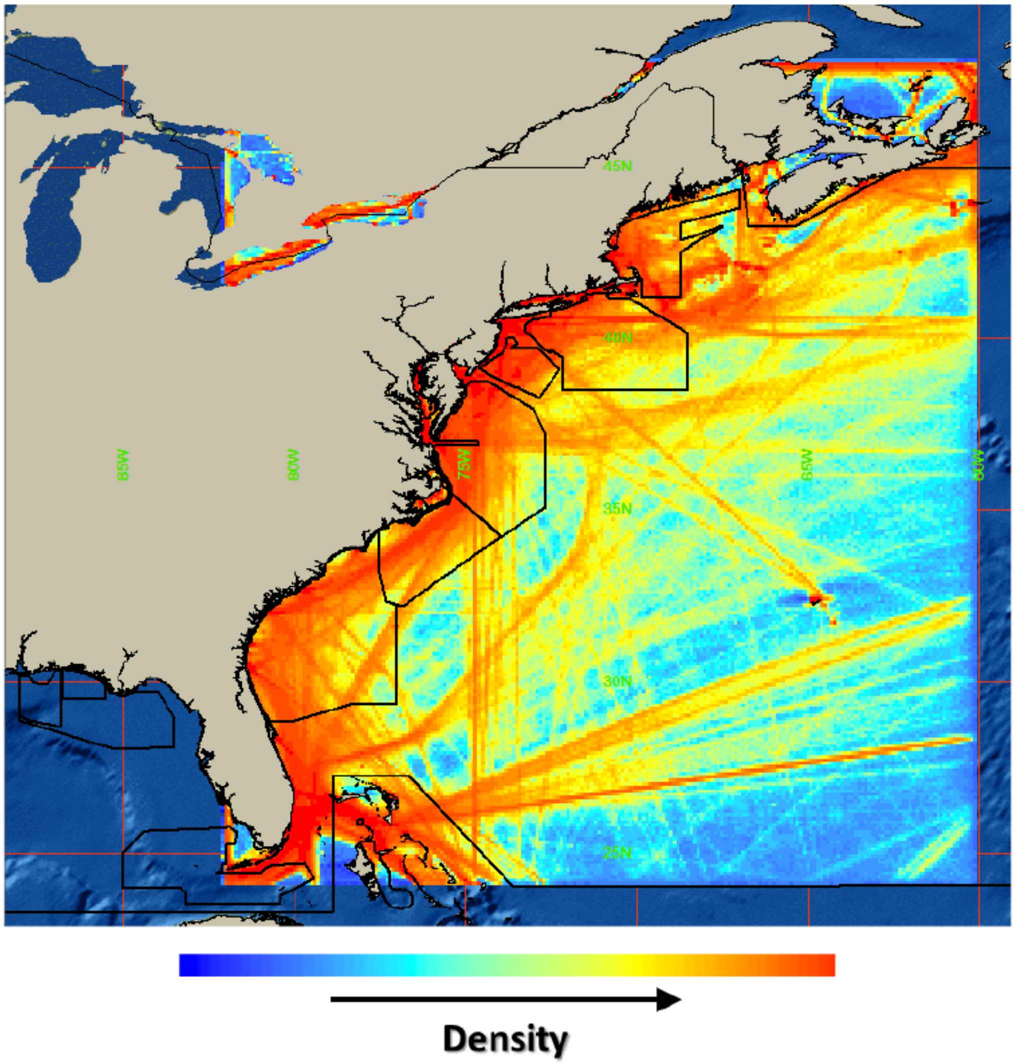
Source: Mintz (2016)
Note: % = percent; < = less than

Table D.2-2: Non-Military Vessel Traffic in the Study Area

Vessel Class	*% of Traffic*
Tugs	24%
Cargo	17%
Other	14%
Fishing	11%
Tanker	11%
Bulk Carriers	9%
Passenger	8%
Service	5%
Research	1%

Source: Mintz (2016)
Notes: % = percent; < = less than

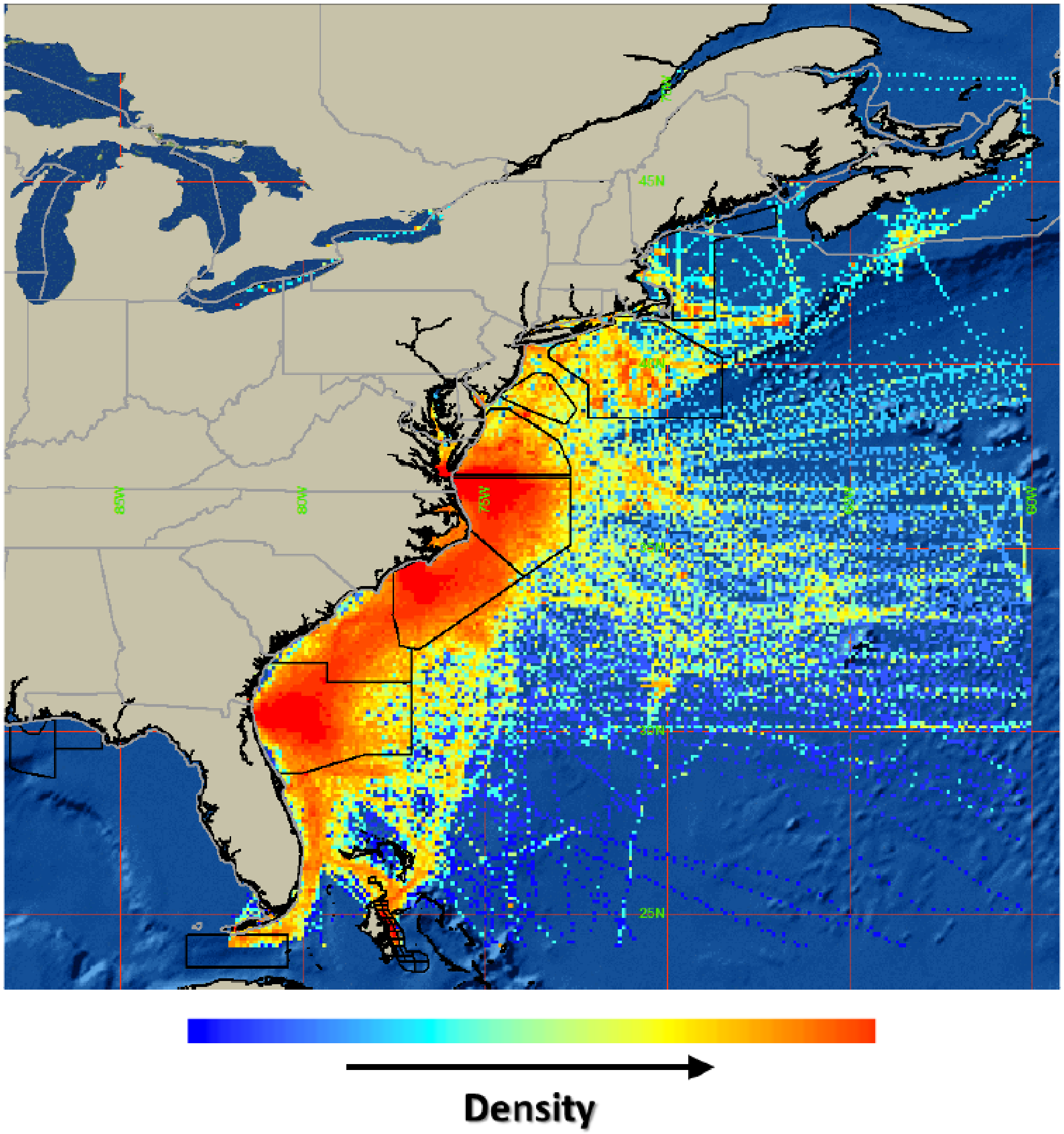
Spectral characteristics of individual ships can be observed at short ranges and in isolated environments. At long ranges, multiple vessels contribute to the overall background noise from ocean traffic in the 10 Hz to 1 kHz band (Figure D.2-1). In shallow water, vessel noise repeatedly interacts with the seafloor and surface and is attenuated by reflection, scattering, and absorption. In deep water, vessel noise propagates downward with fewer interactions with the seafloor and surface and undergoes less attenuation (Erbe et al., 2019). Low-frequency components of vessel noise can propagate long distances in deep water and can travel across ocean basins with minimal energy loss especially within the sound fixing and ranging (SOFAR) channel (Erbe et al., 2019). In areas with sloping bathymetry, vessel noise generated in shallow water can radiate into deeper water due to downward propagation and can couple into the SOFAR channel and propagate long distances (Erbe et al., 2019; Hildebrand, 2009). As a result, vessel noise generated in shallow nearshore waters can still be present in deep offshore waters many kilometers away from the source.



Source: Mintz (2016).

Figure D.2-2: Relative Distribution of Commercial Vessel Traffic in the Study Area

Commercial shipping's contribution to ambient noise in the ocean increased by as much as 12 dB between approximately the 1960s and 2005 and has been attributed to economic growth (Hildebrand, 2009; McDonald et al., 2008). Frisk (2012) confirmed the trend and reported that between 1950 and 2007 ocean noise in the 25 to 50 Hz frequency range has increased 3.3 dB/decade. Assuming a constant baseline level of 52 dB (decibels re 1 $\mu Pa^2/Hz$) during this time results in a cumulative increase of approximately 19 dB. In areas with high levels of shipping traffic, daily average sound levels in the 63 and 125 Hz one-third octave bands were found to be near or higher than 100 dB re 1 μPa (Haver et al., 2021). Daily average sound levels were between approximately 10 to 20 dB higher relative to areas with lower levels of shipping activity (Haver et al., 2021). Temporary reductions in vessel traffic following the events of September 11, 2001 showed an overall decrease of 6 dB (from 50 Hz to 20 kHz), with a notable decrease under 150 Hz (Rolland et al., 2012). Similarly, reduced vessel traffic at the onset of the COVID-19 pandemic resulted in a decrease of 1.5 to 1.7 dB (below 100 Hz) (Breeze et al., 2021; Dahl et al., 2021; Thomson, 2020). Reductions during the COVID-19 pandemic can be attributed to reduced economic activity and shipping (Thomson, 2020); however, noise levels were also subject to local variations such as seasonal environmental conditions and the types of vessels active (Breeze et al., 2021; Dahl et al., 2021).



Source: Mintz (2016).

Figure D.2-3: Relative Distribution of U.S. Navy Vessel Traffic in the Study Area

D.2.2.2 Sonar

Active sonar and other transducers emit non-impulsive sound waves into the water to detect objects, safely navigate, and communicate. The contribution of sonar to the acoustic habitat is highly varied and depends on source characteristics (e.g., frequency, source level, directionality, and duty cycle) and factors that affect sound propagation (e.g., temperature, salinity, pressure, and bathymetry). Temporal and spatial usage are also highly varied and can range from minutes to approximately a month, and from tens to hundreds of kilometers (National Research Council, 2003). Frequency ranges for categorizing sonars are relative, and generalized divisions that are commonly used include: low-frequency (less than 1 kHz), mid-frequency (1 to 10 kHz), high-frequency (10 to 100 kHz), and very high-frequency (greater than 100 kHz) (National Research Council, 2003). Given appreciable differences in usage and source characteristics, the contribution of sonar to the acoustic habitat is distinguished between military and commercial sonar systems.

Military sonar systems encompass all three frequency divisions and includes sources with wider beam widths and higher source levels compared with commercial sonar systems. Spatial and temporal usage is well defined both in terms of hours of operation, and the locations where activities occur. Activities are episodic and can last from hours, days to weeks, and over a month (National Research Council, 2003). Examples of military specific applications include low-frequency surveillance sonar, mid-frequency tactical sonar, and high-frequency sonar from weapons and countermeasures.

Compared with military sonar systems, commercial sonar systems use higher frequency signals, have lower source levels, narrower beam patterns that are downward directed, shorter pulse lengths, and are typically operated for minutes to days (National Research Council, 2003). Usage is widespread across locations and sectors including recreation, fishing, shipping, and research. Sources such as depth finders, multi-beam echosounders, and side-scan sonar are also utilized for military applications. Examples of common commercial sonar systems include depth finders and fish finding sonar (15 to 200 kHz) (Širović et al., 2020), both of which focus sound in a downward beam. Depth finders tend to be used in shallow and nearshore waters for navigation whereas fish finding sonar are operated in both shallow and deep waters. Sea floor mapping for seismic surveys and research utilize multi-beam echosounders (12 to 600 kHz) and side-scan sonar (65 to 500 kHz) (Crocker & Fratantonio, 2016; Ruppel et al., 2022).

D.2.2.3 Explosions

Underwater explosions generate broadband high intensity impulsive sounds that propagate equally in all directions. The spectral and amplitude characteristics of explosions vary with the weight of the charge and the depth of the detonation. Most energy is at lower frequencies from tens to hundreds of Hertz. Explosions are typically localized and propagate tens of kilometers, with the exception of acoustic tomography experiments that measure temperatures and currents over large regions of the ocean and can propagate hundreds to thousands of kilometers (National Research Council, 2003). Military applications of underwater explosives include bombs, mines, missiles, rockets, torpedoes, and projectiles. Spatial and temporal usage under the current action is well defined both in terms of counts of explosives, and the locations where activities occur. Commercial applications of underwater explosives include using explosives as an acoustic sound source for reflection seismology (i.e., rock/sediment penetration and determination) in geophysical exploration (i.e., oil and gas surveys) and for oceanographic research to study underwater acoustic tomography. The use of explosive sound sources for seismic surveys have largely been replaced by air guns due to environmental and handling safety concerns, as well as the lack of control when reproducing signals. Explosives are commonly used for decommissioning marine structures such as offshore oil and gas platforms by severing pilings and conductor pipes at the seafloor (Klima et al., 1988). In addition, small explosive charges known as seal bombs are commonly used by the fishing industry to protect fishing equipment and catch from predation by deterring marine mammals (Krumpel et al., 2021).

D.2.2.4 Industrial Activities

In many areas of the world, oil and gas seismic exploration in the ocean is undertaken using a group of air guns towed behind large research vessels. The air guns convert high-pressure air into very strong shock wave impulses that are designed to return information from the various buried layers of sediment under the seafloor. Most of the impulse energy (analogous to underwater explosions) produced by air guns is heard as low-frequency noise, which can travel long distances, especially in deep water. Most energy is below 200 Hz with additional energy extending to the kilohertz range (Greene & Richardson, 1988; Ruppel et al., 2022). Similar to air guns, other sources that generate an impulse for sub-bottom profiling include: boomers, which use an actuator to displace a near-surface and downward oriented metal plate; sparkers, which discharge a high voltage electric field to vaporize salt water; and bubble guns, which compress air within a plate or pair of plates (Crocker & Fratantonio, 2016; Ruppel et al., 2022). In the Study Area, seismic surveys are prevalent in the Gulf of Mexico Outer Continental Shelf, on the Atlantic Ocean Outer Continental Shelf from Delaware Bay to south of Cape Canaveral, and from the inner edge of federal waters to 403 miles offshore (Bureau of Ocean Energy Management, 2014). Seismic exploration surveys can encompass areas from tens of kilometers to over one hundred kilometers, and last from days to months (National Research Council, 2003).

The operation of offshore oil and gas extraction platforms produces nearly continuous noise primarily from 20 to 1,000 Hz (Greene & Richardson, 1988) and includes ancillary noise from support vessels and machinery. Oil and gas extraction is typically conducted on offshore platform rigs, drill ships, or artificial islands. Emplacement of permanent structures produces localized noise and lasts for weeks (National Research Council, 2003). Drill ships are generally the loudest with most broadband energy between 10 Hz and 10 kHz (Richardson et al., 1995b). This is because internal ship noise from machinery is effectively transmitted through the hull, and from the use of thrusters for dynamic positioning during drilling operations.

Pile driving is conducted for construction of nearshore structures such as piers, and for offshore structures including wind farm turbines and oil and gas platforms. Installing piles uses an impact hammer which results in an impulsive sound emanating from the length of the pile into the water column as well as from the bottom of the pile through the sediment. Because the impact wave travels through a steel pile at speeds faster than the speed of sound in water, a steep-fronted acoustic shock wave is formed in the water (Reinhall & Dahl, 2011). Piles can also be installed by vibratory pile driving and removed by vibratory extraction, which generates continuous non-impulsive noise with peak pressures lower than impact pile driving. Sound levels can vary depending on the size and power level of the equipment, pile material and diameter, and seafloor sediment type. Installation and removal can encompass areas from less than one kilometer to hundreds of kilometers, and near-continuous activity can last from days to months (National Research Council, 2003).

In the Study Area, the development of offshore wind farms is prevalent in waters between Massachusetts and North Carolina. The construction of offshore wind farms can take weeks to months to complete and produces localized low-frequency noise less than 2 kHz (Amaral, 2020). Most construction noise is produced from pile driving with ancillary noise from laying cable and support vessels. During operation, wind farms produce continuous low-frequency underwater noise primarily below 1 kHz, with tonals between 20 and 330 Hz (Pangerc et al., 2016).

D.3 VEGETATION

There is no evidence that underwater acoustic stressors impact marine vegetation under conditions they would experience with the Proposed Action. While there is at least one recent study (Solé et al., 2021) that documented a negative and non-trivial impact of sound on seagrass, the sound exposure was continuous for two hours at low frequencies and high intensities (157 to 175 dB re 1 µPa) in an artificial/tank environment (refer to Section D.1.5, Acoustic Propagation in Small Tanks, for context). The results of this study are not applicable to the proposed action due to either lack of proximity and/or duration of exposure. A prolonged exposure to sounds would only occur around highly modified pierside locations devoid of seagrass habitat. Elsewhere in the Study Area, seagrass may be exposed to distant and highly mobile sources of sound, but there would be no meaningful effects. Acoustic stressors were discounted for vegetation in the 2018 *Final Atlantic Fleet Training and Testing Environmental Impact Statement/Overseas Environmental Impact Statement* (hereinafter referred to as the 2018 Final EIS/OEIS) Section 3.3.3.1 (Acoustic Stressors) and continue to be discounted for Section 3.4 (Vegetation).

D.4 INVERTEBRATES

The synthesis of information regarding acoustic stressors effects on marine invertebrates has not changed appreciably from what is described in the 2018 Final EIS/OEIS. The key synthesis papers recently published (Ferrier-Pages et al., 2021; Murchy et al., 2019; Solé et al., 2023c; Wale et al., 2021) mention most of the same studies referenced in the 2018 Final EIS/OEIS Section 3.4.3.1 (Acoustic Stressors), and there is no recent research that suggests an escalation of noise effects on marine invertebrates under the mostly temporary and localized exposure they would experience from the Proposed Action stressors. Most of the studies were conducted on bivalves, cephalopods, and crustaceans. Relative few were conducted on gastropods, bryozoans, echinoderms, cnidarians, tunicates, and zooplankton.

Despite claims that anthropogenic noise effects on invertebrates are a direct indicator of ocean health (Solé et al., 2023c), the presentation of information does not support the claim. The synthesis papers often neglect to report the duration of exposures and ranges at which the stated effects occurred. They also mostly neglect to report details about the recovery of invertebrates from the stated effects. To address this deficiency, a summary and evaluation of the updated literature is presented under the following subsections: Sound Sensing and Production, Hearing Loss and Auditory Injury, and Masking.

With regards to Endangered Species Act (ESA)-listed or ESA-proposed invertebrates, recent literature was published for the taxonomic group that includes queen conch (gastropods). The information for acoustic and explosive stressor effects on corals in the 2018 Final EIS/OEIS remains valid for this Supplemental EIS/OEIS.

D.4.1 HEARING LOSS AND AUDITORY INJURY

The following studies provided updated information regarding the potential for temporary hearing loss among invertebrates exposed to acoustic stressors.

- Solé et al. (2021) investigated the effect of sinusoidal wave sweeps on a gastropod (apple snail, *Pomacea maculate*) and found damage to the statocysts. However, the experiment was conducted in tanks with exposures to 40 to 400 Hz sweeps at 157 dB plus or minus 5 dB re 1 µPa with peak SPL up to 175 dB re 1 µPa for two hours.
- Solé et al. (2023a) investigated the effect of artificial sound on the sea anemone (*Calliactis parasitica*) and its host the red hermit crab (*Dardanus calidus*) and found damage to statocysts.

However, the experiment was conducted in tanks with exposures to continuous 50 to 400 Hz frequencies at 157 plus or minus 5 dB re 1 μPa^2 with peak levels up to 175 dB re μPa^2 for two hours.

- Solé et al. (2023b) investigated the effect of natural sounds and sinusoidal wave sweeps on blue crabs and found some damage to sensory statocysts. However, the experiment was conducted in a tank maze with exposures to continuous 100 to 500 Hz sinusoidal wave sweeps at 171 dB re 1 μPa with peak levels up to 180 dB re 1 μPa for two hours.

These experiments and their results are consistent with the studies on squid, octopus, crabs, zooplankton, and jellyfish presented in the 2018 Final EIS/OEIS. The effects were from prolonged noise exposure at mostly very close ranges (e.g., tank dimensions) and attributable primarily to particle motion. Noise exposure experiments in small tanks and at a close distance are complicated (see Section D.1.5, Acoustic Propagation in Small Tanks). Small tanks are not representative of the natural environment because of how noise travels and interacts within the walls of the tank. While it is believed that invertebrates primarily detect particle motion (see Section D.1.4 Auditory Perception), an individual has to be in the near field of the sound source (see Section D.1.2.4, Particle Motion) to be impacted by the noise. In reality, very few individuals will be close enough to the sound source for them to be impacted in the ways suggested by these experiments. Not to mention that the small tanks may even amply the effects well beyond what would be realistic in the natural environment.

D.4.2 MASKING

The following studies provided updated information regarding the masking effect of acoustic stressors on invertebrates.

- Anderson et al. (2021) evaluated the premature settlement response of reef-associated sessile invertebrate larvae to simulated anthropogenic noise at 1, 10, 25, 50, 100, 200, and 500 m. The noise source characteristics were recordings of pure tones of known frequency (100, 500, 1,000, 1,500, and 2,000 Hz) amplified to 180 dB re 1 μPa and played for 30 minutes at each distance. The After five consecutive nights of collecting invertebrate larvae settling on the bottom in otherwise quiet shallow-water reef environment, the effect of anthropogenic sound was considered not ecologically significant by the principal investigators. The results provide support for the general conclusion in the 2018 Final EIS/OEIS that the masking effect of Proposed Action sounds reaching shallow-water reef environments is likely negligible.
- Jézéquel et al. (2021) evaluated the sound production response of caged American lobsters to actual vessel noise. The noise was recorded in close proximity to their cages. The result indicates that lobsters significantly increase their call rate in the presence of shipping noise. The recordings included both periods of ambient soundscape properties (1 to 2 kHz with a mean SPL of 102.3 plus or minus 4 dB re 1 μPa) and periods of vessel activity (up to 146.2 dB re 1 μPa). Masking was assumed to some degree for lobster and other sound producing invertebrates in the 2018 Final EIS/OEIS, but the effect of the new information on the qualitative analysis is negligible.

D.4.3 BEHAVIORAL REACTIONS

The following studies provided updated information regarding the behavioral reaction of invertebrates to acoustic stressors.

- Charifi et al. (2018) investigated the effect of cargo ship noise and uptake of trace metal contaminants on Pacific oysters (*Magallana gigas*) and found decreased valve activity, resulting in lower metal contamination and decreased growth, with exposure to ship noise relative to no ship noise. However, the study was conducted in a tank environment with recorded vessel noise (see Section D.1.5, Acoustic Propagation in Small Tanks). The noises recorded and played back were from cargo vessels approximately 500 m away that exposed the test organisms to 92 replicates of 12-minute noise exposures per day for 3 days.

- Hudson et al. (2022) investigated the effect of vessel noise and mid-frequency sonar on blue crabs (*Callinectes sapidus*) and found changes in competitive behavior. However, the experiment was conducted in tanks with exposures of 50 minutes to either sonar noise of 1.67 and 2.5 to 4 kHz at 177 to 182 dB re 1 µPa or vessel noise of 60 Hz to 1 kHz at 169 to 172 dB re 1 µPa.
- Jézéquel et al. (2022) evaluated the response of giant scallops (*Placopecten magellanicus*) to pile driving nose. The pile driving noise employed mimicked what happens during construction of off-shore wind turbines and induced repeated valve closure in giant scallops located 8 to 50 m from the driven pilings. The tested individuals returned to normal behavior after negligible acclimation to the episodic noises. Valve closure with pile driving was referenced in the 2018 Final EIS/OEIS, though little was said about the acclimation and return to normal behavior after the noise stopped.
- Solé et al. (2023b) investigated the effect of natural sounds and sinusoidal wave sweeps on blue crabs and found no impact on behavior from either treatment. However, the experiment was conducted in a tank maze with exposures of two hours to 100 to 500 Hz sinusoidal wave sweeps at 171 dB re 1 µPa with peak levels up to 180 dB re 1 µPa.

These experiments and their results are consistent with the studies on various crustaceans, squid, bivalves, and sea stars presented in the 2018 Final EIS/OEIS. The effects were from prolonged exposure at mostly very close ranges (e.g., tank dimensions) and attributable primarily to particle motion.

D.4.4 PHYSIOLOGICAL RESPONSE

The following studies provide updated information regarding the physiological responses of invertebrates to acoustic stressors.

- Charifi et al. (2018) investigated the effect of cargo ship noise and uptake of trace metal contaminants on Pacific oysters (*Magallana gigas*) and found lower growth rates (2.6 times slower) with exposure to ship noise relative to no ship noise. However, the study was conducted in a tank environment with recorded vessel noise. The noises recorded and played back were from cargo vessels approximately 500 m away that exposed the test organisms to 92 replicates of 12-minute noise exposure per day for 3 days.
- Wale et al. (2019) investigated the effect of ship noise playbacks on blue mussels and found higher breaks in the deoxyribonucleic acid (DNA), lower algal clearance rates and higher oxygen consumption rates. However, the study was conducted in a tank environment with recorded vessel noise. The noises recorded and played back were from ships approximately 200 to 300 m away that exposed the test organisms to a peak SPL of 150 to 155 dB re 1 μPa^2 relative to 85 to 95 dB re 1 μPa^2 during control conditions. The duration of passing vessel noise during the six hours of treatment exposure was not specified.
- Vazzana et al. (2020a) investigated the effect of linear chirp playback on Mediterranean mussel (*Mytilus galloprovincialis*) and found changes in biochemical and immunological parameters in their digestive glands. However, the experiment was conducted in tanks with continuous exposures of three hours to 100 to 200 kHz chirps at 145 to 160 dB re 1 µPa rms.

- Vazzana et al. (2020b) investigated the effect of playback noise on blue mussels (*Mytilus edulis*) and found changes in biochemical and immunological parameters in their digestive gland. However, the experiment was conducted in tanks with continuous exposures of three hours to 100 to 200 kHz noise at 145 to 160 dB re 1 µPa.
- Zhao et al. (2021) evaluated the effect of 10 days of simulated pile driving noise at received sound intensities of approximately 70 and 100 dB re 1 µPa on blue mussel attachment tissue. The results document a significant reduction in the secretion of attachment tissue and resulting strength of shell attachments. Whereas this may seem to contradict the observations reported in Horton (2016) that healthy shellfish populations exist around Navy piers subjected to regular piling replacement, the effects were observed after more continuous pile driving than is being proposed; the simulated noise was played on a loop for 10 days which does not simulate the sporadic nature of actual pile driving noise.
- Hudson et al. (2022) evaluated the response of blue crabs (*Callinectes sapidus*) and American lobsters (*Homarus americanus*) to simulated vessel and sonar noise. After 50-minute exposures to less than 1 kHz simulated vessel noise at 169 to 172 dB re 1 µPa (receptor exposure at less than 500 m from mid-sized container vessel), there was no effect on blue crab or lobster mortality and stress chemicals returned to normal level after 48 hours. However, there was evidence of elevated stress chemicals for seven days after blue crabs were exposed to simulated sonar for the same duration. The exposures were to one-second 1.67 kHz continuous wave pulses following by 2.5 to 4 kHz one-second chirps at 177 to 182 dB re 1 µPa (receptor exposure at some unspecified distance from Navy sonar). The results expand upon those of Dossot et al. (2017) that is referenced already in the 2018 Final EIS/OEIS, but they are not substantively different in terms of the qualitative factors employed for analysis.
- Olivier et al. (2023) investigated the effect of pile driving and drilling playback on king scallop (*Pecten maximus*) and found less than four percent mortality rates without any noise influence and an interactive impact on postlarval growth between trophic environment and noise level/spectra and no change in fatty acid profiles. However, the experiment was conducted in tanks with 6- and 19-hour exposures to continuous pile driving and drilling noise, respectively. The intensity of treatment sounds ranged from 147 to 187 dB re 1 µPa p-p for pile driving and 107 to 175 dB re 1 µPa rms for drilling.

These experiments and their results are consistent with the studies on various crustaceans and bivalves presented in the 2018 Final EIS/OEIS. The effects were from prolonged exposure and mostly at very close ranges (e.g., tank dimensions) and attributable primarily to particle motion.

D.4.5 LONG-TERM CONSEQUENCES

The following studies provide updated information regarding the long-term consequences of acoustic stressors on invertebrates.

- Hubert et al. (2022) evaluated the long-term response of blue mussels (*Mytilus edulis*) to repeated exposure with simulated impulsive noise. The results suggested habituation or sensory adaptation to the stressor. Habituation to noise was not referenced specifically in the 2018 Final EIS/OEIS, but the possibility of habituation was already included in the qualitative analysis.
- Jézéquel et al. (2023) exposed groups of longfin squid (*Doryteuthis pealeii*) in cages at multiple distances from consecutive pile driving events and quantified responses. Whereas there as a short-term alarm response to sound levels of 112 to 123 decibels referenced to 1 micrometer

per second squared (dB re 1 µms^{-2}) (zero-peak), the individual rate of response quickly decreased within and across pile driving events. This is consistent with the information presented in the 2018 Final EIS/OEIS for long-term consequences.

D.5 FISHES

D.5.1 HEARING AND VOCALIZATIONS

All fishes have two sensory systems that can detect sound in the water (Popper et al., 2019; Popper & Schilt, 2008; Schulz-Mirbach et al., 2020). The first system discussed herein is the lateral line, which consists of a series of neuromasts (i.e., receptors) along the body that are directly exposed to the environment. When a vibration occurs within the water column that reaches the fish, the receptors along the laeral line move and this movement is transferred through the nervous system to the brain, where it is interpreted. These receptors are sensitive to external particle motion, specifically at frequencies up to 400 Hz (Coombs & Montgomery, 1999; Hastings & Popper, 2005; Higgs & Radford, 2013; Webb et al., 2008), created by sources within a few body lengths of an animal (i.e., in the near field, see Section D.1.2.4, Particle Motion, for additional information).

The second sensory system is the inner ear. The inner ear in fishes functions similarly to the inner ear in other vertebrates. Generally, the inner ears of bony fishes contain three dense otoliths (i.e., small calcareous bodies, although some fishes may have more) that sit atop many delicate mechanoelectrical hair cells within the inner ear. Underwater sound waves pass through the fish's body due to different structural densities (i.e., soft tissue versus bone) and vibrate the otoliths. As a result, sound waves cause relative motion between the dense otoliths and the surrounding tissues, causing movement of the hair cells back and forth, which is sensed by the nervous system like the stimulation of the receptors along the lateral line. Note, the inner ears are directly sensitive to acoustic particle motion like sensory receptors along the lateral line rather than acoustic pressure. However, some fishes possess morphological adaptations or specializations that can enhance their sensitivity to sound pressure, such as a gas-filled swim bladder (Astrup, 1999; Popper & Fay, 2010). The swim bladder can enhance sound detection by converting acoustic pressure into localized particle motion, which may then be detected by the inner ear (Radford et al., 2012). Fishes with a swim bladder generally have greater auditory sensitivity and can detect higher frequencies than fishes without a swim bladder (Popper & Fay, 2010; Popper et al., 2014). In addition, some fishes contain small horn-like projections that can either partially or fully connect the swim bladder and the inner ear increasing sensitivity and allowing for higher frequency detection (up to a few kilohertz or higher for some species) and better sound pressure detection (e.g., Vetter & Sisneros, 2020). For simplicity and consistency with terminology used in other taxa sections within this SEIS, and peer-reviewed research, acoustic detection capabilities by either sensory system will generally be described as 'hearing' throughout this discussion.

Propagating sound waves contain pressure and particle motion components but, particle motion is most prominent at low frequencies and is most detectible at high-sound pressures or very close to a sound source. Historically, studies have investigated acoustic detection (e.g., hearing research) and its effects on fishes. However, when exposed to a sound, often only sound pressure is measured and not particle motion. Although particle motion may be the more relevant exposure metric, few data are available that actually measure particle motion due to a lack of standard methodology and experience with particle motion detectors (Hawkins et al., 2015; Martin et al., 2016). In these instances, particle motion can be estimated from pressure measurements (Nedelec et al., 2016a). Similarly, although the lateral line likely plays a significant role in a fish's auditory capabilities, this portion of the sensory system is not always included in hearing experiments. Due to the limited research on lateral line sound detection, the remainder of this section will be focusing on inner ear sound detection.

Although many researchers have investigated acoustic detection in fishes (Ladich & Fay, 2013; Popper et al., 2014), hearing data (i.e., audiograms) only exist for just over 100 of the estimated 36,000 species of fish worldwide (Fricke et al., 2020). Therefore, fish categories are defined by species that possess a similar continuum of anatomical features, which result in varying degrees of estimated acoustic detection capabilities (Popper & Fay, 2010; Popper & Hastings, 2009b; Schulz-Mirbach et al., 2020; Stanley et al., 2020; Wiernicki et al., 2020). Specifically, fishes with specialized adaptations connecting the swim bladder to the inner ear have traditionally been categorized as “hearing specialists,” while fishes that do not possess specialized structures or swim bladders have been referred to as “hearing generalists” (Popper et al., 2003). Specialists can detect a wide range of frequencies at lower sound levels (i.e., auditory thresholds) compared to generalists that typically detect a much narrower range of frequencies at higher sound levels. Categories and descriptions of the general acoustic detection capabilities for these groups are further defined in Table D.5-1 (modified from Popper et al., 2014). Additional research is still needed to better understand species-specific frequency detection capabilities and continues to help clarify how various anatomical features interact within the auditory system and influence overall sensitivity to sound.

Table D.5-1: Fish Hearing Groups and Categories

Hearing Group	***Fish Category***	***Description***
Hearing Generalists	Fishes without a swim bladder	Acoustic detection capabilities are limited to particle motion detection at frequencies well below 2 kHz (e.g., sharks, rays, and halibut).
	Fishes with a swim bladder not involved in hearing	Fishes lack notable anatomical specializations and primarily detect particle motion at frequencies below 2 kHz (e.g., salmonids, sturgeon, and groupers).
Hearing Specialists	Fishes with a swim bladder involved in hearing	Fishes can detect frequencies below 2 kHz, possess anatomical specializations to enhance hearing, and are capable of sound pressure detection up to a few kHz (e.g., herring, sardines, anchovy).
	Fishes with a swim bladder and with high-frequency hearing	Fishes possess anatomical specializations and are capable of sound pressure detection at frequencies up to 10 kHz, or over 100 kHz in some species (shad and menhaden).

Note: kHz = kilohertz

Data suggest that most species of marine fish are hearing generalists and either lack a swim bladder (e.g., sharks and flatfishes) or have a swim bladder not involved in acoustic detection (e.g., sturgeon and codfishes) and can only detect sounds below 2 kHz. Fewer marine fishes (Clupeiformes) are hearing specialists (i.e., those with a swim bladder involved in hearing). These species can detect sounds up to about 4 kHz (Colleye et al., 2016; Mann et al., 2001; Mann et al., 1997; Mickle & Higgs, 2021). One subfamily of clupeids (i.e., Alosinae or shads) can detect high- and very high-frequency sounds (i.e., frequencies from 10 to 100 kHz, and frequencies above 100 kHz, respectively), although sensitivity at these higher frequencies are elevated and the range of best sensitivity is still in the low-frequency range (below 1 kHz) like other fishes. It was theorized that this subfamily may have evolved the ability to hear relatively high sound levels at these higher frequencies to detect echolocation signals of nearby foraging dolphins

(Mann et al., 1998; Mann et al., 1997). For fishes that have not had their hearing tested, such as deep sea fishes, the suspected hearing capabilities are based on the structure of the ear, the relationship between the ear and the swim bladder, and other potential adaptations such as the presence of highly developed areas of the brain related to inner ear and lateral line functions (Buran et al., 2005; Deng et al., 2011, 2013). It is believed that most fishes have their best sensitivity from 100 to 400 Hz (Popper et al., 2003). Seasonal variations in auditory sensitivity have been reported in some fishes, such as the plainfish midshipman, which have likely evolved to aid in reproductive behaviors (i.e., detection of suitable mates) (e.g., Rogers et al., 2022; Sisneros & Bass, 2003).

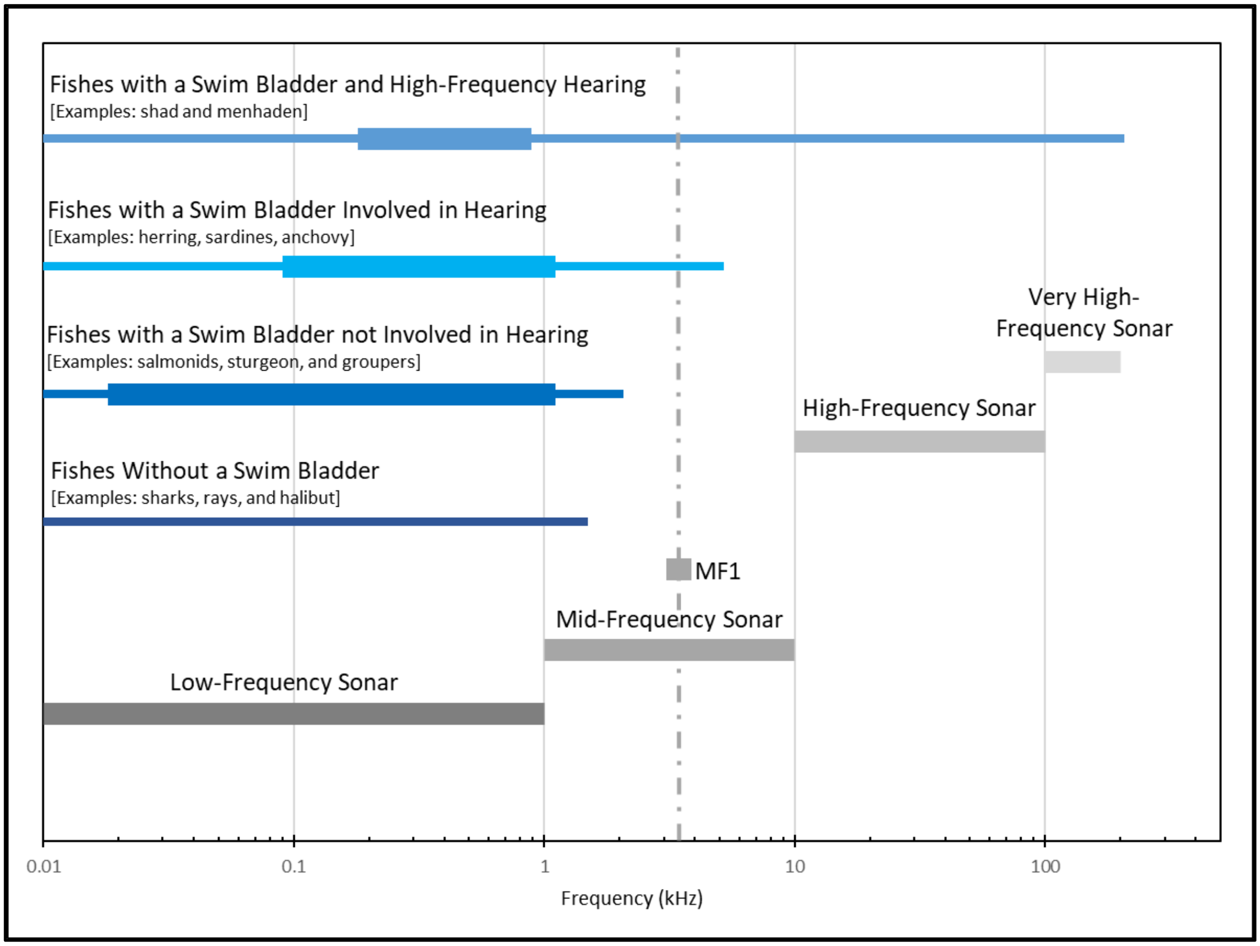
Bony fishes can produce sounds in several ways and use them for a variety of behavioral functions (Kasumyan, 2009; Ladich, 2008, 2014). The most common mechanism for sound production is when the swim bladder and other structures (often muscles that are associated with the swim bladder wall) vibrate and radiate sound into the water (Zelick et al., 1999). Additional mechanisms include, but are not limited to, muscular vibrations, rubbing, or plucking of pectoral fins (including the girdle, spines, or tendons) and grinding or rubbing of teeth, jaw apparatuses, or even bones in the skull (Kasumyan, 2008; Ladich, 2008). Over 30 families of fishes are known to produce acoustic signals in aggressive interactions, and over 20 families of fishes vocalize during courtship or mating (Ladich, 2008). Sounds generated by fishes as a means of communication are generally narrow band and below 500 Hz, though some acoustic signals have been recorded at frequencies up to 5,000 Hz (Kasumyan, 2008; Ladich, 2000; McCauley & Cato, 2000; Slabbekoorn et al., 2010). Acoustic signals may vary in source level depending on factors such as the sound production mechanism, species, size of fish, behaviors associated with the signal, and even environmental factors (Kasumyan, 2009). Some of the loudest recorded vocalizations are from fish choruses with approximate source levels up to 170 dB re 1 µPa (Erisman & Rowell, 2017; McCauley & Cato, 2000; McIver et al., 2014; Sisneros & Bass, 2003; Sprague & Luczkovich, 2004).

Combined research methods that utilize visual surveys (such as baited underwater video and monitoring by divers) and passive acoustic monitoring continue to reveal new sounds produced by fishes both in the marine and freshwater environments. Such experiments allow for specific behaviors to be paired with recorded signals, the identification of sex-specific vocalizations, and may be useful in determining more approximate estimates of the total number of soniferous (e.g., sound producing) fishes in a given habitat (Bussmann, 2020; Parmentier et al., 2021; Radford et al., 2018; Rountree et al., 2018; Rowell et al., 2020; Rowell et al., 2018).

D.5.1.1 Detection of Navy Sonars

As described above, fishes are not equally sensitive to noise at all frequencies. Figure D.5-1 provides a general summary of hearing threshold data from available literature (Casper & Mann, 2006; Deng et al., 2013; Kéver et al., 2014; Mann et al., 2001; Ramcharitar et al., 2006) to demonstrate the potential overlap of frequency detection for each fish category with Navy sonars. Fishes from all categories can detect broadband sound sources such as explosives or vessel noise. But, as displayed, not all fishes would detect some frequency-limited sources, such as high-frequency sonar.

Due to data limitations, these estimated ranges of frequency detection may be overly conservative in that they may extend beyond what some species within a given fish hearing category may detect. For example, although many sharks are most sensitive to lower frequencies, well below 1 kHz, the bull shark (*Carcharhinus leucas*) can detect frequencies up to 1.5 kHz (Kritzler & Wood, 1961; Myrberg, 2001), representing the uppermost known limit of frequency detection for this hearing category. These upper bounds of each fish category's frequency range are outside of the range of best sensitivity for most fishes within that category. As a result, fishes within each group would only be able to detect those upper frequencies at close distances to the source, and from sources with relatively high source levels.



Notes: Thin blue lines represent the estimated minimum and maximum range of frequency detection for each fish category. All groups are assumed to detect frequencies down to 10 Hz regardless of available data. Thicker portions of each blue line represent the estimated minimum and maximum range of best sensitivity for that group. Not all fishes within a given category would be able to detect all the displayed frequencies. For example, flatfish such as halibut can only detect frequencies up to 270 Hz, although other fishes in the same hearing group can detect much higher frequencies (e.g., bull sharks can detect up to 1,500 Hz, the upper limit of the hearing group). Each sonar source class that occurs in the Study Area is represented graphically by the horizontal grey bars. Not all sources within each class would operate at all the displayed frequencies and may not overlap all fish hearing groups as demonstrated by the dotted grey line. kHz = kilohertz, MF1 = center frequency of 3.5 kHz.

Sources: Casper and Mann (2006); Chapman and Hawkins (1973); Chapman and Sand (1974); Hawkins and Johnstone (1978); Mann et al. (2005); Popper (2008); Popper et al. (2007); Tavolga and Wodinsky (1963)

Figure D.5-1: Fish Categories and Navy Sonars

D.5.2 HEARING LOSS AND AUDITORY INJURY

Impairment of auditory detection (more commonly referred to as hearing loss) or auditory injury will have an immediate effect on an animal's ability to detect certain frequencies. For this reason, hearing loss and auditory injury are often discussed together. However, the sensory hair cells of the inner ear and lateral line in fishes are regularly replaced over time when they are damaged, unlike in mammals where sensory hair cells loss is permanent (Lombarte et al., 1993; Popper et al., 2014; Smith et al., 2006). Consequently, PTS has not been known to occur in fishes, and any hearing loss in a fish may be as temporary as the timeframe required to repair or replace the sensory cells that were damaged or destroyed (Popper et al., 2014; Popper et al., 2005; Smith et al., 2006).

Available data for some terrestrial mammals have shown signs of auditory injury in the form of nerve damage after severe threshold shifts (e.g., Kujawa & Liberman, 2009; Lin et al., 2011). In fishes, studies have observed cellular changes in hearing structures after long-term sound exposures (Sapozhnikova et al., 2020), as well as hair cell damage and tearing of the epithelial lining after exposure to underwater detonations at close range (Smith et al., 2022). However, it is not known if physical damage such as those described here would be recoverable in fishes (like hair cell regeneration noted in other studies), or if there would be hearing impairment. One of the few studies to look at both auditory sensitivity (measured by threshold shifts) and potential physical damage to the inner ear include research using zebrafish (*Danio rerio,* a freshwater hearing specialist) (Breitzler et al., 2020). The experiment demonstrated a lack of damage to sensory receptors when temporary threshold shift (TTS) occurred though this has not been investigated in marine species (for additional details on the results of this experiment, see D.5.2.2, Threshold Shift due to Vessel Noise).

D.5.2.1 Threshold Shift due to Sonar and Other Transducers

Several studies have examined the effects of the sound exposures from low-frequency sonar on fish hearing (Halvorsen et al., 2013; Kane et al., 2010; Popper et al., 2007). Hearing was measured both immediately post exposure and for up to several days thereafter (Halvorsen et al., 2013; Kane et al., 2010; Popper et al., 2007). Maximum SELs were 218 or 220 dB re 1 μPa^2s at frequencies ranging from 170 to 320 Hz (Kane et al., 2010; Popper et al., 2007) and 215 dB re 1 μPa^2s in a follow-on study (Halvorsen et al., 2013). Two hearing generalists, the largemouth bass (*Micropterus salmoides*) and yellow perch (*Perca flavescens*), showed no loss in detection sensitivity from sound exposure immediately after the test or 24 hours later. Channel catfish (*Ictalurus punctatus*), a hearing specialist, and some specimens of rainbow trout (*Oncorhynchus mykiss)*, a hearing generalist, showed a threshold shift (up to 10–20 dB) immediately after exposure to the low-frequency sonar when compared to baseline and control animals. Small thresholds shifts were detected for up to 24 hours after the exposure in some channel catfish. Although some rainbow trout in one test group showed signs of TTS, rainbow trout in another group showed no TTS. Catfish hearing returned to normal within about 24 hours after exposure to low-frequency sonar. Examination of the inner ears of the fish during necropsy revealed no differences from the control groups in ciliary bundles or other features indicative of hearing loss.

The same investigators examined the potential effects of mid-frequency active sonar on rainbow trout and channel catfish hearing (Halvorsen et al., 2012c; Kane et al., 2010). The maximum received cumulative SEL was 220 dB re 1 μPa^2s. No significant TTS was observed in rainbow trout. Of the catfish tested, only the group tested in October experienced TTS (6.35 dB), which recovered within 24 hours, but fish tested in December showed no TTS (Halvorsen et al., 2012c; Kane et al., 2010).

Some studies have suggested that there may be some loss of sensory hair cells due to high intensity sources, indicating a possible loss in hearing sensitivity; however, none of those studies investigated the subjects' actual hearing range after exposure to these sources. Enger (1981) found loss of ciliary bundles of the sensory cells in the inner ears of Atlantic cod (*Gadus morhua*), hearing specialists, following one to five hours of exposure to pure tone sounds between 50 and 400 Hz with a SPL of 180 dB re 1 µPa.

Hastings (1995) found auditory hair-cell damage in goldfish (*Carassius auratus*), a freshwater hearing specialist. Goldfish were exposed to 250 Hz and 500 Hz continuous tones with maximum peak SPLs of 204 dB re 1 µPa and 197 dB re 1 µPa, respectively, for about two hours. Similarly, Hastings et al. (1996) demonstrated damage to some sensory hair cells in oscars (*Astronotus ocellatus*) observed one to four days following a one-hour exposure to a pure tone at 300 Hz with an SPL of 180 dB re 1 µPa. Both studies found a relatively small percentage of total hair cell loss from hearing organs despite long-duration exposures. Effects from long-duration noise exposure studies are generally informative; however, they are not necessarily representative of effects from intermittent, short-duration exposures produced during Navy activities involving sonar and other transducers.

As noted in the *American National Standards Institute (ANSI) Sound Exposure Guideline* technical report (Popper et al., 2014), some hearing specialists may be more susceptible to TTS from high-intensity, non-impulsive sound sources, such as sonar and other transducers, depending on the duration and frequency content of the exposure. Fishes that are hearing specialists may exhibit TTS from exposure to low- and mid-frequency sonar, specifically at cumulative SELs above 215 dB re 1 µPa^{2}s. However, hearing generalists would be unlikely to detect mid- or other high-frequency sonars and would likely require a much higher SEL to exhibit the same effect from exposure to low-frequency active sonar.

D.5.2.2 Threshold Shift due to Vessel Noise

There are only a few studies on the effects of vessel noise on hearing in fishes. For example, Rogers et al. (2020) examined the effects of vessel noise playbacks on the oyster toadfish, a hearing generalist. Toadfish were exposed to one of three noise conditions and hearing thresholds were measured before and multiple days (up to 9) after exposure. Two groups of fish were exposed to recorded boat noise (30 to 12,000 Hz frequency range) for either 1 or 12 hours continuously, and a third group was exposed to 12 hours of biological noise (male toadfish vocalizations, called boatwhistles, with a fundamental frequency of 180 Hz). SPLs for all noise conditions were maintained at approximately 150 dB re 1 µPa and fell within the oyster toadfish frequency sensitivity of 80-550 Hz. Exposures to biological signals, even for a duration of 12 hours, did not result in any hearing impairment. However, significant TTS of up to 8 and 20 dB was observed after exposures of 1 and 12 hours of vessel noise, respectively. More often, TTS has been studied in captive fishes exposed to elevated background noise and other non-impulsive, broadband[1] sources such as white noise (e.g., Breitzler et al., 2020; Scholik & Yan, 2002b; Smith et al., 2004b; Wysocki & Ladich, 2005).

Caged studies on hearing specialists show some hearing loss after several days or weeks of exposure to increased background sounds, although the hearing loss seems to recover (e.g., Breitzler et al., 2020; Scholik & Yan, 2002a; Smith et al., 2006; Smith et al., 2004a). Smith et al. (2006) and Smith et al. (2004a) exposed goldfish to noise with a SPL of 170 dB re 1 µPa and found a clear relationship between the amount of hearing loss and the duration of exposure until maximum hearing loss occurred 24 hours after exposure. A 10-minute exposure resulted in 5 dB of TTS, whereas a three-week exposure resulted in a 28 dB TTS that took over two weeks to return to pre-exposure levels (Smith et al., 2004a). Recovery times were not measured by investigators for shorter exposure durations.

Scholik and Yan (2001) demonstrated TTS in a hearing specialist, the fathead minnow (*Pimephales promelas*), after a 24-hour continuous exposure to white noise (0.3–2.0 kHz) at 142 dB re 1 µPa that took up to 14 days post-exposure to recover. This is the longest recorded time for a threshold shift to

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[1] A sound or signal that contains energy across multiple frequencies.

recover in a fish. The same authors also found that the bluegill sunfish (*Lepomis macrochirus*), a generalist, did not show significant elevations in auditory thresholds when exposed to the same stimulus (Scholik & Yan, 2002b). This evidence supports that fishes that are hearing specialists may be more sensitive to hearing loss when exposed to noise than fishes that are hearing generalists.

Breitzler et al. (2020) exposed zebrafish (a freshwater hearing specialist) to 24 hours of white noise at various frequencies and sound levels. TTS was observed at frequencies that were within the fish's best hearing sensitivity. Recovery took up to 14 days for fish exposed to the highest SPL (150 dB re 1 µPa). The highest threshold shifts recorded (up to 33 dB) also resulted in significant hair cell loss, whereas lower exposure levels did not. Like the other effects measured in this study, hair cell loss attributed to the highest exposure level returned to baseline levels within 7 days post-exposure. This demonstrates the ability for fish to regenerate hair cells and for hearing thresholds to recover to baseline levels (lacking evidence of PTS).

Wong et al. (2022) exposed zebrafish to 24 hours of white noise with four different temporal patterns (continuous fast and slow, regular and irregular intermittent). Impacts from white noise at SPLs of approximately 150 dB re 1 µPa included noise induced hearing loss, physical damage, and behavioral responses (discussed further in Section D.5.4.1). Auditory evoked potentials were used to measure significant threshold shifts (an average of approximately 13 dB across all tested frequencies) for all four temporal patterns. Although significant hair cell loss was not found, other indications of physical damage were reported including decreased Ribeye b protein and splaying of inner ear epithelial. Wong et al. (2022) proposed that the total acoustic energy of a given signal may play a larger role in observed effects than the temporal patterns of the signal.

Although TTS has been reported in larval zebrafish as early as five days post fertilization exposed to white noise at frequencies below 1.5 kHz with a SPL of 150 dB re 1 µPa, the actual duration of the exposure was not reported (Lara et al., 2022). Unlike the previous study, an analysis of the change in hair cell numbers, epithelia area, and general hair cell density showed varying responses to the sound source. Overall, there were no significant reductions in hair cell density between noise and control groups.

A direct comparison of results from these studies to fishes exposed to continuous sound sources in natural settings should be treated with caution due to differences between laboratory and open ocean or coastal environments. For example, fishes that are exposed to noise produced by a vessel passing by in their natural environment, even in areas with high levels of vessel movement, would only be exposed for short durations (e.g., seconds or minutes) and therefore relatively low SELs by transiting vessels. Fishes used in laboratory experiments are often held in a tank during exposures without any possibility to avoid the noise source and test species are often freshwater hearing specialists (e.g., goldfish or zebrafish) due to ease of availability from commercial sources. Furthermore, small aquariums present issues when transmitting acoustic signals as there may be excessive particle motion not accurately measured and accounted for during the experiment (e.g., Okumura et al., 2002). As evidence suggests that fish can recover from hearing loss (both threshold sensitivity and actual physical damage) even after long duration exposures in a confined space, it also indicates similar results to lower level and shorter duration exposures. Therefore, overall effects would not likely rise to the level of impact demonstrated in the summarized laboratory studies.

As noted in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), hearing specialists may be more susceptible to TTS from long duration continuous noise, such as broadband white noise, depending on the duration of the exposure (thresholds are proposed based on continuous exposure of 12 hours). However, it is less likely that TTS would occur in fishes that are hearing generalists.

D.5.2.3 Threshold Shift due to Impulsive Sound Sources

Popper et al. (2005) examined the effects of a seismic air gun array on a hearing specialist, the lake chub (*Couesius plumbeus*); and two hearing generalists, the northern pike (*Esox lucius*) and the broad whitefish (*Coregonus nasus*), a salmonid. In this study, fish were placed in pens in a shallow river (with water depths of 1.9 m) and exposed to either five or 20 shots from a nearby small air gun array (eight air guns total). Effects were noted at a cumulative SEL of 186 dB re 1 µPa^{2}s, based on an exposure of five shots with a mean single strike SEL of 177 dB re 1 µPa^{2}s (Popper et al., 2014). Like most air gun signals, each shot lasted a few milliseconds with the 5 shot exposure likely lasting a few minutes based on the 15 minutes it took to expose fish to 20 shots (pulse length and pulse interval was not reported). TTS was reported in the lake chub and northern pike, but not in the broad whitefish. Approximately 20 to 25 dB of TTS was reported at some, but not all tested frequencies for both species, and full recovery from threshold shifts took place within 18 hours after sound exposure. Examination of the sensory surfaces of the ears after allotted recovery times (one hour for five shot exposures, and up to 18 hours for 20 shot exposures) showed no damage to sensory hair cells in any of the fish from these exposures (Song et al., 2008).

A small percent (2-15% depending on the region and test group) of sensory hair cells in the inner ear was observed in caged fishes exposed to multiple passes of a towed air gun array at distances from five to several hundred meters (McCauley et al., 2003; McCauley & Kent, 2012). Pink snapper (*Chrysophrys auratus*), a hearing generalist, were exposed to multiple air gun shots for up to one and one-half hours (McCauley et al., 2003) where the maximum received SELs exceeded 180 dB re 1 µPa^{2}s. Though there were no long-term controls to compare to, the loss of sensory hair cells continued to increase for up to at least 58 days post exposure to 2.7 percent of the total cells. Gold band snapper (*Pristipomoides multidens*) and sea perch (*Lutjanus kasmira*), both hearing specialists, were also exposed to a towed air gun array simulating a passing seismic vessel (McCauley & Kent, 2012). Although received levels for these exposures have not been published, hair cell damage increased as the range of the exposure (i.e., distance to the source) decreased. Again, the amount of damage was considered small in each case (McCauley & Kent, 2012). It is not known if this hair cell loss would result in TTS since fish have tens or even hundreds of thousands of sensory hair cells in the inner ear and only a small portion were affected by the sound (Lombarte & Popper, 1994; Popper & Hoxter, 1984). A reason McCauley and Kent (2012) found damage to sensory hair cells, while Popper et al. (2005) did not, may be in their distinct methodologies. Their studies had many differences, including species and the precise sound source characteristics.

Hastings et al. (2008) exposed a hearing specialist, the pinecone soldierfish (*Myripristis murdjan*), and three hearing generalists, the blue green damselfish (*Chromis viridis*), the saber squirrelfish (*Sargocentron spiniferum*), and the bluestripe seaperch (*Lutjanus kasmira*) to a nearby active seismic survey. Fish were located at one of three test sites that varied in distance from the actual survey (approximately 45 m to several kilometers). Fish in cages were exposed to multiple air gun shots with a cumulative SEL of 190 dB re 1 µPa^{2}s. The authors found no TTS in any fish examined up to 12 hours after the exposures.

In an investigation of another impulsive source, Casper et al. (2013b) found that some fishes may actually be more susceptible to barotrauma (e.g., swim bladder ruptures, herniations, and hematomas) than effects to the auditory system when exposed to simulated impact pile driving. Hybrid striped bass (white bass *Morone chrysops* x striped bass *M. saxatilis*) and Mozambique tilapia (*Oreochromis mossambicus*), both hearing generalists, were exposed to SELs between 213 and 216 dB re 1 µPa^{2}s. The subjects exhibited barotrauma, and although researchers began to observe signs of inner ear hair cell loss, these effects were small compared to the other non-auditory injuries that occurred. Smith et al. (2022) observed physical damage in the inner ear of a hearing generalist, Pacific mackerel (*Scomber japonicus*), exposed to underwater explosions starting at received peak to peak SPLs of 220 dB re 1 µPa. Though there are no direct measurements of TTS in fishes exposed to explosive sources, it is assumed that fish would demonstrate similar effects on auditory detection as those exposed to other impulsive sources such as those described above. These received sound levels likely represent thresholds at which hearing effects may occur.

PTS has not been known to occur in fishes tested to date. Any hearing loss in fish may be as temporary as the timeframe required to repair or replace the sensory cells that were damaged or destroyed (Popper et al., 2014; Popper et al., 2005; Smith et al., 2006). The lowest SEL at which TTS has been observed in fishes with hearing specializations exposed to air gun signals is 186 dB re 1 µPa^{2}s. As reviewed in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), hearing generalists would be less susceptible to TTS than specialists, even at higher levels and longer durations. Fishes that are hearing specialists may be susceptible to TTS within very close ranges to an explosive.

D.5.3 MASKING

Masking is likely to occur in most fishes due to varying levels of ambient or natural noise in the environment, such as wave action, precipitation, or other animal vocalizations (Popper et al., 2014). Ambient noise during higher sea states in the ocean has resulted in masking in several fish species (Chapman & Hawkins, 1973; Ramcharitar & Popper, 2004). Although the overall intensity or loudness of ambient or human-generated noise may result in masking effects in fishes, masking is most problematic when human-generated signals or ambient noise levels overlap the frequencies of biologically important signals (Buerkle, 1968, 1969; Popper et al., 2014; Tavolga, 1974).

Auditory sensitivity can be hindered by masking noise. Wysocki and Ladich (2005) investigated the influence of continuous white noise on the auditory thresholds of two hearing specialists, the goldfish and the lined Raphael catfish (*Platydoras costatus*) as well as a hearing generalist, the pumpkinseed sunfish (*Lepomis gibbosus*). Experiments were conducted in aquariums. Continuous white noise with an SPL of approximately 130 dB re 1 µPa at 1 m resulted in 23–44 dB of masking within the goldfish and catfish region of best sensitivity between 500 and 1,000 Hz. The sunfish experienced only 11 dB of masking during the same noise treatment. In a similar study, meagre (*Argyrosomus regius*) exposed to boat noise at relative SPLs of 130 dB re 1 µPa showed a masking effect of up to 20 dB during presentation of the noise stimulus (Vieira et al., 2021). As seen in previous studies, fish calls were masked by up to 20 dB. Masked auditory thresholds were also measured in the croaking gourami (*Trichopsis vittata*, Osphronemidae) during playbacks of white noise at a relative SPL of 110 dB re 1 µPa (Maiditsch & Ladich, 2022). The experiment revealed a significant increase in auditory thresholds during noise presentations. Specifically, the largest effect was noted at frequencies that overlap with female pre-spawning purring vocalizations.

Masking could lead to potential fitness costs depending on the severity of the reaction and the animal's ability to adapt or compensate during an exposure (e.g., de Jong et al., 2020; Krahforst et al., 2016; Radford et al., 2014; Slabbekoorn et al., 2010). For example, masking could result in changes in predator-prey relationships, potentially inhibiting a fish's ability to detect predators and therefore increase its risk of predation, or limiting a fish's ability to classify and locate prey items, reducing foraging success (e.g., Astrup, 1999; Mann et al., 1998; McCormick et al., 2018; Simpson et al., 2015; Simpson et al., 2016; Voellmy et al., 2014). Alternatively, if the masking noise overlaps the hearing range of fish predators (e.g., marine mammals) or their prey, this could be beneficial if the masking signal prevents predators from finding them or increases their chance of capturing prey items.

Masking may also limit the distance over which fish can communicate or detect important signals, including conspecific vocalizations such as those made during reproductive phases or sounds emitted from a reef for navigating larvae (Alves et al., 2016; Codarin et al., 2009; de Jong et al., 2020; Higgs, 2005; Krahforst et al., 2016; Neenan et al., 2016; Ramcharitar et al., 2006; Ramcharitar et al., 2001; Stanley et al., 2017; Vieira et al., 2021). If the masking signal is brief (a few seconds or less), biologically important signals may still be detected, resulting in little effect to the individual. If the signal is longer in duration (minutes or hours) or overlaps with important frequencies for a particular species, more severe consequences may occur such as the inability to attract a mate and reproduce. The Lombard effect has been reported in fishes (both in a laboratory setting and in situ) in an increasing number of experiments (e.g., Holt & Johnston, 2014; Luczkovich et al., 2016b; Somogyi & Rountree, 2023). The Lombard effect is defined as a potentially compensatory behavior where an animal increases the source level of its vocalizations in response to elevated noise levels. The Lombard effect is currently understood to be a reflex that may be unnoticeable to the animal, or it could lead to increased energy expenditure during communication.

Passive acoustic monitoring was conducted during several phases of an offshore windfarm installation project (Siddagangaiah et al., 2021). Installation and active use of the windfarm resulted in increased background noise levels as well as changes in fish chorusing patterns compared to baseline conditions in the Study Area. For example, type 1 choruses occurred for longer durations and at a lower intensity compared to pre-construction monitoring. Type 2 choruses showed an increase in intensity but no change in overall call duration during the same portion of the project installation. After construction was complete, residual effects on call duration and intensity were evident for Type 1 chorusing (increased call duration and intensity) though Type 2 chorusing did not seem affected and returned to baseline levels. Changes in fish vocal behavior may be affected to masking (the Lombard effect) or other factors such as disrupted group cohesion during periods of noise presentation. Although the construction noise included impact pile driving, it is difficult to distinguish whether these impacts were a result of the impulsive signals alone, or if noise from other parts of the activity (vessel movements, dredging, windmill operations) contributed changes in fish chorusing behavior. Additional research has shown that some, but not all species, respond to sound exposures with the Lombard effect (e.g., Brown et al., 2021; Maiditsch & Ladich, 2022).

The *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) highlights the lack of data for masking by sonar but suggests that the narrow bandwidth and intermittent nature of most sonar signals would result in only a limited probability of any masking effects. In addition, most sonars (mid-, high-, and very high-frequency) are above the hearing range of most marine fish species (see Figure D.5-1), eliminating the possibility of masking for these species. In most cases, the probability of masking would further decrease with increasing distance from the sound source.

In addition, few data are available for masking by impulsive signals (e.g., impact pile driving and air guns) (Popper et al., 2014; Siddagangaiah et al., 2021). Impulsive sounds are typically brief, lasting only fractions of a second, where masking could occur only during that brief duration of sound. Biological sounds can typically be detected between pulses within close distances to the source unless those biological sounds are like the masking noise, such as impulsive or drumming vocalizations made by some fishes (e.g., cod or haddock). Masking could also indirectly occur because of repetitive impulsive signals where the repetitive sounds and reverberations over distance may create a more continuous noise exposure. Currently there are no direct observations of masking in fishes due to explosives. The *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) highlights a lack of data that exist for masking by explosives but suggests that the intermittent nature of explosions would result in very limited probability of any masking effects, and if masking were to occur it would only occur during the duration of the sound. Potential masking from explosives is likely to be like masking studied for other impulsive sounds such as air guns.

Although there is evidence of masking because of exposure to vessel noise, the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) does not present numeric thresholds for this effect. Instead, relative risk factors are considered, and it is assumed the probability of masking occurring is higher at near to moderate distances from the source (up to hundreds of meters) but decreases with increasing distance (Popper et al., 2014).

D.5.4 BEHAVIORAL REACTIONS

Behavioral reactions of fishes have been observed across many types of sound sources. Most research has been performed using air guns (including large-scale seismic surveys), sonar, and vessel noise. Fewer observations have been made on behavioral reactions to impact pile driving noise and there are no data available on reactions to explosives, although fish are likely to show similar behavioral reactions to any impulsive noise.

Fish studies have identified the following behavioral reactions to sound: alteration of natural behaviors (e.g., startle or alarm), and avoidance (LGL Ltd Environmental Research Associates et al., 2008; McCauley et al., 2000b; Pearson et al., 1992). In the context of this Supplemental EIS/OEIS, and to remain consistent with available behavioral reaction literature, the terms “startle,” “alarm,” “response,” and “reaction” will be used synonymously. In addition, observed behavioral reactions to sound can include disruption to or alteration of swimming, schooling, feeding, breeding, and migrating. Sudden changes in sound level can cause fish changes in depth and swimming direction. However, some fish either do not respond, or learn to tolerate or habituate to the noise exposure (e.g., Bruintjes et al., 2016; Currie et al., 2020; Hubert et al., 2020b; Nedelec et al., 2016b; Radford et al., 2016).

Research on behavioral reactions can be difficult to understand and interpret. For example, behavioral responses often vary depending on the type of exposure and sound source. Changes in sound intensity may be more important to a fish’s behavior than the maximum sound level. Some studies show that sounds that fluctuate in sound pressure level or have intermittent pulse rates tend to elicit stronger responses from fish than even stronger sounds with a continuous level (Currie et al., 2020; Neo et al., 2014; Schwarz & Greer, 1984). It has also been suggested that unpredictable sounds that last for long durations may have the largest impact on behavioral responses (de Jong et al., 2020). Interpreting behavioral responses can also be difficult due to species-specific behavioral tendencies, motivational state (e.g., feeding or mating), an individual’s previous experience, how resilient a species is to changes in their environment, and whether the fish are able to avoid the sound source (e.g., caged versus free-swimming subjects). Results from caged studies may not provide a representative understanding of how free-swimming fishes may react to the same or similar sound exposures (Hawkins et al., 2015), especially when the experimental population consists of those species bred and raised in captivity (e.g., generations of captive zebrafish used in biological studies).

D.5.4.1 Behavioral Reactions to Sonar and Other Transducers

Behavioral reactions to sonar have been studied both in caged and free-swimming fish, although results can often-times be difficult to interpret depending on the species tested and the study environment. Jorgensen et al. (2005) showed that caged cod and spotted wolffish (*Anarhichas minor*) lacked any response to simulated sonar between 1 and 8 kHz. However, within the same study, reactions were seen in juvenile herring. It is likely that the sonar signals were inaudible to the cod and wolf fish (species that lack notable hearing specializations) but audible to herring (a hearing specialist).

Several experiments studied the reactions of both wild and captive Atlantic herring (*Clupea harengus*) to the Royal Netherlands Navy's experimental mid-frequency active sonar ranging from 1 to 7 kHz with maximum cumulative SELs of 184 dB re 1 μPa^2s (Doksaeter et al., 2009; Doksaeter et al., 2012; Sivle et al., 2015a; Sivle et al., 2012a). No avoidance or escape reactions were observed when herring were exposed to sonar sources and the authors concluded that the use of naval sonar poses little risk to populations of herring. Instead, significant reactions were noted at lower received sound levels of different non-sonar sound types. For example, dive responses (i.e., escape reactions) were observed when herring were exposed to killer whale feeding sounds at received SPLs of approximately 150 dB re 1 µPa (Sivle et al., 2012a). Startle responses were seen when the cages for captive herring were hit with a wooden stick and with the ignition of an outboard boat engine at a distance of one meter from the test pen (Doksaeter et al., 2012). It is possible that the herring were not disturbed by the sonar because they were more motivated to continue other behaviors such as feeding, or did not associate the sound as a threatening stimulus as they likely did for the killer whale and outboard motorboat signals.

Short et al. (2020) studied the effect of a broadband, pulsed, acoustically random noise exposure (60–2,000 Hz) on the swimming behavior of a captive freshwater shoaling species (Eurasian minnows, *Phoxinus phoxinus,* hearing specialists). In response to the noise exposure, group responses were more consistent in their escape behavior (e.g., startled, consistent speed, less erratic path, stronger group cohesion, more synchronized orientation) compared to fish tested individually. Although the pulsed tones were broadband, unlike most sonar sources that have a limited center frequency, the study provides insight into the differences in group versus individual reactions particularly for shoaling species.

There is evidence that elasmobranchs (cartilaginous fish including sharks and rays) also respond to human-generated sounds. A number of researchers conducted experiments in which they played back sounds (e.g., pulsed tones below 1 kHz) and attracted a number of different shark species to the sound source (e.g., Casper et al., 2012a; Myrberg et al., 1976; Myrberg et al., 1969; Myrberg et al., 1972; Nelson & Johnson, 1972). The results of these studies showed that sharks, hearing generalists, were attracted to irregularly pulsed low-frequency sounds (below several hundred Hz), in the same frequency range of sounds that might be produced by struggling prey. However, abrupt and irregularly pulsed human-generated noise (0.02–10 kHz, with most energy below 1 kHz) resulted in withdrawal responses of certain shark species (Chapuis et al., 2019). Sharks are not known to be attracted to continuous signals or higher frequencies that they presumably cannot hear (Casper & Mann, 2006; Casper & Mann, 2009).

Only a few species of marine fishes can detect sonars above about 2 kHz, meaning that most fishes would not detect most mid-, high-, or very high-frequency Navy sonars. The few marine species that can detect above 2 kHz and have some hearing specializations may be able to better detect the sound and would therefore be more likely to react. However, researchers have found little reaction by adult fish in the wild to sonars within the animals' auditory detection range (Doksaeter et al., 2009; Doksaeter et al., 2012; Sivle et al., 2012a). The *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) suggests that fish able to hear sonars would have a low probability of reacting to the source within near or intermediate distances (within tens to hundreds of meters) and a decreasing probability of reacting at increasing distances.

D.5.4.2 Behavioral Reactions to Vessel Noise

Vessel traffic contributes to the amount of noise in the ocean and has the potential to affect fishes. Several studies have reported and reviewed avoidance responses by fishes (e.g., herring and cod) to vessels or playbacks of vessel noise (De Robertis & Handegard, 2013; Engås et al., 1995; Handegard et al., 2003; Waddell & Sirovic, 2023). For example, Misund (1997) found fish showed avoidance reactions at ranges of 50 to 150 m ahead of the ship. When the vessel passed over them, some species of fish responded with sudden escape reactions that included lateral avoidance or downward compression of the school. In some rare cases, there have also been reports of fish attraction to traditional and unmanned underwater vessels (Fernandes et al., 2000; Rostad et al., 2006). Though the mechanism for this response is still unknown it is likely related to the type of fish (i.e., predators) and the way they interpret their environment. It is important to note that vessel noise alone may not be the only mechanism for some of these observed responses (De Robertis & Handegard, 2013). Rather, it is likely that other cues (e.g., visual cues, water displacement) play a large role in observed responses of fishes to passing vessels.

As mentioned above, behavioral reactions are variable depending on a number of factors such as (but not limited to) the type of fish, its life history stage, behavior, time of day, location, the sound source (e.g., type of vessel or motor vs. playback of broadband sounds), and the sound propagation characteristics of the water column (Popper et al., 2014; Schwarz & Greer, 1984). Reactions to playbacks of continuous noise or passing vessels generally include basic startle and avoidance responses. Other widely observed responses include: changes in vocalizations; modifications in movement patterns such as changes in vertical distribution in the water column, swim speeds, distance traveled or changes to group cohesion; modified attention or evidence of distractions; effects on foraging success and antipredator responses (e.g., Bracciali et al., 2012; Gendron et al., 2020; Handegard et al., 2015; Jimenez et al., 2020; Krahforst et al., 2016; Luczkovich et al., 2016a; Luczkovich et al., 2016b; Magnhagen et al., 2017; Mauro et al., 2020; Mills et al., 2020; Nedelec et al., 2017a; Neo et al., 2015; Roberts et al., 2016a; Simpson et al., 2015; Stasso et al., 2022; Vieira et al., 2021; Voellmy et al., 2014). Both playbacks and actual noise conditions from nearby boats have also resulted in alterations in reproductive and nesting behaviors; signaling and aggression towards potential mates, competitors, and conspecifics; diminished territorial interactions; and reduced parental care behaviors (Amorim et al., 2022; Butler & Maruska, 2020; McCloskey et al., 2020).

Behavioral responses may be dependent on the type of vessel to which a fish is exposed. For example, juvenile damselfish (*Pomacentrus wardi*) exposed to sound from a two-stroke engine resulted in startle responses, reduction in boldness (increased time spent hiding, less time exhibiting exploratory behaviors) and space use (maximum distance ventured from shelter or traveled within the test enclosure). However, damselfish exposed to sound from a four-stroke engine generally displayed similar responses as control fish exposed to ambient noise (e.g., little or no change in boldness) (McCormick et al., 2019). Although the two sound sources were very similar, the vessels powered by the four-stroke engine were of lower intensity (i.e., less energy across all frequencies) compared to vessels powered by the two-stroke engine, which may explain the overall reduced response to this engine type.

Vessel noise may also lead to changes in anti-predator response, but these responses vary by species. During exposures to vessel noise, juvenile Ambon damselfish (*Pomacentrus amboinensis*) and European eels showed slower reaction times and lacked startle responses to predatory attacks. Subsequently these fish showed signs of distraction and increased their risk of predation during both simulated and actual predation experiments (Simpson et al., 2015; Simpson et al., 2016). However, it is not known if these responses would decrease over time as repeated measures were not performed. Juvenile Ambon damselfish showed a reduction in learned anti-predator behaviors likely because of distraction (Ferrari et al., 2018). Spiny chromis (*Acanthochromis polyacanthus*) exposed to chronic (12 consecutive days) boat noise playbacks spent less time feeding and interacting with offspring and displayed increased defensive acts. In addition, offspring survival rates were lower at nests exposed to chronic boat noise playbacks versus those exposed to ambient playbacks (Nedelec et al., 2017b). This suggests that chronic or long-term exposures could have more severe consequences.

In contrast to results from the previous study, larval Atlantic cod showed a stronger anti-predator response and was more difficult to capture during simulated predator attacks (Nedelec et al., 2015). There are also observations of a general lack of response to shipping noise (e.g., Higgs & Humphrey, 2019; Roberts et al., 2016b). Mensinger et al. (2018) found that Australian snapper located in a protected area showed no change in feeding behavior or avoidance during boat passes. Conversely, snapper in areas where fishing occurs startled and ceased feeding behaviors during boat presence suggesting that location and experience have a strong influence on whether fishes react.

Although behavioral responses such as those listed above were often noted during the onset of most sound presentations, most behaviors did not last long, and animals quickly returned to baseline behavior patterns. In fact, in one study with zebrafish, when given the chance to move from a noisy tank (with SPLs reaching 120–140 dB re 1 µPa) to a quieter tank (110 dB re 1 µPa SPL), there was no evidence of avoidance. The fish did not seem to prefer the quieter environment and continued to swim between the two tanks comparable to control sessions (Neo et al., 2015). However, many of these reactions are difficult to extrapolate to real-world conditions due to the captive environment in which testing occurred.

To investigate potential avoidance on a larger scale, Ivanova et al. (2020) tagged Arctic cod and recorded movement and behavior during exposure to noise produced by cargo and cruise ship traffic. Overall, cod increased their horizontal movement outside of their estimated home range when vessels were either present or moving, compared to periods where vessels were absent, indicating periods of potential avoidance. Changes in feeding, travel, and search behaviors were also observed when comparing each sound treatment. The authors note that future studies should continue to investigate whether these observed effects are prolonged or how quickly fish may return to their home range and baseline behaviors.

Most fish species should be able to detect vessel noise due to its low-frequency content and their hearing capabilities. The *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014) suggests that fishes have a high to moderate probability of reacting to nearby vessel noise (i.e., within tens of meters) with decreasing probability of reactions as distance from the source increases (hundreds or more meters).

D.5.4.3 Behavioral Reactions to Impulsive Noise

It is assumed that most species would show similar behavioral responses across all impulsive sounds, regardless of the source (e.g., weapons noise and explosions). Data on fish behavioral reactions exposed to impulsive sound sources is mostly limited to studies using caged fishes and seismic air guns and impact pile driving. General reactions include startle or alarm responses and increased swim speeds at the onset of impulsive sounds (Fewtrell & McCauley, 2012; Løkkeborg et al., 2012; Pearson et al., 1992; Roberts et al., 2016a; Spiga et al., 2017).

Several species of caged rockfish (*Sebastes* species), white trevally (*Pseudocaranx dentex*) and pink snapper (all hearing generalists) exhibited startle or alarm reactions to seismic air gun pulses between 180 dB re 1 µPa and 205 dB re 1 µPa peak-to-peak SPL (Pearson et al., 1992). More subtle behavioral changes were noted at lower SPLs, including changes in swim speeds. At the presentation of the sound, all three species moved to the bottom of the experimental enclosure. Both white trevally and pink snapper also exhibited changes in schooling behaviors including changes in group cohesion when exposed to air gun noise (Fewtrell & McCauley, 2012). These behavioral responses were seen during SELs as low as 147 and up to 161 dB re 1 µPa^{2}s but habituation occurred in all cases, either within a few minutes or within 30 minutes after the final air gun shot (Fewtrell & McCauley, 2012; Pearson et al., 1992).

A study by a research group in the Netherlands conducted an in situ experiment and exposed tagged Atlantic cod to a simulated seismic survey event (Hubert et al., 2020a). Thirty six air guns were utilized in the array and the seismic event was conducted continuously over three-and-a-half days. The location was selected due to high site fidelity of cod in the areas immediately surrounding windfarm turbines in the North Sea and allowed the research group to monitor general movements patterns and overall behavior before, during, and after the survey. Cod were more likely to be inactive during sound exposures and immediately following the surveys, compared to baseline movement patterns (van der Knaap et al., 2021).

Some studies have shown a lack of behavioral reactions to air gun noise. The same research group in the Netherlands exposed cod to playbacks of an air gun in a large net pen (Hubert et al., 2020a). Unlike the study conducted in the North Sea, cod exposed in a net pen showed very little change in behavior or overall use of space within the pen. Herring exposed to an approaching air gun survey (from 27 to 2 km over 6 hours), resulting in single pulses of 125 to 155 dB re 1 µPa^{2}s SEL, did not react (Pena et al., 2013). Although these levels are similar to those tested in other studies that exhibited responses (Fewtrell & McCauley, 2012), the distance of the exposure to the test enclosure, the slow onset of the sound source, and potential motivation for feeding may have affected the observed response (Pena et al., 2013).

Wardle et al. (2001) observed marine fish on an inshore reef before, during, and after air gun surveys at varying distances. The air guns were calibrated at a peak level of 210 dB re 1 µPa at 16 m and 195 dB re 1 µPa at 109 m from the source. Other than observed startle responses and small changes in the position of adult pollack (*Pollachius pollachius* [a hearing generalist]), when the air gun was located within 10 m of the test site, they found no substantial or permanent changes in the behavior of the fish on the reef (including juvenile saithe [*Pollachius virens*] and cod) throughout the course of the study. A similar study monitored species abundance, composition, behavior and movement patterns over the course of several months to capture long-term responses to a five-day seismic survey (Meekan et al., 2021). This study utilized multiple methods such as underwater baited cameras, tagging, and passive acoustic monitoring to understand each variable under investigation. Overall, the results suggested that there was little, if any, short- or long-term impacts on the demersal fishes (i.e., those that hover slightly above the bottom) from exposure to the full-scale survey.

McQueen et al. (2022) tagged Atlantic cod (*Gadus morhua L.*) to analyze potential responses to a nearby seismic survey. Tagging and analysis was conducted over multiple years (2019-2021) in known spawning locations. Hydrophones and acoustic receivers were placed in two locations; the test site located close to the 'racetrack' where the seismic survey event occurred, and a control site in a nearby area but separated from the racetrack by islands and other features to prevent any sound exposure at this portion of the Study Area. Exposures consisted of a three-hour treatment period with active seismic signals present, and a three-hour control period where no seismic activity was detectable. These periods were repeated in random order over the course of a week in a given test year. SELs varied from 120 to 145 dB re 1 µPa^{2}s at the closest point at the treatment site to the seismic survey. Overall, spawning cod did not avoid the noise from the seismic survey and remained at the spawning site despite elevated sound levels. It is likely the cod's preference for the spawning location motivated them to remain in the area despite the presence of the noise.

In contrast, other research on the effects of impulsive seismic survey sound that can last weeks to months has indicated that this level of behavioral response is unlikely (McQueen et al. 2022; Meeken et al. 2021). For example, Meekan et al. (2021) observed no short-term (days) or long-term (months) effects of exposure to the composition, abundance, size structure, behavior, or movement to assemblages of tropical demersal fishes, including hearing specialist species (e.g., Lutjanidae sp.), in Western Australia exposed to noise from a commercial-scale seismic air gun survey with received SELs of up to approximately 180 dB re 1 µPa^2·s. McQueen et al. (2022) examined the responses of spawning cod in the North Sea exposed to seismic air gun noise over two 1-week periods, with fluctuating SELs of up to 145 dB re 1 µPa^2·s, comparable to a full-scale industrial survey 5 to 40 km away (Handegard et al. 2003). Tagged cod in this study were not displaced from spawning grounds (McQueen et al. 2022). McQueen et al. (2022) speculated that strong affinity to selected spawning sites overcame the behavioral effects of stressor exposure. Although the sound source (i.e., seismic air guns) is not analogous to pile driving, they both produce high-intensity, impulsive sound primarily in the 100-Hz or lower frequency bands that overlap the spectral range of cod communication and hearing sensitivity and are informative in the absence of studies assessing the impacts of pile driving to Atlantic cod. Overall, these findings suggest that, although noise exposure during sensitive life stages is a potential concern, disturbances resulting from impulsive sound sources, such as pile driving or seismic air guns, may not necessarily result in adverse effects, such as the complete abandonment of an area for the duration of a spawning season versus temporary displacement or disturbance of Atlantic cod or other hearing specialist species.

Unlike the previously described studies, Slotte et al. (2004) used fishing sonar (38 kHz echo sounder) to monitor behavior and depth of blue whiting (*Micromesistius poutassou*) and Norwegian spring herring (hearing generalists) spawning schools during air gun exposures. They reported that fishes near the air guns appeared to move to greater depths after the air gun exposure compared to their vertical position prior to the air gun usage. Moreover, the abundance of animals 30–50 km away from the air guns increased during seismic activity, suggesting that migrating fish left the zone of seismic activity and did not re-enter the area until the activity ceased. It is unlikely that either species was able to detect the fishing sonar. However, it should be noted that these behavior patterns may have also been influenced by other variables such as motivation for feeding, migration, or other environmental factors (e.g., temperature, salinity).

Alterations in natural behavior patterns due to exposure to pile driving noise reported noted thus far are like those seen in response to seismic surveys. These changes in behavior include startle responses, changes in depth (in both caged and free-swimming subjects), swim speeds, group cohesion, and in attention and anti-predator behaviors, breaching, and directional avoidance (e.g., Hawkins et al., 2014; Kok et al., 2021; Mueller-Blenkle et al., 2010; Neo et al., 2015; Roberts et al., 2016a; Spiga et al., 2017). The severity of responses varies greatly by species and received SPL. For example, Neo et al. (2014) observed slower recovery times in fishes exposed to intermittent sounds (similar to pile driving) compared to continuous exposures. However, at some higher SPLs (152 - 157 dB re 1 µPa) some free-swimming fishes avoided pile driving noise (Iafrate et al., 2016). Using a baited remote underwater video Roberts et al. (2016a) showed that although multiple species of free swimming fish responded to simulated pile driving recordings, not all responded consistently. In some cases, only one fish would respond while the others continued feeding. In other instances, various individual fish would respond to different strikes. Similar results were reported at an existing windfarm in the Belgian part of the North Sea where tagged free-range Atlantic cod (*Gadus morhua*) showed no significant avoidance response to a largescale pile driving effort and a high variance in measured behavioral responses (van der Knaap et al., 2022). As part of the same experiment, echosounders also indicated that fish abundance and group cohesion changed when pelagic fishes were exposed to pile driving and seismic activities. However, the location of schooling fishes in the water column differed by sound source type, and some of these effects were also noted at the control site (i.e., no sound exposure) which may be explained by other abiotic factors such as seasonality (Kok et al., 2021). The repetition rate of pulses during an exposure may also influence what behaviors are observed during many of these experiments and how quickly these behaviors recovered as opposed to the overall sound pressure or exposure level (Neo et al., 2014).

As summarized in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), species may react differently to the same sound source depending on a number of variables, such as the animal's life stage or behavioral state (e.g., feeding, mating). Without specific data, it is assumed that fishes with similar hearing capabilities react similarly to all impulsive sounds outside the zone for hearing loss and injury. Observations of fish reactions to large-scale air gun surveys are informative, but not necessarily directly applicable to analyzing impacts from the short-term, intermittent use of all impulsive sources. It is assumed that fish have a high probability of reacting to an impulsive sound source within near and intermediate distances (tens to hundreds of meters), and a decreasing probability of reaction at increasing distances.

D.5.5 PHYSIOLOGICAL RESPONSE

Fishes naturally experience stress within their environment and as part of their life histories. To simulate physiological stress, it is common to test subjects using a variety of stimuli, such as predator vocalizations and anthropogenic noise exposures. The stress response in an animal is a suite of physiological changes that are meant to help an animal mitigate the impact of a stressor. However, if the magnitude and duration of the stress response is too great or too long, then it can have negative consequences to the animal (e.g., decreased immune function, decreased reproduction, increased likelihood of predation). The initial response to a stimulus is a rapid release of stress hormones into the circulatory system, which may cause other responses such as elevated heart rate and blood chemistry changes. A sudden increase in SPL (i.e., presentation of a sound source or acute/short-term exposure), increase in overall background noise levels, or long-duration or continuous exposures have been shown to cause stress, including measurements of biochemical responses and alteration of metabolic rates indicative of a stress response such as increased ventilation and oxygen consumption (e.g., Goetz et al., 2015; Guh et al., 2021; Lara & Vasconcelos, 2021; Madaro et al., 2015; Pickering, 1981; Popper & Hastings, 2009a; Radford et al., 2016; Remage-Healey et al., 2006; Simpson et al., 2015; Simpson et al., 2016; Smith et al., 2004a; Smith et al., 2004b; Spiga et al., 2017; Wysocki et al., 2007; Wysocki et al., 2006). However, results from these studies have varied in part this is due the variety of stimuli used in a study as well as the complicated physiological responses each individual undergoes.

A common response that has been observed in fishes involves the production of cortisol (a stress hormone) when exposed to sounds such as boat noise, tones, or predator vocalizations. For example, Nichols et al. (2015) exposed the giant kelpfish (*Heterostichus rostratus*), a hearing generalist, to intermittent boat noise and found increased cortisol levels with increased sound levels. Cod exposed to a short-duration upsweep (a tone that sweeps upward) across 100–1,000 Hz showed increases in cortisol levels, which returned to normal one hour post-exposure (Sierra-Flores et al., 2015). Remage-Healey et al. (2006) found elevated cortisol levels in Gulf toadfish (*Opsanus beta* [a hearing generalist]), when exposed to low-frequency bottlenose dolphin sounds, but observed no cortisol change when exposed to low-frequency “pops” produced by snapping shrimp. Butler and Maruska (2020) exposed mouth-brooding freshwater female African cichlids (hearing specialist) to noise within their hearing range (0.1–2.0 kHz) for three hours and then measured the effects of sound on several factors, including cortisol levels. Like other findings, cortisol levels were higher immediately after exposure, in fish exposed to noise.

While studies have explored the impacts of vessel noise on hormones, results varied in that some fish species demonstrated increases in cortisol levels (Remage-Healey et al., 2006) and others showed no evidence of change (Mills et al., 2020). One study did show a change in androgen hormone levels in both male and female fish (Mills et al., 2020), however, the impacts of this change are unknown.

Smith et al. (2004b) found no increase in corticosteroid (a class of stress hormones) in goldfish exposed to a continuous, band-limited noise (0.1–10 kHz) at 170 dB re 1 µPa SPL for one month. Wysocki et al. (2007) exposed rainbow trout to continuous band-limited noise with an SPL of about 150 dB re 1 µPa for nine months with no observed stress effects (i.e., growth rates and secondary stress measures via blood samples). Growth rates and effects on the trout’s immune systems were not significantly different from control animals exposed to 110 dB re 1 µPa SPL. In addition, although there was a difference of 10 dB in overall background level and boat activity between test sites, reef fish (*Halichoeres bivittatus*) showed similar levels of whole-body cortisol (Staaterman et al., 2020). This suggests that boat noise, in this context, was not as stressful as handling of the fish for this experiment and contradicts previous conclusions that follow similar study designs.

Kusku (2020) measured respiratory changes as secondary indicators of stress in Nile tilapia (*Oreochromis niloticus*) to determine potential effects of long-term exposure to underwater sound playback, including shipping noise. Fish exposed to noise showed as much as a two-fold increase in respiratory indicators (opercular beat rate and pectoral wing rate) after 10 minutes of sound exposure as compared to controls and pre-exposure rates. Over the next 120 days of continuous sound exposure, respiratory indicators declined steadily and returned to baseline. The authors conclude that the data support habituation of fish to chronic noise exposure.

Zhang et al. (2022) studied the effects of simulated ship noise on liver metabolite production and gene expression of hybrid sturgeon (*Acipenser baerii* × *A. schrencki*). During continuous exposure to underwater noise (12 hours), cell motility increased, while protein synthesis (the process of creating protein in the body) and several metabolic pathways were inhibited. Results suggested that immune response was initiated when exposed to underwater noise stress and that immune-related pathways were activated to protect the normal activities of the fish, despite evidence that underwater noise may have caused some inflammatory responses.

Factors such as early-stage development or survival rates as indicators of stress from a given noise exposure have also been investigated. For example, reef fish embryos exposed to boat noise have demonstrated changes in morphological development and increases in heart rate, another indication of a physiological stress response, although survival rates were unchanged (Fakan & McCormick, 2019; Jain-Schlaepfer et al., 2018). Faria et al. (2022) found evidence of detrimental effects of chronic boat noise on wild Lusitanian toadfish (*Halobatrachus didactylus*) development, and of increased physiological stress assessed by oxidative stress and energy metabolism biomarkers. It has been shown that chronic or long-term (days or weeks) exposures of continuous man-made sounds can also lead to a reduction in embryo viability, decreased growth rates, and early mortality including in larvae and fishes infected with parasites (Lara & Vasconcelos, 2021; Masud et al., 2020; Nedelec et al., 2015; Sierra-Flores et al., 2015). Furthermore, Masud et al. (2020) found that guppies exposed to 24 hours of broadband white noise showed increased disease susceptibility compared to those exposed for longer durations (up to 7 days).

Contrary to previous findings, meagre larvae and embryos showed little change in development after exposure to playbacks of boat noise. Specifically, eggs were either provided with either a silent treatment (the controls) or exposed to playbacks of boat noise. On average, playback levels were 25 dB higher than control conditions. Overall, boat noise did not affect measured stress or development responses such as hatching rate, larval size, and yolk sack area. Effects that were noted, such as the size of the lipid droplet area, were small and should be verified with additional data (Trabulo et al., 2023).

Research on physiological stress in fishes due to exposure to explosive sources is limited. Sverdrup et al. (1994) studied levels of stress hormones in Atlantic salmon after exposure to multiple detonations in a laboratory setting. Increases in cortisol and adrenaline were observed following the exposure, with adrenaline values returning to within normal range within 24 hours.

In summary, fishes may have physiological stress reactions to sounds that they can hear. Generally, stress responses are more likely to occur in the presence of potentially threatening sound sources, such as predator vocalizations, or the sudden onset of impulsive signals rather than from non-impulsive or continuous sources such as vessel noise or sonar. If an exposure is short, the stress responses are typically brief (a few seconds to minutes). In addition, research shows that fishes may habituate to (i.e., learn to tolerate) the noise that is being presented after multiple exposures or longer duration exposures that prove to be non-threatening. However, exposure to chronic noise sources can lead to more severe impacts over time, such as reduced growth rates which can lead to reduced survivability for an individual. It is assumed that any physiological response (e.g., hearing loss or injury) or significant behavioral response is also associated with a stress response.

D.5.6 DIRECT INJURY

Injury to fishes refers to the direct effects on the tissues or organs of a fish. Auditory injuries are generally discussed above in Section D.5.2. No research on the potential injuries from moderate- to low-level noise from vessels, aircraft, and weapons firing is available. However, these sound sources lack the amplitude and energy to cause any direct injury and are not discussed further.

D.5.6.1 Injury due to Sonar and Other Transducers

Non-impulsive sound sources (e.g., sonar, acoustic modems, and sonobuoys) have not been known to cause direct injury or mortality to fish under wild conditions (Halvorsen et al., 2012a; Kane et al., 2010; Popper et al., 2007). Potential direct injuries (e.g., barotrauma, hemorrhage or rupture of organs or tissue) from non-impulsive sound sources, such as sonar, are unlikely because of slow rise times, lack of a strong shock wave such as that associated with an explosive, and relatively low peak pressures.

The effects of mid-frequency sonar-like signals (1.5–6.5 kHz) on larval and juvenile Atlantic herring, Atlantic cod, saithe, and spotted wolffish (*Anarhichas minor*) were examined by Jorgensen et al. (2005). Researchers investigated potential effects on survival, development, and behavior in this study. Among fish kept in tanks and observed for one to four weeks after sound exposure, no significant differences in mortality or growth-related parameters between exposed and unexposed groups were observed. Examination of organs and tissues from selected herring experiments did not reveal obvious differences between unexposed and exposed groups. However, two (out of 42) of the herring groups exposed to continuous wave of 189 dB re 1 µPa and 179 dB re 1 µPa SPL had a post-exposure mortality of 19 and 30 percent, respectively. It is not clear if this increased mortality was due to the received level or to other unknown factors, such as exposure to the resonance frequency of the swim bladder (see Section D.1.1.8 for discussion of resonance). Jorgensen et al. (2005) estimated a resonant frequency of 1.8 kHz for herring and saithe ranging in size from 6.3 to 7.0 centimeters, respectively, which lies within the range of frequencies used during sound exposures and, therefore, may explain some of the noted mortalities. Frequency-modulated sonar signals of the same frequency range and intensities did not cause mortality.

Past research has demonstrated that fish species, size, and depth influence the resonant frequency (defined in Section D.1.1.8) of the swim bladder (Løvik & Hovem, 1979; McCartney & Stubbs, 1971). For example, lower frequencies (i.e., generally below 1 kHz) are expected to produce swim bladder resonance in adult fishes from about 10 to 100 centimeters (McCartney & Stubbs, 1971); higher frequencies, greater than 1 kHz, could produce swim bladder resonance in smaller fishes. At resonance, the swim bladder may absorb much of the acoustic energy in the impinging sound wave. It was hypothesized that the resulting oscillations may cause mortality or harm the auditory organs or the swim bladder (Jorgensen et al., 2005; Kvadsheim & Sevaldsen, 2005). However, damage to the swim bladder and to tissues surrounding the swim bladder was not observed in fishes exposed to multiple sonar pulses from approximately 165–195 dB re 1 µPa at their presumed swim bladder resonant frequency (Jorgensen et al., 2005). Fishes may be more susceptible to injury from swim bladder resonance when exposed to continuous signals within the resonant frequency range; although, based on the above studies, injury or mortality from swim bladder resonance under real-world conditions is unlikely.

Hastings (1991); (1995) tested the limits of acoustic exposure on two freshwater fish species. Hastings found “acoustic stunning” (loss of consciousness) in blue gouramis (*Trichogaster trichopterus*) following an eight-minute continuous exposure in captivity to a 150 Hz pure tone of 198 dB re 1 µPa SPL (1995). This species of fish has an air bubble in the mouth cavity directly adjacent to the animal’s braincase that may have caused this injury. Hastings (1991); (1995) also found that goldfish, exposed to a 250 Hz continuous wave sound with peak pressures of 204 dB re 1 µPa for two hours, and blue gourami exposed to a 150 Hz continuous wave sound at an SPL of 198 dB re 1 µPa for 0.5 hour did not survive.

To investigate potential injury to the auditory system in fishes, Sapozhnikova et al. (2020) exposed freshwater fish (peled, *Coregonus peled*) to tonal, 300 Hz sound at 176–186 dB re 1 µPa SPL (peak to peak), for up to 18 days. After exposure, cellular changes to hearing structures were assessed. Hair cell bundles of the saccule (one of three otolithic organs in the inner ear) were significantly less dense in sound-exposed fish compared to untreated controls, and changes were only observed for fish exposed longer than five days. Changes to otolith crystal structure and fusion of stereocilia ("hair-like" structures within the inner ear) like that observed after ototoxic antibiotic exposure were also observed after sound exposure, but no direct measurements of hearing loss were taken. The exposure was intended to simulate conditions of common aquaculture systems and therefore may not be applicable to exposures in other environments. Additionally, freshwater fishes are known to have better hearing than marine species, making them more susceptible to auditory impacts. This study does demonstrate some of the more severe physical impacts to the auditory system that could result from extremely long duration exposures to low-frequency tonal signals.

Although these studies (Hastings, 1991; Hastings, 1995; Sapozhnikova et al., 2020) illustrate some of the highest known exposures (long duration exposures to moderately high sound levels) of tonal signals on freshwater fishes with enhanced hearing capabilities, direct comparisons of these results to impacts from transitory signals (e.g., sonar or vessels) should be treated with caution. The conditions of the exposures (e.g., exposure duration, fish's inability to avoid the source) are not synonymous with conditions to an open ocean or coastal environment. Stunning and mortality due to exposure to non-impulsive sound exposure has not been observed in other studies.

Three freshwater species of fish, the rainbow trout, channel catfish, and the hybrid sunfish (*Lepomis* sp.), were exposed to both low- (170 to 320 Hz) and mid-frequency (2.8 to 3.8 kHz) sonar (Kane et al., 2010; Popper et al., 2007). Low-frequency exposures with received SPLs of 193 dB re 1 µPa occurred for either 324 or 648 seconds. Mid-frequency exposures of 210 dB re 1 µPa SPL occurred for 15 seconds. No fish mortality resulted from either experiment. During examination after test exposures, both studies found that none of the subjects showed signs of tissue damage related to exposure (Kane et al., 2010; Popper et al., 2007). As summarized in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), although fish have been injured and killed due to intense, long-duration, non-impulsive sound exposures, fish exposed under more realistic conditions have shown no signs of injury. In the absence of other proxies to rely upon, those species tested to date can be used as surrogates for estimating injury in other species exposed to similar sources.

D.5.6.2 Injury due to Impulsive Sound Sources

Non-explosive impulsive sounds, such as those produced by seismic air guns and impact pile driving, may cause injury or mortality in fishes. Mortality and potential damage to the cells of the lateral line have been observed in fish larvae, fry, and embryos after exposure to single shots from a seismic air gun within close proximity to the sound source (0.1–6 m) (Booman et al., 1996; Cox et al., 2012). However, exposure of adult pallid sturgeon (*Scaphirhynchus albus*) and paddlefish (*Polyodon spathula*) to a single shot from an air gun array (four air guns) within similar ranges (6 m) did not result in any signs of mortality within seven days after exposure (Popper et al., 2016). Although injuries occurred in adult fishes, they were like injuries seen in control subjects so there is little evidence that the air gun exposure solely contributed to the observed effects.

Injuries, such as ruptured swim bladders, hematomas, and hemorrhaging of other gas-filled organs, have been reported in fish exposed to a large number of simulated impact pile driving strikes with cumulative

SELs up to 219 dB re 1 µPa^{2}s under highly controlled settings where fish were unable to avoid the source (Casper et al., 2013a; Casper et al., 2012b; Casper et al., 2013b; Halvorsen et al., 2012a; Halvorsen et al., 2011, 2012b). However, it is important to note that these studies exposed fish to 900 or more strikes as the studies aimed to evaluate the equal energy hypothesis, which suggests that the effects of a large single pulse of energy is equivalent to the effects of energy received from many smaller pulses (as discussed in Smith & Gilley, 2008). Halvorsen et al. (2011) and Casper et al. (2017) propose that the equal energy hypothesis does not apply to effects of pile driving. Specifically, Casper et al. (2017) found the amount of energy in each pile strike had a larger influence on resulting injuries than the number of strikes even when the SEL was equal. For example, hybrid striped bass (white bass x striped bass) exposed to fewer strikes with higher single strike sound exposure values resulted in a higher number of, and more severe, injuries than bass exposed to an equivalent cumulative SEL that contained more strikes with lower single strike sound exposure values. This is important to consider when comparing data from pile driving studies to potential effects from an explosion. Although single strike peak SPLs were measured during these experiments (at average levels of 207 dB re 1 µPa), the injuries were only observed during exposures to multiple strikes; therefore, it is anticipated that a peak value much higher than the reported values would be required to lead to injury in fishes exposed to a single strike or explosion.

The studies discussed in the paragraph above included species both with and without swim bladders. Most fishes that exhibited injuries were those with swim bladders. Lake sturgeon (*Acipenser fulvescens*), a physostomous fish, was found to be less susceptible to injury from impulsive sources than Nile tilapia or hybrid striped bass, both of which are physoclistous fishes (Casper et al., 2017; Halvorsen et al., 2012a). Halvorsen et al. (2012a) proposed that the difference in results is likely due to the type of swim bladder present in each species. Physostomous fishes have an open duct connecting the swim bladder to their esophagus and may be able to quickly adjust the amount of gas in their body by gulping or releasing air. Physoclistous fishes do not have this duct; instead, special tissues or glands regulate gas pressure in the swim bladder and are unable to react quickly enough to reduce pressure appreciably in response to an impulsive sound stressor. There were no mortalities reported during these experiments, and in the studies where recovery was observed, most exposure related injuries healed within a few days in a laboratory setting. In many of these controlled studies, neutral buoyancy was determined in the fishes prior to exposure to the simulated pile driving. However, fishes with similar physiology to those described in these studies that are exposed to actual pile driving activities may not be neutrally buoyant at the onset of an exposure and therefore may show varying levels of injury depending on their state of buoyancy.

By exposing caged juvenile European sea bass (*Dicentrarchus labrax*) to actual pile driving operations, Debusschere et al. (2014) confirmed the results discussed above. No differences in mortality were found between control and experimental groups (215–222 dB re 1 µPa^{2}s SEL), and many of the same types of injuries occurred (Casper et al., 2013a; Casper et al., 2012b; Casper et al., 2013b; Halvorsen et al., 2012a; Halvorsen et al., 2011, 2012b).

Other potential effects from exposure to impulsive sound sources include bubble formation and neurotrauma. It is speculated that high SPLs may cause bubbles to form from micronuclei in the blood stream or other tissues of animals, possibly causing embolism damage (Hastings & Popper, 2005). Fishes have small capillaries where these bubbles could be caught, leading to vessel rupture and internal bleeding. It has also been speculated that this phenomena could take place in the eyes of fish due to potentially high gas saturation within the eye tissues (Popper & Hastings, 2009b). Additional research is necessary to verify if these speculations apply to exposures to non-impulsive sources such as sonars. These phenomena have not been well studied in fishes and are difficult to recreate under real-world conditions.

As summarized in the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), exposure to high intensity and long duration impact pile driving or air gun shots has not caused mortality, and fishes typically recovered from injuries in controlled laboratory settings. Barring other proxies to rely upon, species tested to date can be used as surrogates for investigating injury in other species exposed to similar sources (Popper et al., 2014).

D.5.6.3 Injury due to Explosions

The blast wave from an explosion is lethal to fishes at close range, causing massive organ and tissue damage (Keevin & Hempen, 1997). At greater distance from the detonation point, the extent of mortality or injury depends on many factors including fish size, body shape, depth, physical condition of the fish, and, perhaps most importantly, the presence of a swim bladder. In general, fishes without swim bladders have been shown to be more resilient to explosives compared to those with swim bladders (Gaspin, 1975; Gaspin et al., 1976; Goertner et al., 1994). At the same distance from the source, larger fishes and those with elongated forms that are round in cross-section were generally less susceptible to death or injury than smaller fishes and deep-bodied forms, and fishes oriented sideways to the blast suffer the greatest impact (O'Keeffe, 1984; O'Keeffe & Young, 1984; Wiley et al., 1981; Yelverton et al., 1975).

If a fish is close to an explosive detonation, the exposure to rapidly changing high pressure levels can cause barotrauma. Barotrauma is injury due to a sudden difference in pressure between an air space inside the body and the surrounding water and tissues. Rapid compression followed by rapid expansion of airspaces, such as the swim bladder, can damage surrounding tissues and result in the rupture of the airspace itself. The swim bladder is the primary site of damage from explosives (Dahl et al., 2020; Wright, 1982; Yelverton et al., 1975). Gas-filled swim bladders resonate at different frequencies than surrounding tissue and can be torn by rapid oscillation between high- and low-pressure waves (Goertner, 1978). Swim bladders are a characteristic of most bony fishes, with the notable exception of some flatfishes (e.g., halibut). Sharks and rays are examples of cartilaginous fishes which lack a swim bladder. Small airspaces, such as micro-bubbles that may be present in gill structures, could also be susceptible to oscillation when exposed to the rapid pressure increases caused by an explosion. This may have caused the bleeding observed on gill structures of some fish exposed to explosions (Goertner et al., 1994). Sudden very high pressures can also cause damage at tissue interfaces due to the way pressure waves travel differently through tissues with different densities. Rapidly oscillating pressure waves might rupture the swim bladder, kidney, liver, and spleen and cause venous hemorrhaging (Dahl et al., 2020; Jenkins et al., 2022; Keevin & Hempen, 1997).

Several studies have exposed fish to explosives and examined various metrics in relation to injury susceptibility. Sverdrup et al. (1994) exposed Atlantic salmon in a laboratory setting to repeated shock pressures of around 2 megapascals (300 psi or 246 dB re 1 µPa peak) without any immediate or delayed mortality after a week. Hubbs and Rechnitzer (1952) exposed fish to underwater detonations placed either on the seafloor or buried at various depths along an underwater canyon in La Jolla, California. Data from this experiment showed that when near the surface, fishes began to exhibit injuries around peak pressure exposures of 40–70 psi (229 to 234 dB re 1 µPa peak). However, near the bottom (all water depths were less than 100 feet [ft.]) fish exposed to pressures over twice as high exhibited no sign of injury. Yelverton et al. (1975) found that peak pressure was not correlated to injury susceptibility; instead, injury susceptibility of swim bladder fish at shallow depths (10 ft. or less) was correlated to the metric of positive impulse (pascal seconds [Pa-s]), which takes into account the positive peak pressure, the duration of the positive pressure exposure, and fish mass, with smaller fish being more susceptible than larger fishes.

Two experiments reported the effects of underwater explosions on Pacific sardines (*Sardinops sagax*) and Pacific mackerel (*Scomber japonicus*) to underwater detonations of C4 explosives at the same general test site off the coast of California, though the experiments took place during different years (Dahl et al., 2020; Jenkins et al., 2022). In both efforts, fish were stationed at various distances (out to approximately 800 m) prior to the explosion, in addition to a control group that was not exposed. Necropsies following explosions observed significant injuries, including fat hematoma, kidney rupture, swim bladder rupture, and reproductive blood vessel rupture. Injuries decreased with increasing distance from the explosion, and swim bladder injuries were the most prevalent. While most significant injuries were consistently present at close range (less than 50 m, approximately 240 dB re 1 µPa peak) with decreasing proportion of injury farther from the source in both studies, Dahl et al. (2020) found inconsistent findings at the 50–125 m range (approximately 240 – 232 dB re 1 µPa peak). The inconsistency in observed physical damage at this distance from the detonation was due to possible acoustic refraction effects, including waveform paths that were bottom reflected, surface reflected, or a combination of both. Some fish mortality was observed during the Jenkins et al. (2022) experiment, in a portion of cages at or within 157 m (received level of 231 dB re 1 µPa peak) of the explosion. Additionally, unique video footage from a subset of treatment groups showed most fish at or within 257 m (a peak SPL of 224 dB re 1 µPa) were stunned (immobilized) immediately following exposure. Unlike Yelverton et al. (1975), the statistical model demonstrated that while all three acoustic measures were good predictors of injury, peak pressure and SEL were better predictors of injury than pressure impulse.

Multiple fishes with a swim bladder were exposed to explosions of varying sizes across a variety of depths (Gaspin, 1975; Gaspin et al., 1976). Subsequently, a swim bladder oscillation model was developed, which showed that the severity of injury observed in those tests could be correlated to the extent of swim bladder expansion and contraction predicted to have been induced by exposure to the explosive blasts (Goertner, 1978; Wiley et al., 1981). Per this model, the degree of swim bladder oscillation is affected by ambient pressure (i.e., depth of fish), peak pressure of the explosive, duration of the pressure exposure, and exposure to surface rarefaction (negative pressure) waves. The maximum potential for injury is predicted to occur where the surface reflected rarefaction (negative) pressure wave arrives coincident with the moment of maximum compression of the swim bladder caused by exposure to the direct positive blast pressure wave, resulting in a subsequent maximum expansion of the swim bladder. Goertner (1978) and Wiley et al. (1981) found that their swim bladder oscillation model explained the injury data in the Yelverton et al. (1975) exposure study and that the Yelverton and Richmond (1981) impulse parameter was applicable only to fishes at shallow enough depths to experience less than one swim bladder oscillation before being exposed to the following surface rarefaction wave.

O'Keeffe (1984) provides calculations and contour plots that allow estimation of the range to potential effects of explosions at or near the surface of the water on fish possessing swim bladders using the damage prediction model developed by Goertner (1978). O'Keeffe (1984) parameters include the charge weight, depth of burst, and the size and depth of the fish, but the estimated ranges do not consider unique propagation environments that could reduce or increase the range to effect. Based on these calculations, it was suggested that fish at greater depths and near the surface are predicted to be less likely to be injured because geometries of the exposures would limit the amplitude of swim bladder oscillations. In contrast, detonations at or near the surface, like most proposed activities that utilize bombs, missiles, and gunfire, would result in surface blow off (i.e., loss of energy into the air), resulting in lower overall ranges to effects.

Studies that have documented caged fishes killed during planned underwater explosions indicate that most fish that die do so within one to four hours, after exposure and almost all die within 24-hours (Yelverton et al., 1975). Mortality in free-swimming (uncaged) fishes may be higher due to increased susceptibility to predation. Fitch and Young (1948) found that the type of free-swimming fish killed changed when blasting was repeated at the same location within 24 hours of previous blasting. They observed that most fish killed on the second day were scavengers, presumably attracted by the victims of the previous day’s blasts.

Fitch and Young (1948) also investigated whether a significant portion of fish killed would have sunk and not been observed at the surface. Comparisons of the numbers of fish observed dead at the surface and at the bottom in the same affected area after an explosion showed that fish found dead on the bottom comprised less than 10 percent of the total observed mortality. Gitschlag et al. (2000) conducted a more detailed study of both floating fishes and those that were sinking or lying on the bottom after explosive removal of nine oil platforms in the northern Gulf of Mexico. Results were highly variable. They found that 3–87 percent (46 percent average) of the red snapper killed during a blast might float to the surface. Currents, winds, and predation by seabirds or other fishes may be some of the reasons that the magnitude of fish mortality may not have been accurately captured.

There have been few studies of the impact of underwater explosives on early life stages of fish (eggs, larvae, juveniles). Fitch and Young (1948) reported mortality of larval anchovies scooped opportunistically during underwater blasting off the coast of California. Detonations used during these operations varied in size (from 10 to 160 pounds), with some explosives placed just beneath the water’s surface and others buried under the seafloor. Although the authors mention observations of live fish within the “lethal range” of these detonations, specific distances and search patterns were not provided. Another experiment reported dead anchovy and smelt larvae within approximately 50 m of buried charges weighing from 90 to 180 pounds in a shallow water channel during a pipeline construction project (Nix & Chapman, 1985). Although this provides useful insight into potential impacts to fishes from explosives, acoustic measures were not taken during either of these studies to correlate mortality with received levels. Similar to adult fishes, the presence of a swim bladder contributes to shock wave-induced internal damage in larval and juvenile fish (Settle et al., 2002). Explosive shock wave injury to internal organs of larval pinfish (*Lagodon rhomboids*) and spot (*Leistomus xanthurus*) exposed at shallow depths was documented at impulse levels similar to those predicted by Yelverton et al. (1975) for very small fish and provide the lowest measured received level that injuries have been observed in larval fish (peak SPLs of 220 dB re 1 µPa) (Govoni et al., 2003; Govoni et al., 2008; Settle et al., 2002). Researchers have suggested that egg mortality may be correlated with peak particle velocity exposure [i.e., the localized movement or shaking of water particles, as opposed to the velocity of the blast wave (Faulkner et al., 2006; Faulkner et al., 2008; Jensen, 2003)], although sufficient data from direct explosive exposures is not available.

Observations of the inner ear and lateral line across fishes exposed to explosives are lacking. Smith et al. (2022) provide the first examination of the physical effects of underwater explosions on the inner ear of Pacific mackerel (*Scomber japonicus*). Results showed varying amounts of hair cell loss as well as evidence of hair cell shearing and even holes in the epithelial tissue along the saccule related to the explosive exposure. Significant impacts were observed starting at peak SPLs of 220 dB re 1 µPa. Additional impacts on these sensory system organs have been observed during exposure to other impulsive sources such as air guns and playbacks of impact pile driving noise, which would indicate that similar effects may be possible in fishes exposed to explosions (Booman et al., 1996; Casper et al., 2013a; McCauley et al., 2003). Rapid pressure changes could cause mechanical damage to sensitive ear structures due to differential movements of the otolithic structures. Bleeding near otolithic structures was the most commonly observed injury in non-swim bladder fish exposed to a close explosive charge (Goertner et al., 1994). Additional research is needed to understand the potential for sensory cell damage from explosive exposures, the severity and implication of such affects for individual fish, and at what sound levels these impacts may occur.

As summarized by the *ANSI Sound Exposure Guideline* technical report (Popper et al., 2014), exposure to explosive energy poses the greatest potential threat for injury and mortality in marine fishes. Fishes with a swim bladder are more susceptible to injury than fishes without a swim bladder. The susceptibility also probably varies with size and depth of both the detonation and the fish. Fish larvae or juvenile fish may be more susceptible to injury from exposure to explosives.

D.5.7 LONG-TERM CONSEQUENCES

Mortality removes an individual fish from the population and injury can reduce the fitness of an individual. Fishes with injuries from any sound exposure may not survive in the wild due to harsher conditions and risk of predation. They may also have long-term competitive disadvantages for prey and mates, relative to uninjured individuals of the same species. Few studies have been conducted on any long-term consequences from repeated hearing loss, stress, or behavioral reactions in fishes due to exposure to loud sounds (Hawkins et al., 2015; Popper & Hastings, 2009a; Popper et al., 2014).

Repeated exposures of an individual to multiple sound-producing activities over a season, year, or life stage could cause reactions with costs that can accumulate over time to cause long-term consequences for the individual. These long-term consequences may affect the survivability of the individual, or if impacting enough individuals may have population-level effects, including alteration from migration paths, avoidance of important habitat, or even cessation of foraging or reproductive behavior (Hawkins et al., 2015). For example, Soudijn et al. (2020) attempted to design a theoretical population consequences model without quantitative data on SELs. Atlantic cod energy expenditure, food intake, mortality rate, and reproductive output were analyzed to assess cod's potential impacts from sound exposure. The model predicted decreased food intake, increased energy expenditure, and decreased population growth rate because of increased continuous noise. Models such as these are common among other taxa and often come to similar conclusions. Conversely, some animals may habituate to or become tolerant of repeated exposures over time, learning to ignore a stimulus that in the past has not accompanied any overt threat. In fact, Sivle et al. (2016a) and Sivle et al. (2015a) predicted that exposures to sonar at the maximum levels tested would only result in short-term disturbance and would not likely affect the overall population in sensitive fishes such as Atlantic herring. Additional research is needed to understand the complex relationship of sound exposure to potential long-term consequences to individuals and populations.

D.6 MARINE MAMMALS

This section describes general effects to marine mammals from exposure to acoustic sources.

D.6.1 HEARING

The typical mammalian ear has an outer ear that collects and transfers sound to the eardrum and then to the middle ear (Fay & Popper, 1994; Rosowski, 1994). The middle ear contains bones that amplify and transfer acoustic energy to the inner ear, which contains sensory cells (called hair cells) that transform acoustic energy into electrical signals. Those electrical signals are then carried by the auditory nerve to the brain (Møller, 2013).

All marine mammals display some modifications to the typical mammalian ear; furthermore, there are differences between the hearing of marine mammals that are fully aquatic and those that are amphibious – or live partially out of the water (Wartzok & Ketten, 1999). Marine mammals with an amphibious ear include the marine carnivores: pinnipeds, sea otters, and polar bears (Ghoul & Reichmuth, 2014b; Owen & Bowles, 2011; Reichmuth et al., 2013). Outer ear adaptations in this group include outer ears that are reduced or absent, and in seals, specialized tissues that act as valves to seal off water from entering the ear canal when submerged (Wartzok & Ketten, 1999). In marine mammals with a fully aquatic ear (cetaceans and sirenians), bone and fat channels in the head conduct sound to the ear; while the ear canal still exists, it is narrow and sealed, and outer ears are absent (Castellini et al., 2016; Ketten, 1998) (see Figure D.6-1). These adaptations reflect specializations for hearing in both air and water for amphibious marine mammals, and for hearing in water for fully aquatic marine mammals.

Amphibious
California sea lion

Amphibious
Harbor seal

Aquatic
Bottlenose dolphin

Notes: The amphibious California sea lion outer ear is reduced compared to terrestrial mammals, while the harbor seal lacks an outer ear and has specialized valve-like tissue to close off the ear canal from water. The aquatic bottlenose dolphin lacks an outer ear and has a drastically reduced pinhole-like ear canal yet has specialized hearing for underwater sounds.

Source: https://pediaa.com/difference-between-seal-and-sea-lion, https://www.shutterstock.com/pic.mhtml?id=69136297

Figure D.6-1: Examples of Marine Mammal Ears

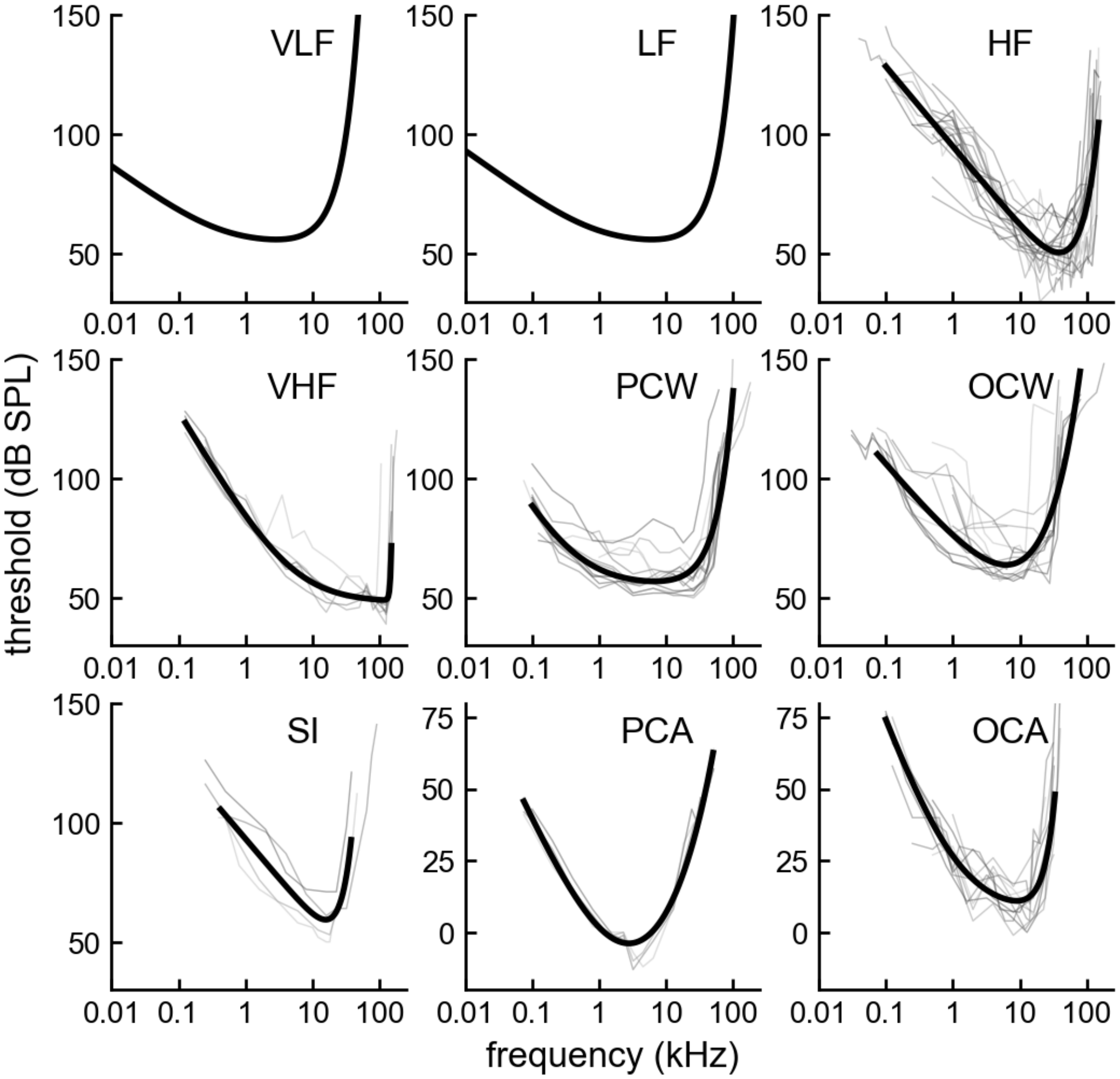
Marine mammal audiograms, like those of terrestrial mammals, typically have a “U-shape,” with a frequency region of best hearing sensitivity at the bottom of the “U” and a progressive decrease in sensitivity outside of the range of best hearing (Southall et al., 2019c) (see Figure D.1-8).

Direct measurements of hearing sensitivity exist for about a quarter of the nearly 130 species of marine mammals. Marine mammals are arranged into the following functional hearing groups based on their generalized hearing sensitivities: very high-frequency cetaceans (VHF group: porpoises, Kogia spp.), high-frequency cetaceans (HF group: delphinids, beaked whales, sperm whales), low-frequency cetaceans (LF group: mysticetes), sirenians (SI group: manatees), otariids and other non-phocid marine carnivores in water and air (OCW and OCA groups: sea lions, otters), and phocids in water and air (PCW and PCA groups: true seals) (Southall et al., 2019c). Representative composite audiograms (U.S. Department of the Navy, 2024a) have been created for each functional hearing group using audiograms from published literature (see Figure D.6-2).

Since the composite audiograms were developed for this analysis, Houser et al. (2022) published new AEP audiograms for stranded odontocetes of six species for which no audiograms had previously existed: dwarf sperm whale (*Kogia sima*), pygmy sperm whale (*Kogia breviceps*), northern right whale dolphin (*Lissodelphis borealis*), melon-headed whale (*Peponocephala electra*), long-beaked common dolphin (*Delphinus capensis*), and Atlantic spotted dolphin (*Stenella frontalis*). Hearing data was also provided on the pygmy killer whale (*Feresa attenuata*). The audiograms had frequency ranges, shapes, and upper frequency limits that were generally consistent with the hearing groups in which these species are categorized (see Table 2 in U.S. Department of the Navy, 2024a).

For marine mammals that are impractical to test or have limited hearing data (e.g., mysticete whales and rare species), some aspects of hearing can be estimated from anatomical structures, frequency content of vocalizations, behavioral responses to sound and inferences from related species (U.S. Department of the Navy, 2024a). For example, behavioral responses of gray whales suggests that they can hear 21 - 25 kHz signals (Frankel & Stein, 2020). The only hearing measurement in a mysticete (minke whale) suggests that LF cetaceans have a upper-frequency limit of hearing between 45 and 90 kHz (Houser et al., 2024). Although there have been no direct measurements of hearing sensitivity in larger mysticetes (VLF hearing group), an audible frequency range of approximately 10 Hz to 30 kHz has been estimated from measured vocalization frequencies, observed reactions to playback of sounds, and anatomical analyses of the auditory system (Cranford & Krysl, 2015; Houser et al., 2001a). See the technical report titled Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase IV) (U.S. Department of the Navy, 2024a) for a complete description of marine mammal composite audiograms.

Research has shown that hearing in marine mammals is directional: the relative angle between the sound source location and the animal's position affects the hearing threshold. This is important because how an animal perceives sound is dependent on the hearing threshold. For example, a sound presented from directly in front of an animal might be heard clearly, while the same sound presented from directly behind an animal might not be heard. For bottlenose dolphins, hearing sensitivity becomes more directional as the sound frequency increases, with the greatest sensitivity to sounds presented in front and below the dolphin (Accomando et al., 2020; Au & Moore, 1984). Hearing sensitivity is asymmetrical in the vertical and horizontal planes, which might be beneficial for localizing a sound source. Harbor porpoises and belugas exhibit direction-dependent hearing, but to a lesser degree than the dolphin (Kastelein et al., 2019b; Kastelein et al., 2005a; Popov & Supin, 2009). Based on experiments in harbor seals, phocids likely have well-developed directional hearing for biologically relevant sounds (Byl et al., 2016; Byl et al., 2019). Directional hearing is important to consider when assessing masking effects.



Notes: For hearing in water (top two rows) and in air (bottom row, phocids and otariids only). VLF = very low frequency; LF = low frequency; HF = high frequency; VHF = very high frequency; PCW = phocids in water; OCW = otariids and other non-phocid marine carnivores in water; SI = sirenians in water; OCA = otariids and other non-phocid marine carnivores in air; PCA = phocids in air

Figure D.6-2: Composite Audiograms used in Marine Mammal Hearing Criteria and Thresholds

D.6.2 ACOUSTIC SIGNALING

Like the diversity of hearing capabilities among species, the wide variety of acoustic signals used in communication and echolocation is reflective of the diverse characteristics of marine mammal species. Detailed reviews of sounds generated by marine mammals are available, see Chapter 7 of Richardson et al. (1995b) and Table 4-1 in Wartzok and Ketten (1999). A general division can be drawn between lower frequency communication signals including vocalizations that are produced by all marine mammals, and the specific, high-frequency echolocation (i.e., biosonar) signals that are used by odontocetes to sense their environment. The general types and frequency characteristics of marine mammal vocalizations are described in Table D.6-1.

Table D.6-1: Marine Mammal Vocalizations

Signal type	*Description*	*Marine mammal group(s)*	*Frequency range*[1]
Echolocation	Broadband, short-duration, high-source level clicks serving a primarily sensory function with a secondary communication function[2].	HF & VHF cetaceans	20 – 160 kHz
Communication	Tonal (e.g., whistles) and non-tonal (e.g., grunts) with a wide variety of durations and source levels and serving primarily for communication (e.g., mating, mother-calf contact, group cohesion/coordination, and other social functions).	VLF cetaceans	0.01 – 0.4 kHz
		LF cetaceans	0.1 – 4 kHz
		HF & VHF cetaceans	4 – 30 kHz
		Sirenians	0.6 – 16 kHz
		Pinnipeds (phocids, otariids)	0.1 – 30 kHz
		Otters	3 – 5 kHz
		Polar bears	0.2 – 1 kHz

[1] The frequencies near maximum energy based on Table 4-1 in Wartzok and Ketten (1999).

[2] Sperm whales use clicks to echolocate and specific click patterns primarily to communicate. Some other species might also use click patterns that function primarily to communicate.

D.6.2.1 Communication

Communication sounds have crucial functions including social (e.g., mating), maintaining mother-calf contact, group cohesion, feeding, and other purposes. Communication signals include calls (i.e., vocalizations) and sounds produced by non-vocal behaviors such as tail/fluke slaps on the water surface or clapping the jaw. Vocalizations might have a tonal quality or pitch resulting from a prominent fundamental frequency, such as whistles in some odontocetes and sirenian calls (Brady et al., 2021), or they might be less tonal because of energy distributed across a wide frequency range such as grunts produced by marine carnivores like pinnipeds. Aerial vocalizations are produced by pinnipeds, otters, and polar bears. The acoustic characteristics of communication signals of marine mammals are quite diverse but can be generally classified as having dominant energy at frequencies between approximately 20 Hz and 30 kHz (Richardson et al., 1995b; Wartzok & Ketten, 1999).

Of note are the lower frequency calls of mysticete whales that range from tens of Hz to several kHz and have source levels of approximately 150–200 dB re 1 µPa. Typically, mysticete calls have fundamental frequencies below 200 Hz. Fin whales and blue whales make exceptionally low frequency calls (10 -16 Hz), while humpback whales make higher frequency calls having harmonics that exceed 20 kHz, (Au et al., 2006; Cummings & Thompson, 1971; Edds-Walton, 1997; Širović et al., 2007; Stimpert et al., 2007; Wartzok & Ketten, 1999). These calls most likely serve social functions such as interspecific attraction or detection over long distances but could serve an orientation function as well (Frazer & Mercado, 2000; Green, 1994; Green et al., 1994; Mercado, 2021; Richardson et al., 1995b).

D.6.2.2 Echolocation

Odontocete cetaceans generate short-duration (50–200 microseconds), high-frequency (10 – 200 kHz peak frequency), specialized echolocation clicks (e.g., biosonar) used to detect, localize, and characterize underwater objects (Au, 1993; Wartzok & Ketten, 1999). This process is essential for hunting, including searching, tracking, and capturing prey. Echolocation clicks are often more intense than communicative signals, with reported source levels as high as 229 dB re 1 µPa peak-to-peak (Au et al., 1974). The echolocation clicks of very high-frequency cetaceans (e.g., porpoises) are narrower in bandwidth (i.e., the difference between the upper and lower frequencies in a sound) and higher in frequency than those of high-frequency cetaceans (Madsen et al., 2005; Villadsgaard et al., 2007). The specific characteristics of echolocation signals such as their repetition patterns and peak frequency can be used to identify species (Baumann-Pickering et al., 2013).

Echolocation can serve communicative functions even though clicks are not usually produced for this purpose by most odontocetes. For example, eavesdropping animals may hear rapid echolocation clicks and other sounds associated with feeding to find food or avoid predators, and sperm whale clicks may reveal the size or general characteristics of the clicking individual. However, some types of clicks or patterns of clicks are thought to be produced for the purpose of communication. For example, click patterns called codas are communicative vocalizations produced by sperm whales (Jacobs et al., 2024; Richardson et al., 1995b; Watkins & Schevill, 1977).

D.6.2.3 Relationship between Hearing and Vocalization

In general, frequency ranges of sounds produced by a species lie within the audible frequency range for that species (i.e., animals vocalize within their audible frequency range). However, auditory frequency range and vocalization frequencies do not perfectly align. For example, odontocete echolocation clicks contain a broad range of frequencies, and not all the frequency content is necessarily heard by the individual that emitted the click. The frequency range of vocalization in a species can therefore be used to infer some characteristics of their hearing capabilities; however, caution must be taken when considering vocalization frequencies alone in predicting the hearing capabilities of species for which data are absent or limited such as mysticete whales.

Aspects of vocalization and hearing sensitivity are subject to evolutionary pressures that are not solely related to communication within the species. For example, hearing and vocalization is influenced by the need to detect or avoid threats such as predators (e.g., Deecke et al., 2002) and listening for prey-generated sounds. Additionally, high-frequency hearing is advantageous to animals with small heads because it facilitates sound localization based on differences in sound levels at each ear (Heffner & Heffner, 1982). These factors might be partially responsible for the difference in best hearing thresholds and dominant vocalization frequencies in some species of marine mammals (e.g., Steller sea lions, Mulsow & Reichmuth, 2010).

D.6.3 HEARING LOSS AND AUDITORY INJURY

All mammals experience normal age-related hearing loss (presbycusis), which is a progressive reduction in the ability to hear higher frequencies that spreads to lower frequencies over time. This type of hearing loss is due to the loss of sensory cells in the inner ear and degeneration of the pathways that connect the ear to the brain. Age-related hearing loss occurs over a lifetime and is distinct from acute noise-induced hearing loss (Møller, 2013).

Noise-induced hearing loss can be temporary (i.e., temporary threshold shift, or TTS) or permanent (i.e., permanent threshold shift, or PTS), and higher-level sound exposures are more likely to cause PTS or other auditory injury. For marine mammals, auditory injury (AINJ) is considered to be possible when sound exposures are sufficient to produce 40 dB of TTS measured approximately four minutes after exposure (U.S. Department of the Navy, 2024a).

Numerous studies have directly examined noise-induced hearing loss in marine mammals. In these studies, hearing thresholds were measured in marine mammals before and after exposure to intense sounds. The difference between the post-exposure and pre-exposure hearing thresholds is used to determine the amount of TTS in dB that was produced as a result of the sound exposure. The data from these studies is detailed in (U.S. Department of the Navy, 2024a) and the major findings are outlined in Table D.6-2Table D.6-2.

Table D.6-2: Major Findings from Studies of Threshold Shift in Marine Mammals

Major Finding	***Supporting Scientific Studies***
Hearing test method	
The method used to test hearing may affect the resulting amount of measured temporary threshold shift (TTS), with auditory evoked potential measures producing larger amounts of TTS compared to behavioral measures.	Finneran (2015); Finneran et al. (2007)
Effect of frequency and sound pressure level (SPL)	
Sound exposures of a narrow frequency range can produce TTS over a large frequency range.	Finneran et al. (2007); Kastelein et al. (2020a); Kastelein et al. (2019d); Kastelein et al. (2019f); Mooney et al. (2009a); Nachtigall et al. (2004); Popov et al. (2013); Popov et al. (2011); Reichmuth et al. (2019); Schlundt et al. (2000)
As the exposure SPL increases, the frequency at which the maximum TTS occurs also increases.	Finneran et al. (2007); Kastelein et al. (2020a); Kastelein et al. (2019d); Kastelein et al. (2019f); Kastelein et al. (2014a); Mooney et al. (2009a); Nachtigall et al. (2004); Popov et al. (2013); Popov et al. (2011); Reichmuth et al. (2019); Schlundt et al. (2000)
Sounds at frequencies well below the region of best sensitivity are generally less hazardous than those near the region of best sensitivity.	Finneran and Schlundt (2013); Kastelein et al. (2020a); Kastelein et al. (2019d); Kastelein et al. (2019f); (Gransier & Kastelein, 2024)
Effect of exposure duration, sound exposure level (SEL), and multiple exposures	
The amount of TTS increases with exposure SPL and duration and is correlated with SEL, but duration of the exposure has a more significant effect on TTS than would be predicted based on SEL alone. As the exposure duration increases, the relationship between TTS and SEL begins to break down.	Finneran et al. (2010b); Kastak et al. (2007); Kastak et al. (2005); Kastelein et al. (2014a); Mooney et al. (2009a); Popov et al. (2014); (Gransier & Kastelein, 2024)
TTS can accumulate across multiple exposures, but the resulting TTS will be less than the TTS from a single, continuous exposure with the same SEL. This means that TTS predictions based on the total, cumulative SEL will overestimate the amount of TTS from intermittent exposures such as sonars and impulsive sources[1].	Finneran et al. (2010b); Finneran et al. (2000); Finneran et al. (2002); Kastelein et al. (2015a); Kastelein et al. (2018a); Kastelein et al. (2014a); Mooney et al. (2009b); Reichmuth et al. (2016)
Growth of TTS and occurrence of permanent threshold shift (PTS)	
Gradual growth of TTS with increased levels of SEL typically occurs before onset of PTS. However, it is possible for PTS to occur without observing gradual growth of TTS or behavioral changes.	Reichmuth et al. (2019)

Table D.6-2: Major Findings from Studies of Threshold Shift in Marine Mammals (continued)

Major Finding	*Supporting Scientific Studies*
Recovery from TTS over time	
The time required for complete recovery of hearing depends on the magnitude of the initial shift; for relatively small shifts recovery may be complete in a few minutes, while large shifts may require several days for recovery. Recovery times are consistent for similar-magnitude TTS, regardless of the type of sound exposure (impulsive, continuous noise band, or sinusoidal wave).	Finneran et al. (2010a, 2010b); Finneran and Schlundt (2013); Kastelein et al. (2012a); Kastelein et al. (2012b); Kastelein et al. (2013a); Kastelein et al. (2019e); Kastelein et al. (2014a); Kastelein et al. (2014b); Kastelein et al. (2014c); Popov et al. (2014); Popov et al. (2013); Popov et al. (2011).
Under many circumstances TTS recovers linearly with the logarithm of time.	Finneran et al. (2010a, 2010b); Finneran and Schlundt (2013); Kastelein et al. (2012a); Kastelein et al. (2012b); Kastelein et al. (2013a); Kastelein et al. (2014a); Kastelein et al. (2014b); Kastelein et al. (2014c); Popov et al. (2014); Popov et al. (2013); Popov et al. (2011).

[1] In most acoustic impact assessments, the scenarios of interest involve shorter duration exposures than the marine mammal experimental data from which impact thresholds are derived; therefore, use of SEL tends to over-estimate the amount of TTS. Despite this, SEL continues to be used in many situations because it is relatively simple, more accurate than SPL alone, and lends itself easily to scenarios involving multiple exposures with different SPL and multiple sources.

Notes: PTS = permanent threshold shift; SEL = sound exposure level; SPL = sound pressure level; TTS = temporary threshold shift

The data from studies of hearing (i.e., composite audiograms, Figure D.6-2) and hearing loss in marine mammals were used to generate exposure functions – or predictions of hearing loss based on sound frequency, level, and type (continuous or impulsive) – for each hearing group (U.S. Department of the Navy, 2024a).

D.6.3.1 TTS Growth and Recovery

SEL is used to predict TTS in marine mammals based on available data (U.S. Department of the Navy, 2024a). These predictions likely hold true for shorter duration exposures, but for longer-duration exposures, SEL likely overestimates TTS (see Table D.6-2). In general, TTS increases with SEL in a non-linear fashion (Finneran, 2015). For lower SEL exposures, TTS will increase at a steady rate, but at higher SELs, TTS will either increase more rapidly or plateau (see U.S. Department of the Navy, 2024a).

Small amounts of TTS (a few dB) typically begin to recover immediately after the sound exposure and may fully recover in minutes, while larger amounts of TTS take longer to recover. Studies have also found substantial individual variation both in the amount of TTS produced by similar SELs (Kastelein et al., 2012a; Popov et al., 2013), and in recovery from similar TTS (Finneran, 2015; Kastelein et al., 2019e). For example, one harbor seal began recovering immediately after a 34 dB TTS, while a 45 dB TTS in another harbor seal only began recovering 4 - 24 hours after the exposure ended and complete recovery was observed after four days (Kastelein et al., 2020b). In general, recovery from TTS occurs linearly with the logarithm of time (Finneran, 2015).

Most of these findings are from studies that used continuous sound exposures, but intermittent, impulsive sound exposures have also been tested. The sound resulting from an explosive detonation is considered an impulsive sound and shares important qualities (i.e., short duration and fast rise time) with other impulsive sounds such as those produced by air guns, although explosive signals are characterized by sharper rises and higher peak pressures. There are no direct measurements of hearing loss in marine mammals due to exposure to explosive sources. Few studies using impulsive sounds have produced enough TTS to make predictions about hearing loss due to this source type (see U.S. Department of the Navy, 2024a). In general, predictions of TTS based on SEL for this type of sound exposure is likely to overestimate TTS because some recovery from TTS may occur in the quiet periods between impulsive sounds – especially when the duty cycle is low. Peak SPL (unweighted) is also used to predict TTS due to impulsive sounds (Southall et al., 2007; Southall et al., 2019c; U.S. Department of the Navy, 2024a).

D.6.3.2 Self-Mitigation of Hearing Sensitivity

Several studies have shown that certain odontocete cetaceans (toothed whales) may learn to reduce their hearing sensitivity (presumably to protect their hearing) when warned of an impending intense sound exposure or the duty cycle is predictable (Finneran, 2018; Finneran et al., 2024; Nachtigall & Supin, 2013, 2014, 2015; Nachtigall et al., 2015; Nachtigall et al., 2016a, 2018; Nachtigall et al., 2016b). The effect has been demonstrated in the false killer whale (*Pseudorca crassidens*) (Nachtigall & Supin, 2013), bottlenose dolphin (*Tursiops truncatus*) (Finneran, 2018; Nachtigall & Supin, 2014, 2015; Nachtigall et al., 2016b), beluga (*Delphinapterus leucas*) (Nachtigall et al., 2015), and harbor porpoise (*Phocoena phocoena*) (Nachtigall et al., 2016a).

Based on these experimental measurements with captive odontocetes, it is possible that wild odontocetes would also suppress their hearing if they could anticipate an impending, intense sound, or during a prolonged exposure (even if unanticipated). Based on results from these conditioned hearing sensitivity experiments, odontocetes participating in some previous TTS experiments could have been protecting their hearing during exposures (Finneran, 2018; Finneran et al., 2024; Finneran et al., 2023). A better understanding of the mechanisms responsible for the observed hearing changes is needed for proper interpretation of some existing TTS data, particularly for TTS due to short-duration, unpredictable exposures.

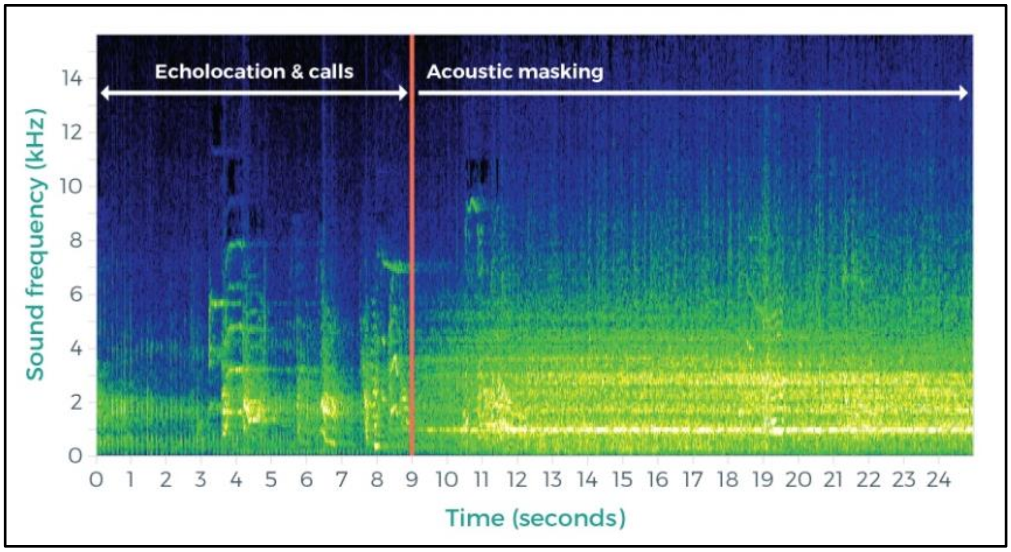
D.6.4 MASKING

This section provides an overview of masking in marine mammals, discusses the potential impacts of masking including communication space reduction and vocalization changes in response to noise, and reviews scientific literature specific to masking by anthropogenic sources. Detailed reviews and analysis of masking in marine mammals are provided by Clark et al. (2009), Erbe et al. (2016), and Branstetter and Sills (2022).

Most research on auditory masking measures the ability of the listener to detect a signal in noise. This is also called “energetic” masking. Energetic masking has been measured for pinnipeds (Sills et al., 2014, 2015; Southall et al., 2000, 2003), odontocetes (Au & Moore, 1990; Branstetter et al., 2021; Branstetter et al., 2017; Johnson et al., 1989; Kastelein & Wensveen, 2008; Lemonds et al., 2011; Thomas et al., 1990a), sirenians (Gaspard et al., 2012), and sea otters (Ghoul & Reichmuth, 2014b). These measurements allow predictions of masking if the spectral density of noise is known (Branstetter et al., 2017). Although energetic masking is typically estimated in controlled laboratory conditions using white noise, results can vary considerably depending on the noise type (Branstetter et al., 2013; Trickey et al., 2010). These fundamental measurements of the ability of marine mammals to detect different signal types under different masking noise conditions are useful for prediction of masking in real-world scenarios.

The frequency overlap between the signal and masker is perhaps the most important consideration when assessing the potential effect of noise. For example, higher frequency noise is more effective at masking higher frequency signals, (Au & Moore, 1990; Lemonds et al., 2011). Signal type (e.g., whistles, burst-pulse, echolocation clicks) and spectral characteristics (e.g., frequency modulation and/or harmonics) may further influence masked detection thresholds (Branstetter et al., 2016; Branstetter & Finneran, 2008; Branstetter et al., 2013; Cunningham et al., 2014). Figure D.6-3 shows an example of lower-frequency ship noise masking communication calls.

Much emphasis has been placed on signal detection in noise and, as a result, most masking studies and models have used masked signal detection thresholds. However, from a fitness perspective, signal detection does not equate to the ability to determine the sound source location and recognize “what” is producing the sound. Marine mammals use sound to recognize conspecifics, prey, predators, or other biologically significant sources. Masked recognition thresholds for whistle-like sounds, have been measured for bottlenose dolphins (Branstetter et al., 2016) and are approximately 4 dB above detection thresholds (signal detection masking) for the same signals. It should be noted that the term “threshold” typically refers to the listener’s ability to detect or recognize a signal 50 percent of the time. For example, human speech communication, where only 50 percent of the words are recognized, would result in poor communication. Likewise, recognition of a conspecific call or the acoustic signature of a predator at only the 50 percent level could have severe impacts (Branstetter et al., 2016). Masking that may not result in a loss of signal detection, but results in loss of a signal’s meaning is called informational masking.



Notes: Spectrogram showing killer whale communication calls and echolocation sounds in the first nine seconds, which are then masked by the passing of a ship. The ship’s masking noise is predominant at 1.5 kHz and extends up to about 6 kHz. Some communication calls can be seen at 11 and 19 seconds. Echolocation calls (small vertical stripes) extend to much higher frequencies and are not masked as much as communication calls in this example. Figure from Kathy Heise and Tracy Saxby, Coastal Ocean Research Institute, https://oceanwatch.ca/bccoast/wp-content/uploads/sites/4/2018/10/OceanWatch-BC-Coast-underwater-noise.pdf

Figure D.6-3: Masking of Killer Whale Calls by a Passing Ship

Marine mammals use sound to recognize predators (Allen et al., 2014; Cummings & Thompson, 1971; Curé et al., 2015; Fish & Vania, 1971). Auditory recognition may be reduced in the presence of a masking noise, particularly if it occurs in the same frequency band. Therefore, the occurrence of masking may prevent marine mammals from responding appropriately to the acoustic cues produced by their predators. For example, studies have shown that for marine mammals that are preyed upon by killer whales, some recognition of predator cues might be missed if the killer whale vocalizations were masked (Curé et al., 2016; Curé et al., 2015; Deecke et al., 2002; Isojunno et al., 2016; Visser et al., 2016). This possibility depends on the duration of the masking and the likelihood of encountering a predator during the time that detection and recognition of predator cues are impeded. Relatively little data exists on informational masking in marine mammals despite its potential importance in models of how noise affects communication.

D.6.4.1 Masking Concepts

D.6.4.1.1 Release from Masking

Masking is less likely or is expected to be less impactful when the noise is intermittent, such as low-duty cycle sonars or impulsive noise, compared to when the noise is continuous, such as vessel noise, high-duty cycle sonar, or continuous active sonar. This is because for intermittent noise, the signal of interest can be detected during the quiet periods between noise events. This is often called “dip” or “gap” listening. The effect of masking on communication space is often modeled using constant-amplitude noise, whereas many anthropogenic sources contain gaps or fluctuations in the noise. Studies have shown that the signal duration, duty cycle, masker level, and fluctuations should be considered when modeling the effect of noise on signal detection (Branstetter & Finneran, 2008; Branstetter et al., 2013; Kastelein et al., 2021; Sills et al., 2017; Trickey et al., 2010).

Spatial release from masking (SRM) occurs when a noise and signal are separated in space, resulting in a reduction or elimination of masking (Holt & Schusterman, 2007; Popov et al., 2020). The relative position of sound sources can act as one of the most salient cues that allow the listener to segregate multiple sounds in a complex auditory scene. Many sounds are emitted from a directional source that is spatially separated from biologically relevant signals. Under such conditions, minimal masking will occur, and existing models of masking will overestimate the amount of actual masking. Marine mammals have excellent sound source localization capabilities (Branstetter & Mercado, 2006; Byl et al., 2019; Renaud & Popper, 1975) and directional hearing (Accomando et al., 2020; Au & Moore, 1984; Mooney et al., 2008; Popov & Supin, 2009) which likely combine to aid in separating auditory events and improving detection. Spatial release from masking has been empirically demonstrated using behavioral methods in a harbor seal a California sea lion, three harbor porpoises, and a bottlenose dolphin (Holt & Schusterman, 2007; Kastelein et al., 2021; Popov et al., 2020), where maximal spatial release from masking was 19, 12, 14.5, and 24 dB for each species respectively. The spatial positions of the receiver and noise source are often considered in terms of distance but the relative angles between the vocalizing and/or listening animal and the noise source are also important to consider when estimating masking effects.

D.6.4.1.2 Communication Space Models of Masking

Communication space models estimate how the distance at which animals can communicate is reduced in noise. The term “communication space” typically means the distance an animal’s call can travel and feasibly be heard and interpreted by a listener. Since the range of available communication space varies widely with species and habitat, reduction in communication space is usually quantified as a percentage loss or a percentage of space available during increased anthropogenic and ambient noise.

Models typically include the source level and frequency characteristics of both the animal of interest’s vocalization and the noise, and the spatial relationship between the noise source and the calling animal and/or the listener. The listener (i.e., receiver) is considered in the best available communication space models, which use the listener’s hearing characteristics when data are available. Models vary in their implementation of propagation modeling – or how the sound (signal and noise) levels are reduced with distance. Some use simple spherical spreading loss while others employ more sophisticated location-specific estimates, and these choices are related both to the specific research question and the availability of empirical data or existing propagation models.

Clark et al. (2009) estimated masking effects on communication signals for three species of calling mysticete whales (LF cetaceans), including calculating the cumulative impact of multiple noise sources. For example, the model estimates that a right whale's optimal communication space (around 20 km) is decreased by as much as 84 percent when two commercial ships pass by. When one ship passed, communication space for singing fin and humpback whales briefly decreased by approximately 20 and 8 percent respectively when the ship passed close to the whales. For the same ship passage, right whale communication space was reduced by approximately 77 percent. These differences were due to the call repetition rate, source level, and call frequency differences between species. Notably, the right whale calls had a much lower repetition rate in comparison to humpback and fin whale calls. In another study, Hatch et al. (2012) found that North Atlantic right whale communication space was reduced by 67 percent during exposure to vessel noise.

An experiment in a shallow water environment (less than 50 m depth) investigating humpback whale sounds (vocalizations and surface-generated sounds) determined that, in typical ambient (wind) noise, their communication range extends to approximately 2 - 4 km (Dunlop, 2018). Considering this baseline space restricted by ambient noise, Dunlop (2019) used vocalization and whale interactions to show a reduction in humpback whale communication space in vessel noise. This study concluded that the physical presence of the vessel could possibly explain changes in social behavior. This example illustrates the overall concept that changes in behavior observed in the field, including vocalization, often cannot be ascribed solely to masking noise, but also to the physical presence of the noise source.

Results from additional scientific studies on communication space, primarily from vessel noise are listed in Table D.6-3.

Table D.6-3: Communication Space Models of Masking in Marine Mammals

Species	***Location***	***Anthropogenic Noise Source***	***Communication Space Reduction***	***Call Type***	***Study***
North Atlantic right whale (*Eubalaena glacialis*)	Stellwagen Bank National Sanctuary, USA	Passing vessels	77% (single vessel) 84% (two vessels)	71 – 224 Hz contact call	Clark et al. (2009)
		AIS[1], fishing, and whale-watching vessels	5%	36 – 891 Hz “gunshot” call	Cholewiak et al. (2018)
Humpback whale (*Megaptera novaeangliae*)	Stellwagen Bank National Sanctuary, USA	Single vessel passing	8%	224 – 708 Hz song	Clark et al. (2009)
		AIS, fishing, and whale-watching vessels	80 – 99%[2]	36 – 355 Hz song and social sounds	Cholewiak et al. (2018)
	Glacier Bay National Park, USA	AIS vessel traffic, summer season	13 – 28% (song) 18 – 51% (calls)	224 – 708 Hz song, 50 – 700 Hz “whup” calls	Gabriele et al. (2018)[2]
	Peregian Beach, Australia	Vessel-dominated noise	25 – 50%	Low-frequency calls (≤ 126 Hz min., ≤ 159 Hz center frequency) and	Dunlop (2019)

Table D.6-3: Communication Space Models of Masking in Marine Mammals (continued)

Species	*Location*	*Anthropogenic Noise Source*	*Communication Space Reduction*	*Call Type*	*Study*
				high-frequency calls (> 159 Hz center frequency)	
	Colombian Pacific	Vessel (whale-watching/ecotour)	63%	350 Hz peak frequency	Rey-Baquero et al. (2021)
Fin whale (*Balaenoptera physalus*)	Stellwagen Bank National Sanctuary, USA	Single vessel passing	20%	18 -28 Hz song	Clark et al. (2009)
		AIS[1], fishing, and whale-watching vessels	80 – 99%[2]	18 – 22 Hz song	Cholewiak et al. (2018)
Bryde's whale (*Balaenoptera edeni*)	Hauraki Gulf, New Zealand	AIS[1] vessels	≤ 87%	23.5 – 207.8 Hz calls	Putland et al. (2018)
Minke whale (*Balaenoptera acutorostrata*)	Stellwagen Bank National Sanctuary, USA	AIS[1], fishing, and whale-watching vessels	≥ 80%	56 – 355 Hz pulse trains	Cholewiak et al. (2018)
Killer whale (*Orcinus orca*)	Haro Strait, USA	Vessels	62 – 97%	1.5 – 3.5 kHz	Williams et al. (2014a)
	Salish sea	Vessels and wind	50 – 90%	1 – 50 kHz	Burnham et al. (2023)
Beluga whale (*Delphinapterus leucas*)	Saguenay - St. Lawrence Marine Park, Canada	Car ferries, whale watching vessels, small vessels	70 – 85%	2.5 kHz center frequency	Gervaise et al. (2012)
	St. Lawrence Estuary, Canada	Vessels	53 – 57%	Adult, sub-adult, and calf communication calls	Vergara et al. (2021)
Bottlenose dolphin (*Tursiops* sp.)	Tenerife, Canary Islands, Spain	Vessels	26%	4 – 10 kHz whistles	Jensen et al. (2009)
Short-finned pilot whale (*Globicephala macrorhynchus*)			58%	2 – 12.5 kHz tonal sounds	
Harbor seal (*Phoca vitulina*)	Glacier Bay National Park, USA	AIS[1] vessel traffic, summer season	32 – 61%	4 – 500 Hz "roar"	Gabriele et al. (2018)

[1] AIS = Automatic Identification System, certain types of vessels are outfitted with transponders that provide position information.

[2] This communication space reduction value is based on increase in anthropogenic noise and ambient (natural) background noise increases combined

Notes: % = percent; < = less than; > = greater than; ≤ = less than or equal to; ≥ = greater than or equal to; AIS = Automatic Identification System; Hz = Hertz; kHz = kilohertz; sp. = species;

These studies demonstrate that anthropogenic sounds – especially broadband vessel noise – can reduce the communication space available to marine mammals. Existing models tend to simplify the noise characteristics such as how the sound propagates away from the noise source, and the auditory capabilities of the listener (e.g., do not consider directional hearing). Additionally, as pointed out by Branstetter and Sills (2022), many of these models are based on an assumed signal detection and recognition threshold – usually a 10 dB signal-to-noise ratio (Clark et al., 2009).

D.6.4.1.3 Noise-Induced Vocal Modifications

Masking noise can result in vocal modifications or other acoustic signaling behaviors that might reduce or compensate for the overall effect of masking. These noise-induced vocal modifications (NIVM) include increasing the source level (Lombard effect), modifying the frequency, increasing the repetition rate of vocalizations, or ceasing to vocalize in the presence of increased noise (Hotchkin & Parks, 2013). With increased natural background (ambient) noise levels, a switch from vocal communication to physical, surface-generated sounds such as pectoral fin slapping or breaching was has been observed in mysticete whales (Dunlop et al., 2010).

Vocalization changes have been reported from exposure to anthropogenic noise sources such as sonar, vessel noise, and seismic surveying (Gordon et al., 2003; Holt et al., 2011; Holt et al., 2008; Lesage et al., 1999; McDonald et al., 2009; Nowacek et al., 2007; Rolland et al., 2012) as well as changes in the natural acoustic environment (Brumm & Slabbekoorn, 2005). It is often difficult to discriminate NIVM from potential effects of context, measurement tools, and analysis methods. For example, vocalizations may be masked from the recorder, or confounded by other behavioral responses of the marine mammal such as moving away from the noise and recorder or increasing dive duration (Castellote et al., 2012; Cerchio et al., 2014). The ability to observe NIVM might also depend on the methods used to quantify baseline behavior and timescale over which recordings are analyzed (Casey et al., 2024). Table D.6-4 details some examples of the best available scientific observations of noise-induced vocal modifications in marine mammals due to anthropogenic and ambient noise.

Table D.6-4: Examples of Noise-Induced Vocal Modifications in Marine Mammals

Species	***Study***	***Noise source***	***Vocalization Change***			
			Rate	***Duration***	***Frequency***	***Amplitude***
Blue whale (*Balaenoptera musculus*)	Di Lorio and Clark (2010)	Seismic survey (sparker pulses, average received SELs of 131 dB re 1 $\mu Pa^2 s$)	↑			
	Shabangu et al. (2022)	Vessel (10 – 500 Hz)	↑			
		Wind (1 – 4 kHz)	↓[2] masked			
North Atlantic right whale (*Eubalaena glacialis*)	Parks et al. (2011); Parks et al. (2009).	Ambient (20 Hz – 8 kHz)	↓		↑[1]	↑ Lombard
Humpback whale (*Megaptera novaeangliae*)	Girola et al. (2023)	Wind				↑ Lombard, 0.5 dB for every 1 dB increase
		Vessels				NC

Table D.6-4: Examples of Noise-Induced Vocal Modifications in Marine Mammals (continued)

Species	***Study***	***Noise source***	***Vocalization Change***			
			Rate	***Duration***	***Frequency***	***Amplitude***
	Shabangu et al. (2022)	Wind				↑ Lombard
	Laute et al. (2022)	Vessels	↓			
	Dunlop et al. (2014)	Ambient wind noise				↑ Lombard, 0.9 dB for every 1 dB noise increase
	Dunlop (2016)	Vessels		NC	NC	↓ Masked[2]
	Fournet et al. (2018)	Vessels and ambient	↓ 9% for every 1 dB noise increase			↑ Lombard, 0.8 dB for every 1 dB noise increase
	Fristrup et al. (2003); Miller et al. (2000),	Low-frequency active sonar		↑ overall song length		
Bowhead whale (*Balaena mysticetus*)	Blackwell et al. (2015); Blackwell et al. (2017)	Seismic survey (air gun pulses) and large-scale drilling operation (tonal drilling, vessels)	↑ Noise levels < 127 dB ↓ Noise levels > 127 dB X Noise level 170 dB			
Beluga (*Delphinapterus leucas*)	Lesage et al. (1999)	Small vessels	↓ overall ↑ certain call types		↑ bandwidth	
Beluga (*Delphinapterus leucas,* St. Lawrence Estuary)	Scheifele et al. (2005)	Vessels				↑ Lombard
Killer whale (*Orcinus orca*)	Foote et al. (2004)	Vessels	NC	↑		
Killer whale	Wieland et al. (2010)	Vessels		↑14 call types		

Table D.6-4: Examples of Noise-Induced Vocal Modifications in Marine Mammals (continued)

Species	*Study*	*Noise source*	*Vocalization Change*			
			Rate	*Duration*	*Frequency*	*Amplitude*
(*Orcinus orca,* Southern Resident)				↓ 2 call types		
	Holt et al. (2011); (2008)	Vessels				↑ Lombard
Bottlenose dolphin (*Tursiops sp.*)	Buckstaff (2004)	Vessels	↑ Vessel approach	NC	NC	
	Luís et al. (2014)	Vessels	↓			
	Gospić and Picciulin (2016)	Vessels (low-frequency noise)			↑	
	Antichi et al. (2022)	Vessels (single small vessel passages)			Coastal dolphins ↑, Oceanic dolphins ↓ (after approach)	
Delphinids (multiple species)	Papale et al. (2015)	Anthropogenic and ambient noise			↑ min/max frequency	
Dugong (*Dugong dugon*)	Ando-Mizobata et al. (2014)	Vessels (within 400 m)	NC	↑	↑ bandwidth	
Harbor seal (*Phoca vitulina*, pups, 1 – 3 weeks old)	Torres Borda et al. (2021)	Broadband recorded ambient noise playback	NC	NC	↓ fundamental frequency	↑ Lombard in three of eight seals
Bearded seal (*Erignathus barbatus*)	Fournet et al. (2021)	Ambient (below 900 Hz)				↑ Lombard

[1] Call frequency and Lombard effect are often interrelated.

[2] In many studies, decreases in call amplitude or detections (calling rates) can result from masking of the recording hydrophone (receiver) rather than a change in the animal's vocal behavior.

Notes: ↑ = increase; ↓ = decrease; % = percent; < = less than; > = greater than; $\mu Pa^2 s$ = micropascal squared seconds; dB = decibel; Hz = Hertz; kHz = kilohertz; NC = no change; SEL = sound exposure level; X = ceased calling

In some scenarios, depending on the capability of the individual animal to adjust the frequency and/or source levels of their calls and the characteristics of anthropogenic noise, vocal modifications might not compensate for masking. For example, Fournet et al. (2021) showed that estimated source levels of seal calls increased with ambient noise up to approximately 100 to 105 dB rms, above which no further Lombard effect was observed. This suggests that masking of bearded seal mating calls may occur in the presence of noise that exceeds 100 dB.

Vocal and other behavioral changes in response to masking noise might have fitness consequences, such as those that could result from an increase in metabolic rates and oxygen consumption, as was found for bottlenose dolphins when increasing their call amplitude (Holt et al., 2015). Some species might avoid changing the source levels or frequencies of their vocalizations to avoid predation or suffer increased risks of predation due to these vocal modifications. For example, beaked whales that modify their vocalizations might compromise otherwise cryptic foraging strategies which function to avoid predation by killer whales (Aguilar de Soto et al., 2006; Brumm & Slabbekoorn, 2005).

D.6.4.2 Masking by Anthropogenic Noise Sources

This section summarizes the predicted effects of masking by each type of anthropogenic noise source on marine mammals based on the information presented above. Examples of studies specific to vessel noise, sonar, and impulsive sounds are also discussed. The most important considerations for potential masking are the source level, frequency, duty cycle, and range (distance between masker and listening or calling animal).

D.6.4.2.1 Masking by Vessel Noise

Masking of marine mammal vocalizations is most likely to occur in the presence of broadband, relatively continuous noise sources such as vessels. This type of noise overlaps in frequency with many marine mammal sounds and can effectively reduce their communication space. Both signal detection and informational masking are likely to occur in the presence of vessel noise (Erbe et al., 2016). Models of communication space reduction (Table D.6-3) have predicted substantial decreases in communication space for a variety of species. When there is persistent vessel noise such as in a busy harbor, this effect is likely to be pervasive in nearby habitats as compared to intermittent when vessels pass through a habitat with lower ambient noise levels.

It is also possible that high source level vessel noise could mask marine mammal echolocation sounds. Hermannsen et al. (2014) estimated that broadband vessel noise could extend up to 160 kHz at ranges from 60 to 1,200 m, and that the higher frequency portion of that noise might mask harbor porpoise clicks. However, masking might not occur in practice, since harbor porpoises may avoid vessels and therefore may not be close enough to have their clicks masked (Dyndo et al., 2015; Polacheck & Thorpe, 1990; Sairanen, 2014). Liu et al. (2017) found that broadband shipping noise could cause masking of humpback dolphin whistles within 1.5 to 3 km, and masking of echolocation clicks within 0.5 to 1.5 km. Williams et al. (2014a) found that killer whale echolocation clicks (18 to 60 kHz) in Haro Strait were not masked by vessel noise over a 2 km distance. Gervaise et al. (2012) showed that the echolocation frequency range of belugas in the Saguenay-St. Lawrence Marine Park was masked by car ferry noise.

Overall, vessel noise has a substantial probability of masking marine mammal communication sounds and can also mask echolocation sounds in some cases. However, many studies of vessel noise masking do not consider spatial release from masking [e.g., (Brewer et al., 2023)], which is likely to reduce the effect of masking if the vessel is spatially separated from the signal of interest; this is especially relevant for situations where avoidance behavior is also exhibited. The overall potential effects of masking by vessel noise are (1) a reduction in the ability of marine mammals to communicate, detect, or interpret biologically relevant sounds, (2) costs associated with noise-induced vocal modifications such as the Lombard effect, or (3) costs associated with other behavioral responses to masking noise or the physical presence of vessels (see Behavioral Reactions D.6.5).

D.6.4.2.2 Masking by Sonar

Because military sonars typically have low duty cycles, relatively short duration, and narrow bandwidth that does not overlap with vocalizations for most marine mammal species, masking would be limited as compared to continuous sources (e.g., vessel noise). Dolphin whistles and mid-frequency active sonar are similar in frequency, so masking is possible but less likely due to the low-duty cycle and short durations of most sonars and the probability that dip listening would occur. For similar reasons, masking caused by low-frequency active sonar may be limited where it overlaps in frequency with some mysticete vocalizations (e.g., minke and humpback whales) (Fristrup et al., 2003; Miller et al., 2000).

High-duty cycle or continuous active sonars have the potential to mask marine mammal vocalizations. These sonars transmit more frequently than intermittent sonars, but at substantially lower source levels.

While the lower source levels limit the range of impact compared to other systems, animals close to the sonar source are likely to experience masking on a much longer time scale than those exposed to intermittent sonars. Continuous noise at the same frequency of communicative vocalizations may cause disruptions to communication, social interactions, and acoustically mediated cooperative behaviors (Sørensen et al., 2023) such as foraging and mating. Similarly, because the high-duty cycle or continuous active sonar are mid-frequency, there is the potential for the sonar signals to mask important environmental cues like predator vocalizations (e.g., killer whales), possibly affecting prey (including other marine mammals). Spatial release from masking may occur with higher duty cycle or continuous active sonars.

von Benda-Beckmann et al. (2021) modeled the effect of pulsed and continuous 1-2 kHz active sonar on sperm whale echolocation clicks and found that the presence of upper harmonics in the sonar signal increased masking of clicks produced in the search phase of foraging compared to buzz clicks produced during prey capture. Different levels of sonar caused intermittent to continuous masking (120 to 160 dB re 1 µPa^2, respectively), but varied based on click level, whale orientation, and prey target strength. Continuous active sonar resulted in a greater percentage of time that echolocation clicks were masked compared to pulsed active sonar. This means that sonar sounds could reduce the ability of sperm whales to find prey under certain conditions. However, echoes from prey are most likely spatially separated from the sonar source, and so spatial release from masking would be expected.

Overall, sonar has the potential to mask marine mammal communication sounds and echolocation clicks. Continuous active sonar is more likely to mask vocalizations than intermittent sonar, and in general, sonar is less likely than vessel noise to have masking effects on sounds that are biologically relevant to marine mammals.

D.6.4.2.3 Masking by Impulsive Sound Sources

Impulsive sound sources, including explosions, are intense and short in duration (see D.1.1.5). Since impulsive noise is intermittent, the length of the gap between sounds (duty-cycle) and received level are pertinent when considering the potential for masking. Impulsive sounds with lower duty cycles or lower received levels are less likely to result in masking than higher duty cycles or received levels. There are no direct observations of masking in marine mammals due to exposure to explosive sources. Potential masking from explosive sounds or weapon noise is likely similar to masking studied for other impulsive sounds, such as air guns or pile-driving.

Masking of mysticete calls could occur due to the overlap between their low-frequency vocalizations and the dominant frequencies of impulsive sources (Castellote et al., 2012; Nieukirk et al., 2012). For example, blue whale feeding/social calls increased when seismic exploration was underway (Di Lorio & Clark, 2010), indicative of a possible compensatory response to masking effects of the increased noise level. However, mysticetes that call at higher rates are less likely to be masked by impulsive noise with lower duty cycles (Clark et al., 2009) because of the decreased likelihood that the noise would overlap with the calls, and because of dip listening. Field observations of masking effects such as vocal modifications are difficult to interpret because when recordings indicate that call rates decline, this could be caused by (1) animals calling less frequently (actual noise-induced vocal modifications), (2) the calls being masked from the recording hydrophone due to the noise (e.g., animals are not calling less frequently but are being detected less frequently), or (3) the animals moving away from the noise, or any combination of these causes (Blackwell et al., 2013; Cerchio et al., 2014).

Masking of pinniped communication sounds at 100 Hz center frequency is possible when vocalizations occur at the same time as an air gun pulse (Sills et al., 2017). This might result in some percentage of vocalizations being masked if an activity such as a seismic survey is being conducted in the vicinity, even when the sender and receiver are near one another. Release from masking due to "dip listening" is likely in this scenario.

While a masking effect of impulsive noise can depend on the received level (Blackwell et al., 2015) and other characteristics of the noise, the vocal response of the affected animal to masking noise is an equally important consideration for inferring overall impacts to an animal. As illustrated in Table D.6-4, it is possible that the receiver would increase the rate and/or level of calls to compensate for masking; or, conversely, cease calling.

In general, impulsive noise has the potential to mask sounds that are biologically important for marine mammals, reducing communication space or resulting in noise-induced vocal modifications that might impact marine mammals. Masking by close-range impulsive sound sources is most likely to impact marine mammal communication.

D.6.5 BEHAVIORAL REACTIONS

Any stimulus in the environment can cause marine mammals to react, including noise from anthropogenic sources such as vessels, sonar, or aircraft, as well as the physical presence of a vessel or aircraft. Marine mammal responses to anthropogenic sound were reviewed by Richardson et al. (1995b). Other reviews (Nowacek et al., 2007; Southall et al., 2007) addressed studies conducted since 1995 and focused on observations where the received sound level was known or could be estimated, and discussed the role of context. Southall et al. (2007) synthesized data from many past behavioral studies and observations to determine the likelihood of behavioral reactions at specific sound levels, and Southall et al. (2016) reviewed the range of experimental field studies that have been conducted to measure behavioral responses of cetaceans to sonar.

Considerable variability has been observed in marine mammal responses to sound. Methods have been developed and refined to categorize and assess the severity of acute responses, considering impacts to individuals that may consequently impact populations (Southall et al., 2007; Southall et al., 2021). These severity scales assess immediate discrete responses in relation to behaviors affecting vital rates, including survival, reproduction, and foraging. Using these scales, a behavioral response by a wild (non-captive) marine mammal may range from low severity (e.g., detectable interruptions in foraging, diving, or courtship behavior) to moderate severity (e.g., avoidance, sustained foraging reduction) to high severity (e.g., separation of mother-offspring, prolonged displacement from foraging habitat, repeated breeding disruption leading to reduced reproductive success). Captive animal behavior studies allow for controlled, repeated exposures with very precise measures, but captive marine mammals may have training and motivational contexts that make their responses difficult to compare to free-ranging, non-captive animals (Southall et al., 2021). Therefore, behavioral severity scales developed for captive marine mammals consider other factors such as trained behaviors, use of rewards, and habituation.

While in general, the louder the sound source, the more intense the behavioral response, it was clear that the proximity of a sound source and the animal's experience, motivation, and conditioning were also critical factors influencing the response (Southall et al., 2007; Southall et al., 2016). Ellison et al. (2011) submit that "exposure context" greatly influences the type of behavioral response exhibited by an animal and outlined an approach to assessing the effects of sound on marine mammals that considers not just the received level of sound, but also in what activity the animal is engaged, the nature and novelty of the sound (i.e., is this a new sound from the animal's perspective), and the distance between the sound source and the animal. Other variables and contextual factors that may affect the probability and magnitude of a behavioral response include subject-specific factors (e.g., age, sex, presence of a calf, and group size and composition), characteristics of the sound (frequency, duration, similarity to predator sounds, and whether it is continuous or intermittent); whether the sound is approaching or moving away; the presence of predators, prey, or conspecifics; and navigational constraints on the animal (Ellison et al., 2011; Southall et al., 2021; Wartzok et al., 2003).

Extensive research programs have and are investigating the responses of free-ranging marine mammals to anthropogenic sounds, including actual and simulated tactical sonars both on and off military ranges (Southall et al., 2016). These behavioral response studies include controlled exposure studies, in which detailed acoustic dose and behavioral data are obtained from tags on focal animals, as well as data obtained from longer-term tags and passive acoustic monitoring during opportunistic exposures to actual sonar on naval training and testing ocean ranges with bottom-mounted hydrophones (Harris et al., 2018). Table D.6-5 provides an overview of current and past efforts as background to the findings presented in the taxa-specific sections below.

Table D.6-5: Major Non-Captive Behavioral Response Studies

Project/ Location	*Focal Species*	*Sound source*	*Studies*
Opportunistic Studies on Navy instrumented Ranges			
AUTEC, Bahamas	Blainville's beaked whale	Navy hull-mounted sonar	Joyce et al. (2019); McCarthy et al. (2011); Moretti et al. (2014); Tyack et al. (2011)
SOCAL Anti-Submarine Warfare Range	Goose-beaked whale[1]	Navy hull-mounted and dipping sonar	Falcone et al. (2017)
Pacific Missile Range Facility, Hawaii	Minke whale Humpback whale Blainville's beaked whale	Navy hull-mounted sonar	Durbach et al. (2021); Harris et al. (2019b); Henderson et al. (2019); Henderson et al. (2016); Manzano-Roth et al. (2016); Martin et al. (2015)
BRS with Controlled Exposure Experiments			
AUTEC-BRS (Bahamas)	Blainville's beaked whales	Simulated tactical sonar	Tyack et al. (2011)
3S1[2] (Norway)	Killer whale Long-finned pilot whale Sperm whale	Simulated tactical sonar (1 – 2 kHz or 6 – 7 kHz, intermittent)	Antunes et al. (2014); Curé et al. (2016); Isojunno et al. (2016); Isojunno et al. (2017); Miller (2012); Miller et al. (2014); Sivle et al. (2012b); Visser et al. (2016)
3S2[2] (Norway)	Humpback whale Minke whale Northern bottlenose whale	Simulated tactical sonar (1 – 2 kHz or 6 – 7 kHz, intermittent)	Curé et al. (2021); Kvadsheim et al. (2017); Miller et al. (2015); Sivle et al. (2015b); Sivle et al. (2016b); Wensveen et al. (2019); Wensveen et al. (2017)
3S3[2] (Norway)	Sperm whale Long-finned pilot whale	Simulated tactical sonar (1 – 2 kHz continuous and intermittent)	Curé et al. (2021); Isojunno et al. (2021); Isojunno et al. (2020)
SOCAL BRS	Blue whale Fin whale Minke whale Baird's beaked whale Goose-beaked whale[1] Risso's dolphin	Simulated tactical sonar (3.5 – 4 kHz intermittent)	DeRuiter et al. (2013b); Friedlaender et al. (2016); Goldbogen et al. (2013); Kvadsheim et al. (2017); Southall et al. (2019b); Stimpert et al. (2014); Southall et al. (2023)

Table D.6-5: Major Non-Captive Behavioral Response Studies (continued)

Project/ Location	*Focal Species*	*Sound source*	*Studies*
Atlantic BRS[3]	Goose-beaked whale[1] Short-finned pilot whale	Simulated tactical sonar (3 – 4 kHz, intermittent) and Navy hull-mounted sonar	

[1] Formerly Cuvier's beaked whale

[2] 3S = Sea mammals and Sonar Safety

[3] This is the most recent BRS efforts; thus, peer-reviewed publications of findings are not yet available.

Notes: AUTEC = Atlantic Undersea Test and Evaluation Center; BRS = Behavioral Response Studies; kHz = kilohertz; SOCAL = Southern California

For most species, little or no data exist on behavioral responses to any sound source. For the below synthesis of best available science on marine mammal behavioral responses, all species have been grouped into broad taxonomic groups from which general response information can be inferred.

D.6.5.1 Behavioral Reactions of Mysticetes

D.6.5.1.1 Sonar and Other Transducers

The responses of mysticetes to sonar and other duty-cycled tonal sounds depend on the characteristics of the signal, behavioral state of the animal, sensitivity and previous experience of an individual, and other contextual factors including distance of the source, movement of the source, physical presence of vessels, time of year, and geographic location (Goldbogen et al., 2013; Harris et al., 2019a; Harris et al., 2015; Martin et al., 2015; Sivle et al., 2015b). Behavioral response studies have been conducted over a variety of contextual and behavioral states, helping to identify which factors, beyond the received level of the sound, may lead to a response. Observed reactions during behavioral response studies have not been consistent across individuals based on received sound levels alone, and likely were the result of complex interactions between these contextual factors.

In the SOCAL BRS, tagged surface feeding blue whales did not show a change in behavior in response to mid-frequency simulated and incidental Navy sonar sources with received levels between 90 and 179 dB re 1 µPa, but deep feeding and non-feeding whales showed temporary reactions including cessation of feeding, reduced initiation of deep foraging dives, generalized avoidance responses, and changes to dive behavior. The behavioral responses were generally brief, of low to moderate severity, and highly dependent on exposure context (behavioral state, source-to-whale horizontal range, and prey availability), with a return to baseline behavior shortly after the end of the exposure (DeRuiter et al., 2017; Goldbogen et al., 2013; Southall et al., 2019c). When the prey field (krill) was mapped and used as a covariate in models looking for a response in the 2011–2013 SOCAL BRS data set, the response in deep-feeding blue whales was even more apparent, reinforcing the need for contextual variables, such as feeding state, to be included when assessing behavioral responses (Friedlaender et al., 2016). The probability of a moderate behavioral response increased when the range to source was closer for these foraging blue whales, although there was a high degree of uncertainty in that relationship (Southall et al., 2019b). None of the tagged fin whales in the SOCAL BRS demonstrated more than a brief or minor response regardless of their behavioral state (Harris et al., 2019a). The fin whales were exposed to both mid-frequency simulated sonar and pseudorandom noise of similar frequency, duration, and source level. They were less sensitive to disturbance than blue whales, with no significant differences in response between behavioral states or signal types. The authors rated responses as low-to-moderate severity with no negative impact to foraging success (Southall et al., 2023).

Similarly, humpback whale behavioral responses to sonar have been influenced by foraging state. During sonar exposure of tagged whales in the 3S2 study, the rates of foraging lunges generally decreased, but responses varied across individuals (e.g., ceasing or starting to forage); most of the non-foraging humpback whales did not respond to any approaches at all (Sivle et al., 2016b). For foraging whales, lunges decreased (although not significantly) during a no-sonar control vessel approach prior to the sonar exposure, and lunges decreased less during a second sonar approach than during the initial approach. There was also variability in humpback avoidance responses. Some tagged whales in the 3S2 study avoided the sonar vessel only during the first or second exposure, and only one whale avoided both (Sivle et al., 2016b). This suggests that there may have been responses to the vessel or habituation to the sonar and vessel after repeated approaches. Almost half of the animals with avoidance responses were foraging before the exposure; the non-feeding whales that avoided responded at a slightly lower received level and greater distance than those that were feeding (Wensveen et al., 2017). When responses did occur the animals quickly returned to their previous behavior after the sound exposure ended (Sivle et al., 2015b). Changes in foraging duration during mammal-eating killer whale playbacks and mid-frequency sonar were positively correlated across multiple species in the 3S Norwegian studies, including humpback whales, suggesting that tolerance of predation risk may play a role in sensitivity to sonar disturbance (Miller et al., 2022), with the humpback whales responding more severely to the killer whale vocalization playbacks than they did to the sonar playbacks (Curé et al., 2015; Sivle et al., 2015b).

The most severe baleen whale response in any behavioral response study was observed in a minke whale in the 3S2 study, which responded to simulated naval sonar at a received level of 146 dB re 1 µPa by strongly avoiding the sound source (Kvadsheim et al., 2017; Sivle et al., 2015b). Although the minke whale increased its swim speed, directional movement, and respiration rate, none of these were greater than rates observed in baseline behavior, and its dive behavior remained similar to baseline dives. A minke whale tagged in the SOCAL behavioral response study also responded by increasing its directional movement, but maintained its speed and dive patterns, and so did not demonstrate as strong of a response (Kvadsheim et al., 2017). In addition, the 3S2 minke whale demonstrated some of the same avoidance behavior during the controlled ship approach with no sonar, indicating at least some of the response was to the vessel (Kvadsheim et al., 2017).

In addition to behavioral response studies, responses by humpback and minke whales to actual training activities on Navy ranges have been monitored. Several humpback whales have been observed during aerial or visual surveys during Navy training events involving sonar; no avoidance or other behavioral responses were ever noted, even when the whales were observed within 5 km of a vessel with active sonar and maximum received levels were estimated to be between 135 and 161 dB re 1 µPa (Mobley, 2011; Mobley & Milette, 2010; Mobley & Pacini, 2012; Mobley et al., 2012; Smultea et al., 2009). In fact, one group of humpback whales approached a vessel with active sonar so closely that the sonar was shut down and the vessel slowed; the animals continued approaching and swam under the bow of the vessel (U.S. Department of the Navy, 2011a). Another group of humpback whales continued heading towards a vessel with active sonar as the vessel was moving away for almost 30 minutes, with an estimated median received level of 143 dB re 1 µPa. This group was observed producing surface active behaviors such as pec slaps, tail slaps, and breaches; however, these are very common behaviors in competitive pods during the breeding season and were not considered to have occurred in response to the sonar (Mobley et al., 2012).

Monitoring at the Pacific Missile Range Facility off Kaua'i has provided data on humpback and minke responses to naval anti-submarine warfare sonars in actual training conditions. Henderson et al. (2019)

examined the dive and movement behavior of tagged humpback whales, including whales incidentally exposed to sonar during Navy training activities. Tracking data showed that individual humpbacks spent limited time, no more than a few days, in the vicinity of Kaua'i, even without sonar exposure. Potential behavioral responses to sonar exposure were limited and may have been influenced by engagement in breeding and social behaviors. Martin et al. (2015) found that the density of calling minke whales was reduced during periods of Navy training involving sonar relative to the periods before training and increased again in the days after training was completed. The responses of individual whales could not be assessed, so in this case it is unknown whether the decrease in calling animals indicated that the animals left the range or simply ceased calling. Harris et al. (2019b) utilized acoustically generated minke whale tracks to statistically demonstrate changes in the spatial distribution of minke whale acoustic presence before, during, and after surface ship mid-frequency active sonar training. The spatial distribution of probability of acoustic presence was different in the "during" phase compared to the "before" phase, and the probability of presence at the center of ship activity during mid-frequency active sonar training was close to zero for both years. The "after" phases for both years retained lower probabilities of presence suggesting the return to baseline conditions may take more than five days. The results show a clear spatial redistribution of calling minke whales during surface ship mid-frequency active sonar training, however a limitation of passive acoustic monitoring is that one cannot conclude if the whales moved away, went silent, or a combination of the two.

Building on this work, Durbach et al. (2021) used the same data and determined that individual minke whales tended to be in either a fast or slow movement behavior state while on the range, where whales tended to be in the slow state in baseline or before periods but transitioned into the fast state with more directed movement during sonar exposures. They also moved away from the area of sonar activity on the range, either to the north or east depending on where the activity was located; this explains the spatial redistribution found by Harris et al. (2019b). Minke whales were also more likely to stop calling when in the fast state, regardless of sonar activity, or when in the slow state during sonar activity (Durbach et al., 2021). Similarly, minke whale detections made using Marine Acoustic Recording Instruments off Jacksonville, Florida, were reduced or ceased altogether during periods of sonar use (Norris et al., 2012; U.S. Department of the Navy, 2013), especially with an increased ping rate (Charif et al., 2015).

Other opportunistic passive acoustic based studies have also detected behavioral responses of blue and humpback whales to sonar, although definitive conclusions are harder to draw. Blue whales exposed to mid-frequency sonar in the Southern California Bight were less likely to produce low-frequency calls usually associated with feeding behavior, beginning at received levels of 110 to 120 dB re 1 µPa (Melcón et al., 2012); however, without visual observations it is unknown whether there was another factor that contributed to the reduction in foraging calls, such as the presence of conspecifics. In another example, Risch et al. (2012, 2014) determined that humpback whale song produced in the Stellwagen Bank National Marine Sanctuary was reduced while an Ocean Acoustic Waveguide Remote Sensing experiment was occurring 200 km away. They concluded that the reduced song was a result of the Ocean Acoustic Waveguide Remote Sensing. However, Gong et al. (2014) analyzed the same data set while also looking at the presence of herring in the region, and found that the singing humpbacks were actually located on nearby Georges Bank and not on Stellwagen, and that the song rate in their data did not change in response to Ocean Acoustic Waveguide Remote Sensing, but could be explained by natural causes.

Baleen whales have also been exposed to lower frequency sonars, with the hypothesis that they may react more strongly to lower frequency sounds that overlap with their vocalization range. One series of studies was undertaken in 1997–1998 pursuant to the Navy's Low Frequency Sound Scientific Research Program. The frequency bands of the low-frequency sonars used were between 100 and 500 Hz, with received levels between 115 and 150 dB re 1 µPa, and the source was always stationary. Fin and blue whales were targeted on foraging grounds, singing humpback whales were exposed on breeding grounds, and gray whales were exposed during migratory behavior. These studies found only short-term responses to low-frequency sound by some fin and humpback whales, including changes in vocal activity and avoidance of the source vessel, while other fin, humpback, and blue whales did not respond at all. When the source was in the path of migrating gray whales they changed course up to 2 km to avoid the sound, but when the source was outside their path, little response was observed although received levels were similar (Clark & Fristrup, 2001; Croll et al., 2001; Fristrup et al., 2003; Miller et al., 2000; Nowacek et al., 2007). Low-frequency signals of the Acoustic Thermometry of Ocean Climate sound source were also not found to affect dive times of humpback whales in Hawaiian waters (Frankel & Clark, 2000).

In contrast to actual or simulated naval sonar, some studies have examined responses to underwater tones or alarms intended to serve as deterrents (Table D.6-6). Migrating mysticetes sometimes responded by changing their route away from the deterrent (Dunlop et al., 2013; Frankel & Stein, 2020; Watkins & Schevill, 1975) or not at all (Harcourt et al., 2014; Morton & Symonds, 2002; Pirotta et al., 2016). Other behavioral responses caused by acoustic alarms and deterrents include reduced foraging dives, path predictability and reoxygenation rages, as well as increased swim speeds and dive durations (Boisseau et al., 2021; Nowacek et al., 2004a).

Table D.6-6: Major Findings from Studies of Acoustic Alarms and Acoustic Deterrent Devices (ADDs) in Mysticetes

Species	*Major Finding*	*Supporting Studies*
Humpback whales – wild	Changed migration course away from the deterrent (more offshore) and surfaced more frequently during 2 kHz tones.	Dunlop et al. (2013); Watkins and Schevill (1975)
Gray whales – wild	Changed migration course away from the deterrent (towards shore) during stationary sonar transmissions (21 – 25 kHz, 148 dB re 1 µPa).	Frankel and Stein (2020)
Humpback whales – wild	No change in migration route or behavioral response (even within 500 m) during 2 – 5 kHz fisheries deterrents.	Harcourt et al. (2014); Morton and Symonds (2002); Pirotta et al. (2016)
North Atlantic right whales - wild	Interrupted foraging dives during particularly long acoustic alarm (0.5 – 4.5 kHz, several minute long).	Nowacek et al. (2004a)
Minke whales - wild	Increased speed, dive duration, path predictability (straighter paths), and decreased reoxygenation rates while foraging during ADD (15 kHz, 198 dB rms). Path predictability had strong relationship with received level. Speed and dive duration more influenced by the presence of the exposure signal instead of the received sound level.	Boisseau et al. (2021)

Notes: ADD = acoustic deterrent device; dB = decibel; kHz = kilohertz; m = meters; µPa = micropascal; rms = root mean square

Although some strong responses have been observed in mysticetes to sonar and other transducers (e.g., the single minke whale), for the most part mysticete responses appear to be fairly moderate across all received levels. While some responses such as cessation of foraging or changes in dive behavior could carry short-term impacts, in all cases behavior returned to normal after the signal stopped. Mysticete responses also seem to be highly mediated by behavioral state, with no responses occurring in some behavioral states, and contextual factors and signal characteristics having more impact than received level alone. Many of the contextual factors resulting from the behavioral response studies (e.g., close approaches by multiple vessels or tagging) would never be introduced in real Navy training scenarios. While data are lacking on behavioral responses of mysticetes to continuous active sonars, these species are known to be able to habituate to novel and continuous sounds (Nowacek et al., 2004a), suggesting that they are likely to have similar responses to high-duty cycle sonars. Therefore, mysticete behavioral responses to Navy sonar will likely be a result of the animal's behavioral state and prior experience rather than external variables such as ship proximity; thus, if significant behavioral responses occur they will likely be short term. In fact, no significant behavioral responses such as panic, stranding, or other severe reactions have been observed during monitoring of actual training exercises (Smultea et al., 2009; U.S. Department of the Navy, 2011b, 2014; Watwood et al., 2012).

D.6.5.1.2 Vessel Disturbance

Behavioral responses to vessels can be caused by multiple factors. It is difficult for researchers and analysts to separate the effects of vessel noise and vessel presence; therefore, this section will cover both aspects. Baleen whales demonstrate a variety of responses to vessel traffic and noise, including not responding at all to approaching vessels, as well as both horizontal (swimming away) and vertical (increased diving) avoidance (Baker et al., 1983; Fiori et al., 2019; Gende et al., 2011; Watkins, 1981). Avoidance responses can include changes in swim patterns, speed, or direction (Jahoda et al., 2003), staying submerged for longer periods of time (Au & Green, 2000), or performing shallower dives accompanied by more frequent surfacing. Smaller-scale responses to vessels include changes such as altered breathing patterns (e.g., Baker et al., 1983; Jahoda et al., 2003), and larger-scale changes such as a decrease in apparent presence (Anderwald et al., 2013). Other common behavioral reactions include changes in vocalizations, surface time, feeding and social behaviors (Au & Green, 2000; Dunlop, 2019; Fournet et al., 2018; Machernis et al., 2018; Richter et al., 2003; Williams et al., 2002a).

Certain vessel types come with additional associated sound, other than engine and propeller cavitation noise (e.g., icebreakers). Bowhead whales avoided the area around icebreaker ship noise and increased their time at the surface and number of blows (Richardson et al., 1995a). However, bowhead whales showed no discernable long-range (greater than 8 km) behavioral reaction to various types of vessel traffic, similar to their close relative, North Atlantic right whales (Martin et al., 2023b).

Studies show that North Atlantic right whales demonstrate little if any reaction to sounds of vessels approaching or the presence of the vessels themselves. They continue to use habitats in high vessel traffic areas (Nowacek et al., 2004a). This lack of response may be due to habituation to the presence and associated noise of vessels in right whale habitat or may be due to propagation effects that may attenuate vessel noise near the surface (Nowacek et al., 2004a; Terhune & Verboom, 1999). However, right whales have been reported to increase the amplitude or frequency of their vocalizations or call at a lower rate in the presence of increased vessel noise (Parks et al., 2007; Parks et al., 2011), and these vocalization changes may persist over long periods if background noise levels remained elevated.

Other species of mysticetes seem to lack obvious reactions to vessel disturbance as well, but it may be for lack of research or variables studied. Sei whales have been observed ignoring the presence of vessels entirely and even pass close to vessels (Reeves et al., 1998). Historically, fin whales tend to ignore vessels at a distance (Watkins, 1981) or habituate to vessels over time (Watkins, 1986), but still demonstrate vocal modifications (e.g., decreased frequency parameters of calls) during boat traffic. Fin whale calls in Ireland were less likely to be detected for every 1 dB re 1 µPa/minute increase in shipping noise levels as well (Ramesh et al., 2021). In the presence of tour boats in Chile, fin whales were changing their direction of movement more frequently, with less linear movement than occurred before the boats arrived; this behavior may represent evasion or avoidance of the boats (Santos-Carvallo et al., 2021). The increase in travel swim speeds after the boats left the area may be related to the rapid speeds at which the boats left the area, sometimes in front of the animals, leading to more avoidance behavior after the boats have left.

The likelihood of any behavioral response may be driven by the density, distance or approach of vessel traffic, the animal's behavioral state, or by the prior experience of the individual or population. If the threshold of disturbance is not met for a species or group of mysticetes, there may be no behavioral reaction, as seen during a study on fin and humpback whales that largely ignored vessels that remained 100 m or more away (Watkins, 1981). When a fishing vessel conducting an acoustic survey of pelagic fisheries began moving around six whales (species unknown) at close distances (50 to 400 m), whales only slightly changed swim direction (Bernasconi et al., 2012). In areas with high motorized vessel traffic, gray whales were likely to continue feeding when approached by a vessel, but in areas with less motorized vessel traffic they were more likely to change behaviors, either indicating habituation to vessels in high traffic area, or indicating possible startle reactions to close-approaching non-motorized vessels (e.g., kayaks) in quieter areas (Sullivan & Torres, 2018).

Changes in humpback whale behavior were also affected by vessel behavior (e.g., approach type, speed), in addition to time of day and season (Di Clemente et al., 2018; Fiori et al., 2019). Avoidance responses occurred most often after "J" type vessel approaches (i.e., traveling parallel to the whales' direction of travel, then overtaking the whales by turning in front of the group) compared to parallel or direct approaches. Mother humpbacks were particularly sensitive to direct and J type approaches and spent significantly more time diving in response (Fiori et al., 2019). The presence of a passing vessel did not change the behavior of resting humpback whale mother-calf pairs, but fast vessels with louder low-frequency weighted source levels of 173 dB re 1 µPa, equating to weighted received levels of 133 dB re 1 µPa at an average distance of 100 m, led to a decrease in resting behavior and increase in dives, swim speeds, and respiration rates (Sprogis et al., 2020). Humpback whale reactions to vessel disturbance were dependent on their behavioral state. When vessels came within 500 m humpbacks would continue to feed, but were more likely to start traveling if they were surface active when approached (Di Clemente et al., 2018).

Humpback whales changed their dive times, respiration rates, and social behavior when vessels were present. In a study of large Navy vessels in Hawaii, humpback whale avoidance behaviors included increasing dive times and decreasing respiration rates at the surface when vessels were within 0.5 to 2 km (Smultea et al., 2009). Social interactions between migrating males and mother-calf pairs were reduced in the presence of vessels towing seismic air gun arrays, regardless of whether the air guns were active or not; this indicates that it was the presence of ships (rather than the active air guns) that impacted humpback behavior (Dunlop et al., 2020).

The vocal behavior and communication space for humpback whales is also impacted by vessel disturbance. In one study, whales increased the source level of their calls with increased ambient noise levels that include vessel noise (Fournet et al., 2018) and in another humpback whale call rates increased in association with high vessel noise (Doyle et al., 2008). However, there are several studies demonstrating that the probability of humpback whale calls and detections decrease when vessel noise becomes a larger part of the soundscape (Fournet et al., 2018; Laute et al., 2022). When the number of whale watching trips decreased by nearly 70 percent in an Icelandic humpback whale feeding ground, the number of humpback whale calls doubled, even though the median ambient SPL did not change (Laute et al., 2022). Humpback song activity also decreased due to boat traffic near Brazil (Sousa-Lima & Clark, 2008), and in Australia their communication area was reduced by half in average vessel-dominated noise (105 dB re 1 µPa). However the physical presence of vessels was the major contributing factor to decreased social interactions (Dunlop, 2019).

Examples of mysticete responses to tourism vessels, with an emphasis on humpback whale responses, are detailed in Table D.6-7.

Table D.6-7: Examples of Behavioral Responses to Vessel Disturbance in Mysticetes

Species	***Study***	***Boat type***	***Behavioral Change***					
			Feeding or foraging	***Surface behaviors***	***Resting***	***Respiration Rates***	***Diving duration***	***Horizontal avoidance (Δ direction or speed)***
Humpback whales – Hawaii	Baker et al. (1983)	Tour					↑ when < 2,000 m away	↑ when 2,000 – 4,000 m away
Humpback whales – Australia	Stamation et al. (2010)	Tour		↓		NC	↑	↑ or ↓ (avoid or approach)
Humpback whales – Alaska	Schuler et al. (2019); Toro et al. (2021)	Tour		↓		↑		↑
Minke whales – Iceland	Christiansen et al. (2013)	Tour	↓			↑ (↓ IBI)		
Blue whales – Canada	Lesage et al. (2017)	Tour	↓	↓		↓	↓	
Fin whales – Chile	Santos-Carvallo et al. (2021)	Tour						↑
Sperm whales – Portugal	Oliveira et al. (2022)	Tour	NC		↓ (↑ movement)			↑ speed of ascent
Southern right whales – Australia	Sprogis et al. (2023)	Tour			↓	NC		

Notes: ↑ = increase; ↓ = decrease; < = less than; Δ = change in; IBI = Inter-breath interval; m = meters; NC = no change

Blue whale response to vessel disturbance varies from increasing the likelihood of producing certain types of calls when vessels pass (Melcón et al., 2012), to general avoidance behavior (Lesage et al., 2017; Szesciorka et al., 2019). In an area of high whale watch activity, vessels were within 2,000 m of blue whales 70 percent of the time, with a maximum of 8 vessels observed within 400 m of one whale at the same time. In response to repeated exposures to vessels, blue whales decreased time at the surface, had fewer breaths at the surface, shorter dive times and less time foraging as a result (Lesage et al., 2017). In response to an approaching large commercial vessel in an area of high ambient noise levels (125–130 dB re 1 µPa), a tagged female blue whale turned around mid-ascent and descended perpendicular to the ship's path (Szesciorka et al., 2019). The whale did not respond until the ship's closest point of approach (100 m distance, 135 dB re 1 µPa), which was 10 dB above the ambient noise levels. After the ship passed, the whale ascended to the surface again with a three-minute delay.

Minke whale negative, neutral, or positive response to vessels may be influenced by vessel speed and boat traffic density. In the Antarctic minke whales did not show any apparent response to a survey vessel moving at normal cruising speeds (about 12 knots) at a distance of 5.5 NM. However, when the vessel drifted or moved at very slow speeds (about 1 knot), many whales approached it (Leatherwood et al., 1982). Larger-scale negative changes like habitat displacement was found during a construction project in the United Kingdom, when fewer minke whales were observed as vessel traffic increased (Anderwald et al., 2013). Likewise, minke whales on feeding grounds in Iceland responded to increased whale watching vessel traffic with a decrease in foraging, both during deep dives and at the surface (Christiansen et al., 2013). They also increased their avoidance of the boats while decreasing their respiration rates, likely leading to an increase in their metabolic rates. Christiansen and Lusseau (2015) and Christiansen et al. (2014) followed up this study by modeling the cumulative impacts of whale watching boats on minke whales, but found that although the boats cause temporary feeding disruptions, there were not likely to be long-term consequences as a result. This suggests that short-term responses may not lead to long-term consequences and that over time animals may habituate to the presence of vessel traffic.

Longitudinal studies on vessel noise have been conducted, but the consequences of chronic vessel noise are not well understood. Using historical records, Watkins (1986) showed that the reactions of four species of mysticetes to vessel traffic and whale watching activities in Cape Cod had changed over 25 years (1957–1982). Reactions of minke whales changed from initially more positive reactions, such as coming towards the boat or research equipment to investigate, to more uninterested reactions towards the end of the study. Fin whales, the most numerous species in the area, showed a trend from initially more negative reactions, such as swimming away from the boat with limited surfacing, to more uninterested reactions (ignoring), allowing boats to approach within 30 m. Right whales showed little change over the study period, with a roughly equal number of reactions judged to be negative and uninterested; no right whales were noted as having positive reactions to vessels. Humpback whales showed a trend from negative to positive reactions with vessels during the study period. The author concluded that the whales had habituated to the human activities over time (Watkins, 1986).

Overall baleen whale responses to vessel noise and traffic are varied, and habituation or changes to vocalization are predominant long-term responses. When baleen whales do avoid ships, they do so by altering their swim and dive patterns to move away from the vessel. In many cases the whales do not appear to change their behavior at all. This may result from habituation by the whales but may also result from reduced received levels near the surface due to propagation, or due to acoustic shadowing of the propeller cavitation noise by the ship’s hull. Although a lack of response in the presence of a vessel may minimize potential disturbance from passing ships, it does increase the whales’ vulnerability to vessel strike, which may be of greater concern for baleen whales than vessel noise.

D.6.5.1.3 Aircraft Disturbance

Mysticetes either ignore or occasionally dive in response to aircraft overflights (Koski et al., 1998). Richardson et al. (1985); Richardson et al. (1995b) found no evidence that single or occasional aircraft flying above mysticetes causes long-term displacement of these mammals. However, bowhead whales in the Beaufort Sea exhibited a short-term behavioral response to fixed-wing aircraft and vessels. Reactions were frequently observed at less than 1,000 ft. above sea level, infrequently observed at 1,500 ft., and not observed at all at 2,000 ft. (Richardson et al., 1985).

Bowhead whales reacted to helicopter overflights by diving, breaching, changing direction or behavior, and altering breathing patterns. Behavioral reactions decreased in frequency as the altitude of the helicopter increased to 150 m or higher. The bowheads exhibited fewer behavioral changes than did the odontocetes in the same area (Patenaude et al., 2002). It should be noted that bowhead whales in this study may have more acute responses to anthropogenic activity than many other marine mammals since these animals were presented with restricted egress due to limited open water between ice floes.

Additionally, these animals are hunted by Alaska Natives, which could lead to animals developing additional sensitivity to human noise and presence.

Studies on unmanned aerial systems have not found significant behavioral responses from mysticetes so far. These devices are much smaller and quieter than typical aircraft, and so are less likely to cause a behavioral response, although they may fly at much lower altitudes (Smith et al., 2016). Acevedo-Whitehouse et al. (2010) maneuvered a remote-controlled helicopter over large baleen whales to collect samples of their blows, with no more avoidance behavior than noted for typical photo-identification vessel approaches. Bowhead whales did not respond to an unmanned system flying at altitudes between 120 and 210 m above the ocean's surface (Koski et al., 2015; Koski et al., 1998). While collecting humpback photogrammetry and fitness data, Christiansen et al. (2016) did not observe any responses to their unmanned aerial vehicle flown 30–120 m above the water either. Even 10 southern right whale mother-calf pairs showed no change in swim speed, respiration rate, turning angle, or interbreath interval in response to an unmanned aerial vehicle (Christiansen et al., 2020). Some of the animals were equipped with DTAGs to measure the sound of the unmanned aerial vehicle; the received levels in the 100–1,500 Hz band were 86 ± 4 dB re 1 µPa, very similar to ambient noise levels measured at 81 ± 7 dB in the same frequency band.

D.6.5.1.4 Impulsive Noise

Baleen whales have shown a variety of responses to impulsive sound sources, including avoidance, aggressive directed movement towards the source, reduced surface intervals, altered swimming behavior, and changes in vocalization rates (Gordon et al., 2003; McCauley et al., 2000a; Richardson et al., 1985; Southall et al., 2007). Studies have been conducted on many baleen whale species, including gray, humpback, blue, fin and bowhead whales; it is assumed that these responses are representative of all baleen whale species. The behavioral state of the whale seems to be an integral part of whether the animal responds and how they respond, as does the location and movement of the sound source, more than the received level of the sound.

Migratory behavior seems to lead to a higher likelihood of response, with some species demonstrating more sensitivity than others do. For example, migrating gray whales showed avoidance responses to seismic vessels at received levels between 164 and 190 dB re 1 µPa (Malme et al., 1986, 1988). Similarly, migrating humpback whales showed avoidance behavior at ranges of 5–8 km from a seismic array during observational studies and controlled exposure experiments in one Australian study (McCauley et al., 1998), and in another Australian study decreased their dive times and reduced their swimming speeds (Dunlop et al., 2015). However, when comparing received levels and behavioral responses using ramp-up versus a constant noise level of air guns, humpback whales did not change their dive behavior but did deviate from their predicted heading and decreased their swim speeds (Dunlop et al., 2016). In addition, the whales demonstrated more course deviation during the constant source trials but reduced travel speeds more in the ramp-up trials; in either case there was no dose-response relationship with the received level of the air gun noise, and similar responses were observed in control trials with vessel movement but no air guns so some of the response was likely due to the presence of the vessel and not the received level of the air guns. Similar results were found in migrating humpback whales (Dunlop et al., 2020). Social interactions between males and mother-calf pairs were reduced in the presence of vessels towing seismic air gun arrays, regardless of whether the air guns were active or not; this indicates that it was the presence of ships (rather than the active air guns) that impacted humpback behavior (Dunlop et al., 2020).

When looking at the relationships between proximity, received level, and behavioral response, Dunlop et al. (2017) used responses to two different air guns and found responses occurred more towards the smaller, closer source than to the larger source at the same received level, demonstrating the importance of proximity. Responses were found to be more likely when the source was within 3 km or above 140 dB re 1 µPa, although responses were variable and some animals did not respond at those values while others responded below them. In addition, responses were generally small, with short term course deviations of only around 500 m (Dunlop et al., 2017). McDonald et al. (1995) tracked a blue whale with seafloor seismometers and reported that it stopped vocalizing and changed its travel direction at a range of 10 km from the seismic vessel (estimated received level 143 dB re 1 µPa peak-to-peak). Bowhead whales seem to be the most sensitive species. While most bowhead whales did not show active avoidance until within 8 km of seismic vessels (Richardson et al., 1995b), some whales avoided vessels by more than 20 km at received levels as low as 120 dB re 1 µPa. Additionally, Malme et al. (1988) observed clear changes in diving and breathing patterns in bowheads at ranges up to 73 km from seismic vessels, with received levels as low as 125 dB re 1 µPa. Bowhead whales may also avoid the area around seismic surveys, from 6 to 8 km (Koski and Johnson 1987, as cited in Gordon et al., 2003) out to 20 or 30 km (Richardson et al., 1999). However, work by Robertson et al. (2013) supports the idea that behavioral responses are contextually dependent, and that during seismic operations bowhead whales may be less “available” for counting due to alterations in dive behavior but that they may not have left the area after all.

In contrast, noise from seismic surveys was not found to impact feeding behavior or exhalation rates in western gray whales while resting or diving off the coast of Russia (Gailey et al., 2007; Yazvenko et al., 2007); however, the increase in vessel traffic associated with the surveys and the proximity of the vessels to the whales did affect the orientation of the whales relative to the vessels and shortened their dive-surface intervals (Gailey et al., 2016). They also increased their speed and distance from the noise source, and will even travel towards shore to avoid an approaching seismic vessel, as shown in one case study (Gailey et al., 2022). Todd et al. (1996) found no clear short-term behavioral responses by foraging humpbacks to explosions associated with construction operations in Newfoundland but did see a trend of increased rates of net entanglement closer to the noise source, possibly indicating a reduction in net detection associated with the noise through masking or TTS. Distributions of fin and minke whales were modeled with a suite of environmental variables along with the occurrence or absence of seismic surveys, and no evidence of a decrease in sighting rates relative to seismic activity was found for either species (Vilela et al., 2016). Their distributions were driven entirely by environmental variables, particularly those linked to prey including warmer sea surface temperatures, higher chlorophyll-a values, and higher photosynthetically available radiation (a measure of primary productivity). Sighting rates based on over 8,000 hours of baleen and toothed whale survey data were compared on regular vessel surveys versus both active and passive periods of seismic surveys (Kavanagh et al., 2019). Models of sighting numbers were developed, and it was determined that baleen whale sightings were reduced by 88 and 87 percent during active and inactive phases of seismic surveys, respectively, compared to regular surveys. These results seemed to occur regardless of geographic location of the survey; however, when only comparing active versus inactive periods of seismic surveys the geographic location did seem to affect the change in sighting rates.

Vocal and other behavioral responses to seismic surveys have been observed in a number of baleen whale species, including a cessation of calling, a shift in frequency, increases in amplitude or call rate, leaving the area, or a combination of these strategies (Blackwell et al., 2013; Blackwell et al., 2015; Blackwell et al., 2017; Di Lorio & Clark, 2010). For example, responses by fin whales to a 10-day seismic survey in the Mediterranean Sea included possible decreased call production and movement away from the area (Castellote et al., 2012). Models of humpback whale song showed a decrease in the number of singers with increasing received levels of air gun pulses, indicating either a vocal modification or that whales left the area (Cerchio et al., 2014).

Mysticetes seem to be the most behaviorally sensitive taxonomic group of marine mammals to impulsive sound sources, with possible avoidance responses occurring out to 30 km and vocal changes occurring in response to sounds over 100 km away. However, they are also the most studied taxonomic group, yielding a larger sample size and greater chance of finding behavioral reactions to impulsive noise. Also, their responses appear to be behavior-dependent, with most avoidance responses occurring during migration behavior and little observed response during feeding behavior. These response patterns are likely to hold true for Navy impulsive sources; however, Navy impulsive sources would largely be stationary (e.g., explosives fired at a fixed target), and short term (on the order of hours rather than days or weeks) than were found in these studies and so responses would likely occur in closer proximity or not at all.

D.6.5.2 Behavioral Reactions of Odontocetes

D.6.5.2.1 Sonar and Other Transducers

D.6.5.2.1.1 Beaked Whales

Following several beaked whale strandings in which military mid-frequency active sonar was identified as a contributing cause or factor, the scientific community hypothesized that these deep-diving species may be more susceptible to behavioral disturbance or behaviorally mediated physiological consequences. Subsequently, behavioral response studies in which beaked whales were intentionally or incidentally exposed to real or simulated sonar, in some cases on military ranges, found that beaked whales are likely more sensitive to disturbance than most other cetaceans. Observed reactions by Blainville's beaked whales, goose-beaked whales, and Baird's beaked whales, as well as northern bottlenose whales (the largest of the beaked whales), to mid-frequency sonar sounds include cessation of clicking, decline in group vocal periods, termination of foraging dives, changes in direction to avoid the sound source, slower ascent rates to the surface, longer deep and shallow dive durations, and other unusual dive behaviors (DeRuiter et al., 2013b; Hewitt et al., 2022; Jacobson et al., 2022; McCarthy et al., 2011; Miller et al., 2015; Moretti et al., 2014; Southall et al., 2011; Stimpert et al., 2014; Tyack et al., 2011).

Research on beaked whales includes detailed response data from exposures of focal, tagged animals, as well as wide-scale analyses of changes in group vocal behaviors across instrumented ranges. Many of the exposures to tagged animals occurred within 1–8 km of the focal animal, within a few hours of tagging the animal, and with one or more boats within a few kilometers to observe responses and record acoustic data. Thus, while tagged animal data is precise and detailed, the animal's behavior may be influenced by the experimental context. In addition, individual variability can only be assessed with many tagged individuals. In contrast, group vocal behavior observations from instrumented ranges do not provide fine-scale movement and behavior data for individuals but allow for assessing responses across a range-wide population in real-world military training conditions.

Beaked whales have been tagged and exposed to sonar across multiple efforts (e.g., AUTEC, 3S2, SOCAL BRS, Atlantic BRS). During the SOCAL BRS, a tagged Baird's beaked whale exposed to simulated mid-frequency active sonar within 3 km increased swim speed and modified its dive behavior (Stimpert et al., 2014). One goose-beaked whale (formerly Cuvier's beaked whale) was also incidentally exposed to real Navy sonar located over 100 km away in addition to the source used in the controlled exposure study, and the authors did not detect similar responses at comparable received levels. Received levels from the mid-frequency active sonar signals from the controlled (3.4 to 9.5 km) and incidental (118 km) exposures were calculated as 84 to 144 and 78 to 106 dB re 1 µPa, respectively, indicating that context of the exposures (e.g., source proximity, controlled source ramp-up) may have been a significant factor in the responses to the simulated sonars (DeRuiter et al., 2013b).

Long-term tagging work on the SOCAL BRS has demonstrated that the longer duration dives considered a behavioral response by DeRuiter et al. (2013b) fell within the normal range of dive durations found for eight tagged goose-beaked whales on the Southern California Offshore Range (Schorr et al., 2014). However, the longer inter-deep dive intervals found by DeRuiter et al. (2013b), which were among the longest found by Schorr et al. (2014) and Falcone et al. (2017), could indicate a response to sonar. In addition, Williams et al. (2017) note that in normal deep dives or during fast swim speeds, beaked whales and other marine mammals use strategies to reduce their stroke rates, including leaping or wave surfing when swimming, and interspersing glides between bouts of stroking when diving. They determined that in the post-exposure dives by the tagged goose-beaked whales described in DeRuiter et al. (2013b), the whales ceased gliding and swam with almost continuous strokes. This change in swim behavior was calculated to increase metabolic costs about 30.5 percent and increase the amount of energy expending on fast swim speeds from 27 to 59 percent of their overall energy budget. This repartitioning of energy was detected in the model up to 1.7 hours after the single sonar exposure. Therefore, while the overall post-exposure dive durations were similar, the metabolic energy calculated by Williams et al. (2017) was higher. However, Southall et al. (2019a) found that prey availability was higher in the western area of the Southern California Offshore Range where goose-beaked whales preferentially occurred, while prey resources were lower in the eastern area and moderate in the area just north of the Range. This high prey availability may indicate that fewer foraging dives are needed to meet metabolic energy requirements than would be needed in another area with fewer resources.

During the 3S2 Project, the roles of sound source distance and received level in northern bottlenose whales were analyzed in an environment without frequent sonar activity using controlled exposure experiments (von Benda-Beckmann et al., 2019; Wensveen et al., 2019). Researchers observed behavioral avoidance of the sound source over a wide range of distances (0.8 to 28 km) and estimated avoidance thresholds ranging from received SPLs of 117 to 126 dB re 1 µPa. The behavioral response characteristics and avoidance thresholds were comparable to those previously observed in beaked whale studies; however, they did not observe an effect of distance on behavioral response and found that onset and intensity of behavioral response were better predicted by received SPL. One northern bottlenose whale did approach the ship and circle the source, then resumed foraging after the exposure, but the source level was only 122 dB re 1 µPa. A northern bottlenose whale conducted the longest and deepest dive on record for that species after sonar exposure and continued swimming away from the source for over seven hours (Miller et al., 2015; Siegal et al., 2022; Wensveen et al., 2019).

On the AUTEC range, Blainville's beaked whales located on-range appear to move off-range during sonar use and return only after the sonar transmissions have stopped, sometimes taking several days to do so (Boyd et al., 2009; Henderson et al., 2015; Jones-Todd et al., 2021; Manzano-Roth et al., 2022; Manzano-Roth et al., 2016; McCarthy et al., 2011; Tyack et al., 2011). For example, five Blainville's beaked whales that were estimated to be within 2 to 29 km of the AUTEC range at the onset of sonar were displaced a maximum of 28–68 km from the range after moving away from the range, although one whale approached the range during the period of active sonar. Researchers found a decline in deep dives at the onset of the training and an increase in time spent on foraging dives as individuals moved away from the range. Predicted received levels at which presumed responses were observed were comparable to those previously observed in beaked whale studies. Acoustic data indicated that vocal periods were detected on the range within 72 hours after training ended (Joyce et al., 2019). However, Blainville's beaked whales remain on the range to forage throughout the rest of the year (Henderson et al., 2016), possibly indicating that this a preferred foraging habitat regardless of the effects of the noise, or it could be that there are no long-term consequences of the sonar activity. Similarly, photo-identification studies in the SOCAL Range Complex have identified approximately 100 individual goose-beaked whale individuals, with 40 percent having been seen in one or more prior years, with re-sightings up to seven years apart, indicating a possibly resident population on the range (Falcone & Schorr, 2014; Falcone et al., 2009).

The probability of Blainville's beaked whale group vocal periods on the Pacific Missile Range Facility were modeled during periods of no naval activity, naval activity without hull-mounted mid-frequency active sonar, and naval activity with hull-mounted mid-frequency active sonar (Jacobson et al., 2022). At a received level of 150 dB re 1 µPa rms (root mean square), the probability of group vocal period detection decreased by 77 percent compared to periods when general training activity was ongoing and by 87 percent compared to baseline conditions. This study found a greater reduction in probability of a group vocal period with mid-frequency active sonar than observed in a prior study of Blainville's beaked whales at AUTEC (Moretti et al., 2014). The authors suggest that this may be due to the baseline period in the AUTEC study including naval activity without mid-frequency active sonar, potentially lowering the baseline group vocal period activity in that study, or due to differences in the residency of the populations at each range. Along the edge of the Scotian Shelf off eastern Canada, baseline activity from both prior to a period of naval sonar use and a prior year with no known naval activity were used to assess changes in beaked whale detections. Goose-beaked and Mesoplodant beaked whale detection rates dropped both during and after an eight-day, multi-platform anti-submarine warfare training exercise, and remained low seven days after the exercise (Stanistreet et al., 2022).

On the Southern California Anti-Submarine Warfare Range, deep and shallow dive durations, surface interval durations, and inter-deep dive intervals of goose-beaked whales were modeled against predictor values that included helicopter dipping, mid-power mid-frequency active sonar and hull-mounted, high-power mid-frequency active sonar along with other, non-mid-frequency active sonar predictors (Falcone et al., 2017). They found both shallow and deep dive durations increased as the proximity to both mid- and high-powered sources decreased, and found that surface intervals and inter-deep dive intervals increased in the presence of both types of sonars, although surface intervals shortened during periods of no mid-frequency active sonar. The responses to the mid-power mid-frequency active sonar at closer ranges were comparable to the responses to the higher source level ship sonar, again highlighting the importance of proximity. This study also supports context as a response factor, as helicopter dipping sonars are shorter duration and randomly located, so more difficult for beaked whales to predict or track and therefore potentially more likely to cause a response, especially when they occur at closer distances (6 to 25 km in this study). Sea floor depths and quantity of light are also important variables to consider in goose-beaked whale behavioral response studies, as their foraging dive depth increased with sea floor depth up to sea floor depths of 2,000 m. The fraction of time spent at foraging depths and likely foraging was greater at night, although they spent more time near the surface during the night as well, particularly on dark nights with little moonlight, likely avoiding predation by staying deeper during periods of bright lunar illumination (Barlow et al., 2020). Sonar occurred during 10 percent of the dives studied and had little effect on the resulting dive metrics. Watwood et al. (2017) found that helicopter dipping events occurred more frequently but with shorter durations than periods of hull-mounted sonar, and also found that the longer the duration of a sonar event, the greater reduction in detected goose-beaked whale group dives. Therefore, when looking at the number of detected group dives there was a greater reduction during periods of hull-mounted sonar than during helicopter dipping sonar. Similar results were found by DiMarzio et al. (2019).

Echosounders

Beaked whales may respond similarly to shipboard echosounders, commonly used for navigation, fisheries, and scientific purposes, with frequencies ranging from 12 to 400 kHz and source levels up to 230 dB re 1 µPa but typically a very narrow beam (Cholewiak et al., 2017). During a scientific cetacean survey, an array of echosounders was used in a one-day-on, one-day-off paradigm. Beaked whale acoustic detections occurred predominantly (96 percent) when the echosounder was off, with only 4 detections occurring when it was on. Beaked whales were sighted fairly equally when the echosounder was on or off, but sightings were further from the ship when the echosounder was on (Cholewiak et al., 2017). These findings indicate that the beaked whales may be avoiding the area and may cease foraging near the echosounder. Another study also found that echosounders contributed to fewer beaked whale observations, but ultrasonic antifouling devices elicited an even stronger avoidance response (Trickey et al., 2022).

In contrast, goose-beaked whale group vocal periods during multibeam echosounder activity recorded in the Southern California Antisubmarine Warfare Range did not decrease during the echosounder survey (Varghese et al., 2020). The whales did not leave the range or cease foraging, and group vocal periods increased during and after multibeam echosounder surveys. Since echosounders are highly directional and the sound doesn't propagate horizontally, the difference in these results may be due to the locations of beaked whales relative to the echosounder. In fact, one of the surveys by Varghese et al. (2020) was largely conducted on a portion of the range little used by goose-beaked whales. A subsequent analysis suggested that the observed spatial shifts were most likely due to prey dynamics (Varghese et al., 2021).

Predator Sounds

Tyack et al. (2011) hypothesized that beaked whale responses to sonar may represent an anti-predator response. To test this idea, vocalizations of a potential predator—a killer whale—were played back to a Blainville's beaked whale at AUTEC. The killer whale vocalization recording was from a stock of mammal-eating killer whales that are not present at AUTEC. This exposure resulted in a similar but more pronounced reaction than that elicited by sonar playback, which included longer inter-dive intervals and a sustained straight-line departure of more than 20 km from the area (Allen et al., 2014; Tyack et al., 2011). Similarly, De Soto et al. (2020) hypothesized that the high degree of vocal synchrony in beaked whales during their deep foraging dives, coupled with their silent, low-angled ascents, have evolved as an anti-predator response to killer whales. Since killer whales do not dive deep when foraging and so may be waiting at the surface for animals to finish a dive, these authors speculated that by diving in spatial and vocal cohesion with all members of their group, and by surfacing silently and up to 1 km away from where they were vocally active during the dive, they minimize the ability of killer whales to locate them when at the surface. This may lead to a trade-off for the larger, more fit individuals that can conduct longer foraging dives, such that all members of the group remain together and are better protected by this behavior. The authors speculate that this may explain the long, slow, silent, and shallow ascents that beaked whales make when sonar occurs during a deep foraging dive. However, these hypotheses are based only on the dive behavior of tagged beaked whales, with no observations of predation attempts by killer whales, and "anti-predator response" theory needs to be tested further to be validated. This anti-predator hypothesis was also tested by playing back killer whale vocalizations to northern bottlenose whales and several other odontocetes to determine responses by both potential prey and conspecifics (Miller, 2012; Miller et al., 2011). Results varied in other odontocetes, from no response to an increase in group size and attraction to the source (Curé et al., 2012). Changes in foraging duration during killer whale playbacks and mid-frequency sonar were positively correlated across four species in the 3S Norwegian studies, including northern bottlenose whales, suggesting that tolerance of predation risk may play a role in sensitivity to sonar disturbance (Miller et al., 2022).

D.6.5.2.1.2 Harbor Porpoises

There are very few behavioral response studies on harbor porpoise reactions to sonar, but there are many reports of porpoise responding to other tonal sounds such as acoustic harassment devices (AHDs) and acoustic deterrent devices (ADDs). AHDs and ADDs, which transmit sound into the acoustic environment like Navy sources, have been used to deter marine mammals from fishing gear both to prevent entanglement and to reduce depredation (taking fish). These devices have been used successfully to deter harbor porpoises and beaked whales from getting entangled in fishing nets. See Table D.6-8 for a summary of the major findings from studies of the effects of AHDs and ADDs in harbor porpoises.

Table D.6-8: Major Findings from Studies of Acoustic Harassment Devices (AHDs) and Acoustic Deterrent Devices (ADDs) in Harbor Porpoises

Major Finding	***Supporting Studies***
High-frequency acoustic alarms with varied duration, interval, and sweep characteristics can prove to be effective deterrents for harbor porpoises.	Kastelein et al. (2006); Kastelein et al. (2001); Kastelein et al. (2017)
Detection rates were reduced by ADDs, especially in close proximity (< 100 m away, limited to a few 100 m at most). Tested with many pinger parameters (e.g., 10 kHz tone with and without 30 to 60 kHz sweep, 50 – 120 kHz).	Findlay et al. (2024); Kindt-Larsen et al. (2019); Kyhn et al. (2015); Omeyer et al. (2020)
Simulated AHD (12 kHz, 165 dB re 1 µPa) caused avoidance (physically moved away) from the source up to 525 m.	Mikkelsen et al. (2017)
Detection rates within 100 m were reduced by banana pingers designed to avoid pinniped responses, but had no effect at 400 m.	Königson et al. (2021)
Habituation to short-term exposures (2 to 4 exposures).	Kyhn et al. (2015)
No habituation (remained avoidant/silent) while pingers were on, especially over longer-term exposures (28+ days).	Kyhn et al. (2015); Omeyer et al. (2020)
Habituation to a pinger may occur with single tones but is less likely with a mixture of signals.	Kindt-Larsen et al. (2019)
When pinger was shut off, clicking returned to normal levels (no long-term displacement).	Omeyer et al. (2020)
Modeled results found that when pingers were used alone (in the absence of gillnets or time-area closures), caused enough habitat displacement for 21% population-level reduction.	van Beest et al. (2017)
Net pingers are successful at reducing entanglements for harbor porpoise and beaked whales since these species are not depredating from the nets but are getting entangled when foraging in the area and are unable to detect the net.	Carretta et al. (2008); Schakner and Blumstein (2013)
Lower broadband source SPL, SEL, and duty cycle “startle sounds” compared to other ADDs resulted in avoidance behaviors for duration of exposure (+ 31 minutes minimum). Travelled at least 1 km (> 3 km maximum) within 15 minutes of exposure, increased group cohesion and swim speed away from the transducer.	Hiley et al. (2021)

Table D.6-8: Major Findings from Studies of Acoustic Harassment Devices (AHDs) and Acoustic Deterrent Devices (ADDs) in Harbor Porpoises (continued)

Major Finding	*Supporting Studies*
Avoided high-frequency ADDs (60 – 150 kHz, 172 dB re 1 µPa rms) up to 2.5 km away. Reduced occurrence by 30 to 100% at 750 m.	Voß et al. (2023)
Swam quickly away from low received level AHDs (14 kHz, 98-132 dB re 1 µPa), decreased echolocation rate, and either increased or decreased heart rate. Waited 15 – 42 minutes to resume feeding behavior.	Elmegaard et al. (2023) Elmegaard et al. (2021)

Notes: % = percent; > = greater than; < = less than; ADD = acoustic deterrent device; AHD = acoustic harassment device; dB = decibel; kHz = kilohertz; km = kilometer; m = meters; µPa = micropascal; SEL = sound exposure level; SPL = sound pressure level

Behavioral responses by harbor porpoises to a variety of sound sources other than acoustic alarms have been studied (Kastelein et al., 2006; Kastelein et al., 2001), including emissions for underwater data transmission (Kastelein et al., 2005b), and tones such as 1 to 2 kHz and 6 to 7 kHz sweeps with and without harmonics (Kastelein et al., 2014c), 25 kHz tones with and without sidebands (Kastelein et al., 2015e; Kastelein et al., 2015f), and mid-frequency sonar tones at 3.5 to 4.1 kHz at 2.7 percent and 96 percent duty cycles (e.g., one tone per minute versus a continuous tone for almost a minute) (Kastelein et al., 2018b). Responses include increased respiration rates, swim speed, jumping, swimming further from the source, or decreasing echolocation rate which increases risk of wild harbor porpoise becoming by-catch (Elmegaard et al., 2021). However, responses were different depending on the source. For example, harbor porpoises responded to the 1 to 2 kHz upsweep at 123 dB re 1 μPa, but not to the downsweep or the 6 to 7 kHz tonal at the same level (Kastelein et al., 2014c). When measuring the same sweeps for a startle response, the 50 percent response threshold was 133 and 101 dB re 1 μPa for 1 to 2 kHz and 6 to 7 kHz sweeps, respectively, when no harmonics were present, and decreased to 90 dB re 1 μPa for 1 to 2 kHz sweeps with harmonics present (Kastelein et al., 2014c).

Kastelein et al. (2019a) examined the potential masking effect of high sea state ambient noise on captive harbor porpoise perception of and response to high duty cycle playbacks of AN/SQS-53C sonar signals by observing their respiration rates. Results indicated that sonar signals were not masked by the high sea state noise, and received levels at which responses were observed were similar to those observed in prior studies of harbor porpoise behavior. However, in another study sonar sweeps did not elicit a startle response in captive harbor porpoises; instead initial exposures induced bradycardia (slowing of the heart rate), with subsequent habituation that was conserved for at least three years (Elmegaard et al., 2021).

Harbor porpoises did not respond to the low-duty cycle mid-frequency tones at any received level, but one did respond to the high-duty cycle signal with more jumping and increased respiration rates (Kastelein et al., 2018b). Harbor porpoises responded to seal scarers with broadband signals up to 44 kHz with a slight respiration response at 117 dB re 1 μPa and an avoidance response at 139 dB re 1 μPa, but another scarer with a fundamental (strongest) frequency of 18 kHz did not have an avoidance response until 151 dB re 1 μPa (Kastelein et al., 2015d). Exposure of the same acoustic pinger to a striped dolphin under the same conditions did not elicit a response (Kastelein et al., 2006), again highlighting the importance in understanding species differences in the tolerance of underwater noise, although sample sizes in these studies was small so these could reflect individual differences as well.

D.6.5.2.1.3 Other Odontocetes

While there has been a focus on beaked whale (and to a lesser extent harbor porpoise) response to sonar and similar transducers, other species have been studied during behavioral response studies as well, including pilot whales, killer whales, sperm whales, false killer whales, melon-headed whales, bottlenose dolphins, rough-toothed dolphins, Risso's dolphins, Pacific white-sided dolphins, and Commerson's dolphins. Responses by these species include horizontal avoidance, reduced breathing rates, changes in behavioral state, and changes in dive behavior (Antunes et al., 2014; Isojunno et al., 2018; Isojunno et al., 2017; Isojunno et al., 2020; Miller, 2012; Miller et al., 2011; Miller et al., 2014). Some species like pilot whales, false killer whales and Risso's dolphins will also respond by mimicking the sound of the sonar with their whistles (Alves et al., 2014; DeRuiter et al., 2013a; Smultea et al., 2012).

More severe behavioral responses, such as separation of a killer whale calf from its group, have been observed during exposure to mid-frequency sonar playbacks (Miller et al., 2011). However, it is difficult to tease this response to sonar apart from the animals' response to the perusing research vessel in an environment with limited egress. Received level thresholds at the onset of avoidance behavior were generally lower for killer whales (mean 129 dB re 1 µPa) compared to pilot whales (mean 150 dB re 1 µPa) and sperm whales (mean 140 dB re 1 µPa) (Antunes et al., 2014; Curé et al., 2021; Miller, 2012; Miller et al., 2014). Tagged odontocetes (e.g., pilot whales, rough-toothed dolphins, bottlenose dolphins, and false killer whales) did not have an avoidance response to sonar on or near the Pacific Missile Range Facility before Navy training events (Baird et al., 2014; Baird et al., 2017; Baird et al., 2013). In some cases, odontocetes even traveled towards areas of higher noise levels, while estimated received SPLs varied from 130 to 168 dB re 1 µPa and distances from sonar sources ranged between 3.2 and 94.4 km.

Not all responses to sonar cause avoidance responses or deleterious changes in behavior. Navy exercises involving sonar on large ships may also attract odontocetes or cause no reaction, depending on the species. While most of the published literature involving bowriding odontocete observations does not involve sonar, certain species (e.g., bottlenose, spotted, spinner, Clymene, Pacific white sided, rough-toothed dolphins) will sometimes approach vessels to bow ride, indicating either that these species are less sensitive to vessels or that the behavioral drive to bow ride supersedes any impact of the associated noise (Würsig et al., 1998). During aerial and visual monitoring of Navy training events, rough-toothed dolphins and unidentified dolphins were observed approaching the vessel with active sonar as if to bow ride. Spotted dolphins were observed nearby but did not avoid or approach the vessel (Mobley, 2011; U.S. Department of the Navy, 2011a; Watwood et al., 2012). During small boat surveys near the Southern California Offshore Range in southern California, more dolphins were encountered in June compared to a similar survey conducted the previous November after seven days of mid-frequency sonar activity; it was not investigated if this change was due to the sonar activity or was due to the poor weather conditions in November that may have prevented animals from being seen (Campbell et al., 2010). There were also fewer passive acoustic dolphin detections during and after longer sonar activities in the Mariana Islands Range Complex, with the post-activity absence lasting longer than the mean dolphin absence of two days when sonar was not present (Munger et al., 2014; Munger et al., 2015).

Controlled experiments have also been conducted on captive animals to estimate received levels at which behavioral responses occur. In one study, bottlenose dolphin behavioral responses were recorded when exposed to 3 kHz sonar-like tones between 115 and 185 dB re 1 µPa (Houser et al., 2013a), and in another study bottlenose dolphins and beluga whales were presented with one-second tones up to 203 dB re 1 µPa to measure TTS (Finneran et al., 2003a; Finneran et al., 2001; Finneran et al., 2005; Finneran & Schlundt, 2004; Schlundt et al., 2000). During these studies, responses included changes in respiration rate, fluke slaps, and a refusal to participate or return to the location of the sound stimulus. This refusal included what appeared to be deliberate attempts to avoid a sound exposure or to avoid the location of the exposure site during subsequent tests (Finneran et al., 2002; Schlundt et al., 2000). In the behavioral response study, bottlenose dolphins demonstrated a 50 percent probability of response at 172 dB re 1 µPa over 10 trials. In the TTS experiment, bottlenose dolphins exposed to one-second intense tones exhibited short-term changes in behavior above received sound levels of 178 to 193 dB re 1 μPa; beluga whales did so at received levels of 180 to 196 dB re 1 μPa and above. In some instances, animals exhibited aggressive behavior toward the test apparatus (Ridgway et al., 1997; Schlundt et al., 2000). While animals were commonly reinforced with food during these studies, the controlled environment provided insight on received levels at which animals behaviorally respond to noise sources.

There are opportunistic observations of behavioral responses to sonar as well, although in those cases it is difficult to attribute observed responses directly to the sonar exposure, or to know exactly what form the response took. For example, both sperm and pilot whales potentially ceased sound production during the Heard Island feasibility test, with transmissions centered at 57 Hz and up to 220 dB re 1 μPa (Bowles et al., 1994b), although it could not be determined whether the animals ceased sound production or left the area. Killer whales in Haro Strait exhibited what were believed by some observers to be aberrant behaviors, during a time that the USS Shoup was in the vicinity and engaged in mid-frequency active sonar operations. Sound fields modeled for the USS Shoup transmissions (Fromm, 2009; National Marine Fisheries Service, 2005; U.S. Department of the Navy, 2004) estimated a mean received SPL of approximately 169 dB re 1 μPa at the location of the killer whales at the closest point of approach between the animals and the vessel (estimated received SPLs ranged from 150 to 180 dB re 1 μPa). However, attributing the observed behaviors to any one cause is problematic given there were six nearby whale watch vessels surrounding the pod, and subsequent research has demonstrated that "Southern Residents modify their behavior by increasing surface activity (breaches, tail slaps, and pectoral fin slaps) and swimming in more erratic paths when vessels are close" (National Oceanic and Atmospheric Administration, 2014).

Opportunistic sightings of several other odontocete species (i.e., bottlenose dolphins, Risso's dolphins, Pacific white-sided dolphins, common dolphins) have been observed near the Southern California Offshore Range during periods of mid-frequency active sonar. Responses included changes in or cessation of vocalizations, changes in behavior, and leaving the area, and at the highest received levels animals were not present in the area at all (Henderson et al., 2014). However, these opportunistic observations were conducted from a vessel off-range, and so any observed responses could not be attributed to the sonar with any certainty. Similarly, research on sperm whales in the Caribbean in 1983 coincided with the U.S. intervention in Grenada, where animals were presumed to scatter and leave the area because of military sonar (Watkins et al., 1985; Watkins & Schevill, 1975). They also reported similar reactions from noise generated by banging on their boat hull; therefore, it was unclear if the sperm whales were reacting to the sonar signal itself or to a potentially new unknown sound in general.

Behavioral responses by odontocetes to sonar and other transducers appear to range from no response at all to responses that could potentially lead to long-term consequences for individual animals (e.g., mother-calf separation). This is likely in part because this taxonomic group is so broad and includes some of the most sensitive species (e.g., beaked whales and harbor porpoise) as well as some of the least sensitive species (e.g., bottlenose dolphins). This is also the only group for which both field behavioral response studies and captive controlled exposure experiments have been conducted, leading to the assessment of both contextually driven responses as well as dose-based responses. This wide range in both exposure situations and individual- and species-sensitivities makes reaching general conclusions difficult. However, it does appear as though exposures in close proximity, with multiple vessels that approach the animal lead to higher-level responses in most odontocete species regardless of received level or behavioral state. In contrast, in more "real-world" exposure situations, with distant sources moving in variable directions, behavioral responses appear to be driven by behavioral state, individual experience, or species-level sensitivities. These responses may also occur more in-line with received level such that the likelihood of a response would increase with increased received levels. However, these "real-world" responses are more likely to be short term, lasting the duration of the exposure or even shorter as the animal assesses the sound and (based on prior experience or contextual cues) determines a threat is unlikely. Therefore, while odontocete behavioral responses to Navy sonar will vary across species, populations, and individuals, long-term consequences or population-level effects will depend on the frequency and duration of the disturbance and resulting behavioral response.

Responses by Specific Odontocete Species

Killer Whales

A close examination of the tag data from the Norwegian killer whales indicated that responses were mediated by behavior, signal frequency, or received sound energy. Killer whales changed their dive behavior during deep foraging dives at the onset of low-frequency active sonar (1 to 2 kHz, sweeping across frequencies) but did not change their dive behavior if they were deep diving during mid-frequency active sonar (6 to 7 kHz, sweeping across frequencies). Nor did they change their dive behavior if they were conducting shallow dives at the onset of either type of sonar (Sivle et al., 2012b). Killer whale sighting data from the same region in Norway as the behavioral response study in the 3S Project were used to compare the presence or absence of whales from other years against the period with sonar. The authors found a strong relationship between the presence of whales and the abundance of herring, and only a weak relationship between the whales and sonar activity (Kuningas et al., 2013).

Sperm Whales

The behavioral context and parameters of sonar exposure are important variables in sperm whale behavioral response to sonar as well. While there was no change in foraging (deep dive) behavior during mid-frequency active sonar, sperm whales were more responsive to low frequency active sonar (e.g., reduced dive depth, foraging, and vocalization) (Sivle et al., 2012b). In another study, (Harris et al., 2015; Quick et al., 2017) sperm whales were exposed to low frequency active pulsed active sonar at moderate source levels and high source levels, as well as continuous active sonar at moderate source levels for which the summed energy (SEL) equaled the summed energy of the high source level pulsed active sonar (Isojunno et al., 2020). Foraging behavior did not change during exposures to moderate source level sonar, but non-foraging behavior increased during exposures to high source level sonar and to the continuous active sonar, indicating that the SEL was a better predictor of response than SPL. Other studies also demonstrate that higher SELs reduced sperm whale buzzing (i.e., foraging) (Isojunno et al., 2021). The time of day of the exposure and order effects (e.g., the SEL of the previous exposure) were also important covariates in determining the amount of non-foraging behavior (Isojunno et al., 2020). Curé et al. (2021) also found that sperm whales exposed to continuous and pulsed active sonar were more likely to produce low or medium severity responses with higher cumulative SEL. Specifically, the probability of observing a low severity response increased to 0.5 at approximately 173 dB SEL and observing a medium severity response reached a probability of 0.35 at cumulative SELs between 179 and 189 dB.

One study opportunistically observed sperm whale vocalizations during an eight-day multi-platform naval exercise off the Scotian Shelf of Canada. During long bouts of sonar (various waveforms, both pulsed and continuous) lasting up to 13 consecutive hours (median and maximum SPL = 120 dB and 164 dB), sperm whales substantially reduced how often they produced clicks during sonar, indicating a decrease or cessation in foraging behavior (Stanistreet et al., 2022). Few previous studies have shown sustained changes in sperm whales, but there was an absence of sperm whale clicks for six consecutive days of sonar activity.

Melon-Headed Whales

Melon-headed whales responded to each 6−7 kHz signal with "minor transient silencing" (a brief, non-lasting period of silence), and (in a different oceanographic region) pilot whales had no apparent response (DeRuiter et al., 2013a). In a passive acoustic study using Marine Autonomous Recording Units in the Jacksonville Range Complex, the probability of detecting delphinid vocalizations (whistles, clicks, and buzzes) increased during periods of active sonar use (compared to the period prior to its use), while there was no change in the probability of detecting sperm whale clicks (Charif et al., 2015; HDR EOC, 2012).

Common Dolphins

Durban et al. (2022) observed long-beaked common dolphins via land-based observation platform coupled with a drone and multiple acoustic recorders for the first time. Vocal behavior, group cohesion, group size, and group behavior were observed before, during, and after a simulated mid-frequency sonar exposure. The number of whistles and sub-groups increased during the exposure, but the directivity and speed of the tracked subgroup was less affected.

Pilot Whales

Sonar frequency content and behavioral context are important variables in pilot whale behavioral response to sonar. While there was no change in foraging (deep dive) behavior during mid-frequency active sonar, pilot whales had fewer deep dives during low frequency active sonar (Sivle et al., 2012b). Their behavior at the onset of low frequency active sonar was especially important. If they were deep dive foraging at sonar onset, they were more likely to stop feeding and switch to shallow diving, signifying a switch to travel or rest behavior. If they were shallow diving at low frequency active sonar onset, they would not change dive type and would continue to travel or rest (Sivle et al., 2012b). In another study, pilot whales initially reduced foraging time and increased travel behavior during both low frequency active and mid-frequency active sonar exposures, but foraging increased again during subsequent exposures (Isojunno et al., 2017). This kind of behavioral reaction may indicate habituation to sonar or be driven by prey availability. Pilot whales exposed to a 38 kHz downward-facing echosounder did not change their foraging behavior during exposure periods, but their heading variance increased and fewer deep dives were conducted (Quick et al., 2017).

Diving pilot whales are also sensitive to the received level of sonar (around 170 dB re 1 µPa; Antunes et al., 2014). Cessation of foraging appeared to occur at a lower received level (145–150 dB re 1 µPa) than had been observed previously for avoidance behavior (around 170 dB re 1 µPa; Antunes et al., 2014). Pilot whales reduced breathing rates relative to their diving behavior when low frequency active sonar levels were high (180 dB re 1 µPa), but only on the first sonar exposure. On subsequent exposures their breathing rates increased (Isojunno et al., 2018), indicating a change in response tactic with additional exposures (Isojunno et al., 2018). Other behavioral responses to sonar include the synchronization of pilot whale surfacing's with sonar pulses during one exposure, possibly as a means of mitigating the sound (Wensveen et al., 2015), and pilot whales mimicking the sound of the sonar with their whistles (Alves et al., 2014).

None of the tagged pilot whales near sonar activities in Hawaii demonstrated a large-scale avoidance response as they moved on or near the range; estimated received SPLs varied from 130 to 168 dB re 1 µPa and distances from sonar sources ranged between 3.2 and 94.4 km (Baird et al., 2014; Baird et al., 2017; Baird et al., 2013). However, one pilot whale did have reduced dive rates (from 2.6 dives per hour before to 1.6 dives per hour during) and deeper dives (from a mean of 124 m to 268 m) during a period of sonar exposure. Baird et al. (2016) also tagged four short-finned pilot whales from both the resident island-associated population and from the pelagic population. The core range for the pelagic population was over 20 times larger than for the resident population, leading Baird et al. (2016) to hypothesize that that likelihood of exposure to mid-frequency active sonar, and therefore the potential for response, would be very different between the two populations. These diverse examples demonstrate that responses can be varied, are often context- and behavior-driven, and can be species- and even exposure-specific.

These results demonstrate that the behavioral state and environment of the animal mediates the likelihood of a behavioral response, as do the characteristics (e.g., frequency, energy level) of the sound source itself. The highly flexible activity time budgets observed for pilot whales, with a large amount of time spent resting at the surface, may indicate context-dependency on some behaviors, such as the presence of prey driving periods of foraging. That time may be more easily re-allocated to missed foraging opportunities (Isojunno et al., 2017).

Odontocete Responses to Other Sound Sources

Responses to Killer Whale Playbacks

The anti-predator hypothesis tested on beaked whaled was also assessed with other odontocetes. Scientists played recordings of the same mammal-eating killer whale vocalizations to pilot whales, sperm whales, Risso’s dolphins, and even other killer whales, to determine responses by both potential prey and conspecifics (Mobley, 2011; Read et al., 2022; U.S. Department of the Navy, 2011a; Watwood et al., 2012). Results varied, from no response by killer whales to an increase in group size and attraction to the source in pilot whales; rarely does a species have strong aversions as seen in beaked whales (Allen et al., 2014; Tyack et al., 2011), except for the instance of stampeding Risso’s dolphins in Southern California (Read et al., 2022). In this case study, when a group of 20 Risso’s dolphins were exposed to mammal-eating orca calls (30 sec), they became quiet, swam away at a moderate pace, and at a further distance began to porpoise and swim rapidly away (greater than 12 knots) with quick direction changes, rapid surfacings, and increased synchrony and group cohesion. Two similar sized groups of Risso’s followed suite close behind them. They slowed to 5 knots after about 1 hour and 10 km away from their original point of exposure (Read et al., 2022). Changes in foraging duration during killer whale playbacks and mid-frequency active sonar were positively correlated across four species in the 3S Norwegian studies, including long-finned pilot whale and sperm whales, suggesting that tolerance of predation risk may play a role in sensitivity to sonar disturbance (Miller et al., 2022). An alternative explanation to the anti-predator response in odontocete species that respond to sonar is a startle response. Startle responses in bottlenose dolphins occurred at moderate received levels and mid-frequencies, and the relationship between rise time and startle response was more gradual than expected for an odontocete (Gotz et al., 2020).

Responses to Acoustic Harassment and Deterrent Devices

The characteristics of deterrents and the motivation of the animal play a role in the effectiveness of acoustic harassment devices (Schakner & Blumstein, 2013). Deterrents that are strongly aversive or simulate a predator or are otherwise predictive of a threat are more likely to be effective, unless the animal habituates to the signal or learns that there is no true threat associated with the signal. While sperm whales in the Caribbean stopped vocalizing when presented with sounds from nearby acoustic pingers (Watkins & Schevill, 1975), killer whales rapidly habituated to pingers (6.5 kHz, 195 dB re 1 μPa) that were installed to stop them from depredating long lines or aquaculture enclosures. Two groups fled over 700 m away during the first exposure, but they began depredating again after the third and seventh exposures, demonstrating that acoustic harassment devices may be more successful at deterring marine mammals based on their species and context (i.e., prey availability). In some cases, net pingers may even create a “dinner bell effect,” where marine mammals have learned to associate the signal with the availability of prey (Jefferson & Curry, 1996; Schakner & Blumstein, 2013). See Table D.6-9 for a summary of findings from additional studies on these sources.

Table D.6-9: Major Findings from Studies of Acoustic Harassment Devices (AHDs) and Acoustic Deterrent Devices (ADDs) in Other Odontocetes

Species	*Major Finding*	*Supporting Studies*
Sperm whales – wild	Stopped vocalizing when pingers were present.	Watkins and Schevill (1975)
Killer whales – wild	Decreased occurrence when four AHDs deployed on salmon farms. No reduction in occurrence at adjacent location without AHDs. When AHDs removed, whale abundance near farms returned to baseline.	Morton and Symonds (2002)
Killer whales – wild	Habituated to pingers (6.5 kHz, 195 dB re 1 µPa) quickly when next to desired prey species. Fled > 700 m during the first exposure but began depredating again after the third and seventh exposures.	Tixier et al. (2014)
Bottlenose dolphins – captive	Increased surfacing, distance relative to transducer, and reduced clicks when exposed to different deterrent parameters (e.g., pulsed, and continuous tonal signals). Some acclimatization was observed during daily tests, but no habituation was observed over the full duration of the study.	Niu et al. (2012)
Bottlenose dolphins – captive	Different species had different responses to a gillnet pinger (attached to a fishing net and anchor). Bottlenose spent significantly less time in the area when it was present.	Bowles and Anderson (2012)
Bottlenose dolphins – wild	Predated significantly less on commercial fishing catches with pingers nearby (n=2) compared to catches without pingers (n=16).	Ceciarini et al. (2023)
Pacific white-sided dolphins – captive	Approached the gillnet without a pinger but avoided it when the pinger was added.	Bowles and Anderson (2012)
Commerson's dolphins – captive	Increased high-energy behavioral responses (e.g., increased swim speed, use of a refuge pool and rate of vocalization) in response to pinger. Did not habituate to pingers but instead sensitized/ demonstrated even stronger aversive behaviors over time.	Bowles and Anderson (2012)
Franciscana dolphins – wild	Avoided active banana pinger (300 ms, 50 – 120 kHz frequency modulated, 145 dB +/- 3 dB at 1 m SL) within 100 m but not at 400 m. No habituation during length of the experiment (64 days).	Paitach et al. (2022)

Notes: % = percent; > = greater than; < = less than; ADD = acoustic deterrent device; AHD = acoustic harassment device; dB = decibel; kHz = kilohertz; km = kilometer; m = meters; rms = root mean square; µPa = micropascal; n = number; SL = sound level

D.6.5.2.2 Vessel Disturbance

Behavioral responses to vessels can be caused by multiple factors. The type of vessel, approach, and speed of approach can impact the probability of a negative behavioral response (Ng & Leung, 2003). Even the way research vessels approach or move away from cetaceans can cause varied reactions in group structure and vocal behavior (Guerra et al., 2014). One research group made an effort to distinguish behavioral (e.g., nursing and resting) reactions to vessel noise from vessel presence (Arranz et al., 2021). The short-finned pilot whale mother-calf pairs were approached by the same tour boat with either two quiet electric or noisy petrol engines installed. Approach speed, distance, and vessel features other than engine noise remained the same between the two experimental conditions. While mother pilot whales rested less, and calves nursed less, in response to both types of boat engines compared to control conditions, only the louder petrol engine caused significant impacts (29 percent and 81 percent, respectively) to these behaviors. However, in most field studies the influence of vessel sound exposure cannot be decoupled from the physical presence of a surface vessel, complicating interpretations of the relative contribution of each stimulus to the response. This section will cover both aspects (vessel noise and presence) in publications that specifically aim to target response to noise.

Most odontocetes react neutrally to vessels, although both avoidance and attraction behavior have been observed (Hewitt, 1985; Würsig et al., 1998). Würsig et al. (1998) found that Kogia whales and beaked whales were the most sensitive species to vessels and reacted by avoiding marine mammal survey vessels in 73 percent of sightings, more than any other odontocetes. Avoidance reactions include a decrease in resting behavior or change in travel direction (Bejder et al., 2006a). Incidents of attraction include common, rough-toothed, and bottlenose dolphins bow riding and jumping in the wake of a vessel (Norris & Prescott, 1961; Ritter, 2002; Shane et al., 1986; Würsig et al., 1998). Hudson Bay belugas spent most of their time interacting favorably (e.g., glided under, rubbed against, and swam along) with nearby seasonal tour boats that practiced sustainable whale watching practices (Westdal et al., 2023). A study of vessel reactions by dolphin communities in the eastern tropical Pacific found that populations that were often the target of tuna purse-seine fisheries (spotted, spinner, and common dolphins) show evasive behavior when approached; however, populations that live closer to shore (within 100 NM; coastal spotted and bottlenose dolphins), and are not set on by purse-seine fisheries, tend to be attracted to vessels (Archer et al., 2010). Reactions to vessels may also be context-specific. In some studies, the presence of vessels has been shown to interrupt feeding behavior in delphinids (Meissner et al., 2015; Pirotta et al., 2015b). However, in an important foraging area, bottlenose dolphins (a comparatively less sensitive species of odontocete) may continue to forage and socialize even while constantly exposed to high vessel traffic (Mills et al., 2023).

Smaller vessels (e.g., research and tour boats) generate more noise in higher frequency bands, are more likely to approach odontocetes directly and spend more time near an individual whale. Tour boat activity can cause short-term (Carrera et al., 2008) and longer term or repetitive displacement of dolphins due to chronic vessel noise (Haviland-Howell et al., 2007). Delphinid behavioral states also change in the presence of tourist boats that often approach animals, with travel and resting increasing, foraging and social behavior decreasing, and animals reducing the space between each other (e.g., "group dispersal") (Cecchetti et al., 2017; Clarkson et al., 2020; Kassamali-Fox et al., 2020; Meissner et al., 2015). Most bottlenose dolphin studies on the behavioral reactions to vessel traffic have documented at least short-term changes in behavior, activities, or vocalization patterns when vessels are nearby (Acevedo, 1991; Arcangeli & Crosti, 2009; Berrow & Holmes, 1999; Fumagalli et al., 2018; Gregory & Rowden, 2001; Janik & Thompson, 1996; Lusseau, 2004; Marega et al., 2018; Mattson et al., 2005; Perez-Ortega et al., 2021; Puszka et al., 2021; Scarpaci et al., 2000). Table D.6-10 provides some examples of behavioral responses to different vessel types, with a focus on small recreational vessels and tour boats.

Northern and Southern resident killer whales are sought after by numerous small whale watching vessels in the Pacific Northwest and live in a high traffic area with many different types of vessels. For many years (1998 – 2012) these populations had an average of 20 vessels within 0.5 miles of their location during daytime hours every month (Clark, 2015; Eisenhardt, 2014; Erbe et al., 2014). These vessels had source levels that ranged from 145 to 169 dB re 1 µPa and produced broadband noise up to 96 kHz. Received levels of vessel noise did not decrease with the implementation of new policy on vessel distance. Instead noise levels increased as more and faster moving vessels were introduced (Holt et al., 2017). These noise levels can result in behavioral disturbance, interfere with communication, and affect the killer whales' hearing capabilities via masking (Erbe, 2002; Veirs et al., 2015). Factors other than vessel noise that contribute to the severity of killer whales behavioral response to vessels include seasonal data (e.g., year and month), a whale's prior experience with vessels (e.g., age and sex), and the number of other vessels present (Williams et al., 2014a).

Table D.6-10: Examples of Behavioral Responses to Vessels by Odontocetes

Species	***Study***	***Boat type***	***Behavioral Change***				
			Feeding or foraging	***Group dispersal***	***Resting***	***Diving duration***	***Traveling or fleeing***
Common dolphins	Stockin et al. (2008)	Tour	↓		↓		
Bottlenose dolphin	Steckenreuter et al. (2011)	Tour	↓	↓			↑ when approached to 50 m (vs 150 m)
Bottlenose dolphin	Toro et al. (2021)	Tour					↑ (avoid vessel and ↓ surface activity)
Indo-Pacific humpback dolphins	Ng and Leung (2003)	Fishing	↑				
Indo-Pacific humpback dolphins	Ng and Leung (2003)	High-speed				↑ in heavy or oncoming traffic	↑
Killer whales	Kruse (1991); Lusseau et al. (2009); Trites and Bain (2000); Williams et al. (2002a); Williams et al. (2009); Williams et al. (2002b)	Tour	↓ when boats < 100 m				↑ when boats < 100 m
Killer whales (esp. females)	Holt et al. (2021)	Tour	↓ (stop) when boats < 400 m				↑ (start) when boats < 400 m
Pilot whales	Arranz et al. (2021)	Tour	↓ nursing		↓		
Beluga whales	Martin et al. (2023b)	Various[1]				Δ dive behavior	↑ speed (< 13 km) and Δ bearing
Beluga whales	Westdal et al. (2023)	Tour	↓				↓ (↑ interactions with boat < 25 m away)
Harbor porpoises	Frankish et al. (2023)	Large tankers				↑ depth at night	↑ distance during day, especially < 300 m

[1] Various ships = tankers, cargo ships, research vessels, fishing, tug boats

Notes: ↑ = increase; ↓ = decrease; < = less than; Δ = change in; km = kilometer; m = meters

Sperm whales generally only react to vessels approaching within several hundred meters. Some individuals are prone to avoidance behavior, such as quick diving (Magalhães et al., 2002; Würsig et al., 1998) or less time spent at the surface (Isojunno & Miller, 2015). When vessels were present, sperm whales were quicker to emit their first click after diving (Richter et al., 2006). Sperm whales have also been observed reducing clicks while a vessel passes by, as well as up to a half hour after the vessel passed (Azzara et al., 2013). It is unknown whether these whales left the area, ceased to click, or surfaced during this period. However, some of the reduction in click detections may be due to masking of the clicks by the vessel noise, particularly during the closest point of approach.

Little information is available on the behavioral impacts of vessel disturbance on beaked whales (Cox et al., 2006), but it seems like most beaked whales react negatively to vessels with abrupt diving and other avoidance maneuvers (Würsig et al., 1998). There is some evidence that suggests beaked whales respond to all anthropogenic noise (including vessel and sonar) at similar sound levels (Aguilar de Soto et al., 2006; Tyack et al., 2011; Tyack, 2009). A vocalizing goose-beaked whale was disrupted from foraging when a large, noisy vessel passed, which suggests that some types of vessel traffic may disturb foraging beaked whales (Aguilar de Soto et al., 2006). Exposure to broadband ship noise (received level of 135 dB re 1 µPa) does not change the duration of whale foraging dives, but may restrict the movement of a group (Pirotta et al., 2012).

Small dolphins and porpoises are also sensitive to vessel noise. Both finless porpoises (Li et al., 2008) and harbor porpoises (Polacheck & Thorpe, 1990) routinely avoid and swim away from large motorized vessels. A study in the Baltic Sea found that harbor porpoises were more likely to horizontally avoid large commercial ships during the day but vertically avoid them at night (Frankish et al., 2023). Near ships, harbor porpoises respond with fewer clicks (Sairanen, 2014), decreased feeding and behavioral bout durations in general (Akkaya Bas et al., 2017). Specifically, foraging harbor porpoises have fewer prey capture attempts and have disrupted foraging when vessels pass closely and noise levels are higher (Wisniewska et al., 2018). A resident population of harbor porpoise that was regularly near vessel traffic (10 m to 1 km away) had no response in 74 percent of interactions and an avoidance response in 26 percent of interactions. Most avoidance responses were observed in groups of 1 to 2 animals, and were the result of fast-moving or steady plane-hulling motorized vessels (Oakley et al., 2017). Larger groups reacted less often, and few responses were observed to non-motorized or stationary vessels (Oakley et al., 2017). Another study of responses to fast moving vessels found that when the vessels were within 50 m, harbor porpoises had an 80 percent probability of changing their swimming direction, but only a 40 percent probability when vessels were beyond 400 m (Akkaya Bas et al., 2017). A study on harbor porpoise in the Baltic Sea found that porpoises were most likely to avoid close ships (< 300 m), but that 5-10% of the time they would also respond to ships > 2 km away, signifying that were not just attuning to ship presence but ship noise as well (Frankish et al., 2023). Although most vessel noise is constrained to lower frequencies below 1 kHz, at close ranges, vessel noise can extend into mid- and high frequencies (into the tens of kHz) (Hermannsen et al., 2014; Li et al., 2015). These frequencies are what harbor porpoises are likely responding to; the mean M-weighted received SPL threshold for a response at these frequencies is 123 dB re 1 µPa (Dyndo et al., 2015). Hermannsen et al. (2019) estimated that noise in the 16 kHz frequency band resulting from small recreational vessels not equipped with an Automatic Identification System and therefore not included in most vessel noise impact models could be elevated up to 124 dB re 1 µPa and raise ambient levels up to 51 dB; these higher levels were associated with vessel speed and range. These authors determined that the threshold levels found by Dyndo et al. (2015) and Wisniewska et al. (2018) were exceeded by 49 to 85 percent of events with high levels of boat noise, and that recreational vessel noise in the 16 kHz band could cause behavioral responses in harbor porpoises.

Naïve populations of bottlenose dolphins (exposed to lower levels of vessel traffic) had stronger and longer lasting reactions to vessel approaches (Bejder et al., 2006b). Fewer reactions in populations of dolphins regularly subjected to high levels of vessel traffic could be a sign of habituation, or it could be that the more sensitive animals in this population previously abandoned the area of higher human activity.

Odontocetes have been shown to make short-term changes to their vocalizations as an immediate response to vessel noise (see Table D.6-4). For example, bottlenose dolphins in Portuguese waters decrease their call rates and change the frequency parameters of whistles in the presence of boats, while dolphin groups with calves increase their whistle rates when tourist boats are within 200 m and when the boats increase their speed (Guerra et al., 2014). Foraging Lahille's bottlenose dolphins in Brazil increase the duration of their whistles when there is an increase in the speed or number of boats within 250 m; they also increase the frequency parameters of their whistles, especially when group size or calf presence increased. Likewise, modification of multiple vocalization parameters was shown in belugas residing in an area known for high levels of commercial traffic. These animals decreased their call rate, increased certain types of calls, and shifted upward in frequency content in the presence of small vessel noise (Lesage et al., 1999). An increase in the amplitude of vocalizations (Lombard effect) has also been observed when ships were present (Scheifele et al., 2005).

Long-term modifications to vocalizations may be indicative of a learned response to chronic noise, or of a genetic or physiological shift in the populations. This type of change has been observed in killer whales off the northwestern coast of the United States between 1973 and 2003. This population increased the duration of primary calls once a threshold in observed vessel density (e.g., whale watching) was reached, which is suggested as being a long-term response to increased masking noise produced by the vessels (Foote et al., 2004).

The long-term and cumulative implications of vessel noise on odontocetes is largely unknown (National Academies of Sciences Engineering and Medicine, 2017; National Marine Fisheries Service, 2007) although some long-term consequences have been reported (Lusseau & Bejder, 2007). Repeated exposure to acoustic and other anthropogenic stimuli has been studied in several cases, especially as related to vessel traffic and whale watching. Many authors speculate that repeated interruption of foraging behaviors could lead to long-term implications for odontocete populations (Stockin et al., 2008), and in many contexts their localized and coastal home ranges do make them less resilient than mysticetes to this kind of chronic stressor (Southall et al., 2021). The long-term and cumulative implications of ship sound on odontocetes is largely unknown (National Academies of Sciences Engineering and Medicine, 2017; National Marine Fisheries Service, 2007) although some long-term consequences have been reported (Lusseau & Bejder, 2007). Repeated exposure to acoustic and other anthropogenic stimuli has been studied in several cases, especially as related to vessel traffic and whale watching. Many authors speculate that repeated interruption of foraging behaviors could lead to long-term implications for odontocete populations (Stockin et al., 2008), and in many contexts their localized and coastal home ranges do make them less resilient than mysticetes to this kind of chronic stressor (Southall et al., 2021).

Like mysticetes, odontocete responses to vessel noise are varied. Although many odontocete species seem to be more sensitive to vessel presence and noise, these two factors are difficult to tease apart. Some species (e.g., killer whales and porpoises) may be sensitized to vessels and respond at further distances and lower received levels than other delphinids. In contrast, other species (e.g., bottlenose, spotted, spinner, Clymene, and Pacific white sided dolphins) will approach vessels to bow ride, indicating either that these species are less sensitive to vessels or that the behavioral drive to bow ride supersedes any impact of the associated noise (Würsig et al., 1998). With these broad and disparate responses, it is difficult to assess the impacts of vessel noise on odontocetes.

D.6.5.2.3 Aircraft Disturbance

Behavioral responses to aircraft can be caused by multiple factors. It is difficult for researchers and analysts to separate the effects of aircraft noise and aircraft presence; therefore, this section will cover both aspects in publications that specifically aim to target response to noise.

Variable responses to aircraft have been observed in toothed whales, though overall little change in behavior has been observed during flyovers. Some toothed whales dove, slapped the water with their flukes or flippers, or swam away from the direction of the aircraft during overflights; others did not visibly react (Richardson et al., 1995b). Würsig et al. (1998) found that beaked whales were the most sensitive cetacean and reacted by avoiding marine mammal survey aircraft in 89 percent of sightings and at more than twice the rate as Kogia whales, which was the next most reactive of the odontocetes in 39 percent of sightings. These are the same species that were sensitive to vessel traffic.

During standard marine mammal surveys at an altitude of 750 ft., some sperm whales remained on or near the surface the entire time the aircraft was in the vicinity, while others dove immediately or a few minutes after being sighted. Other authors have corroborated the variability in sperm whales' reactions to fixed-wing aircraft or helicopters (Green et al., 1992; Richter et al., 2006; Richter et al., 2003; Smultea et al., 2008; Würsig et al., 1998). Whale watching aircraft (fixed-wing airplanes and helicopters) apparently caused sperm whales to turn more sharply but did not affect blow interval, surface time, time to first click, or the frequency of aerial behavior (Richter et al., 2003).

A group of sperm whales responded to a circling fixed-wing aircraft (altitude of 800 to 1,100 ft.) by moving closer together and forming a defensive fan-shaped semicircle, with their heads facing outward. Several individuals in the group turned on their sides, apparently to look up toward the aircraft (Smultea et al., 2008). Smaller delphinids generally react to overflights either neutrally or with a startle response (Würsig et al., 1998). A change in travel direction was noted in a group of pilot whales as the a fixed-wing aircraft circled while conducting monitoring (HDR, 2011). No changes in group cohesion or orientation behavior were observed for groups of Risso's dolphins, common dolphins, or killer whales when a survey airplane flew at altitudes of 213–610 m, but this may be due to the plane maintaining lateral distances greater than 500 m in all (Smultea & Lomac-MacNair, 2016).

Helicopters may elicit a greater reaction in odontocetes. Beluga whales reacted to helicopter overflights by diving, breaching, changing direction or behavior, and altering breathing patterns to a greater extent than mysticetes in the same area (Patenaude et al., 2002). These reactions increased in frequency as the altitude of the helicopter dropped below 150 m. Sperm whales showed no reaction to a helicopter until they encountered the downdrafts from the rotors (Richardson et al., 1995b).

Much like mysticetes, odontocetes have demonstrated no responses to unmanned aerial systems at altitudes over 30 m. For example, Durban et al. (2015) conducted photogrammetry studies of killer whales using a small helicopter flown 35 to 40 m above the animals with no disturbance noted. However, odontocete responses have been reported with use at reduced altitudes. St. Lawrence belugas responded to drones below 23 m with evasive dive responses; their alert surface active reactions (e.g., tail slap) also increased in larger groups or while socializing (Aubin et al., 2023). These impacts may be species-specific, and could be due either to noise or the shadows created by the vehicle (Smith et al., 2016). Bottlenose dolphins responded to a small portion of unmanned aerial vehicles by briefly orienting when the vehicle was relatively close (10 to 30 m high), but in most cases did not respond at all (Ramos et al., 2018).

D.6.5.2.4 Impulsive Noise

Impulsive signals, particularly at close range, have a rapid rise time and higher instantaneous peak pressure than other signal types, making them more likely to cause startle responses or avoidance responses. However, at long distances the rise time increases as the signal duration lengthens (similar to a "ringing" sound), making the impulsive signal more similar to a non-impulsive signal (Hastie et al., 2019; Martin et al., 2020). Behavioral reactions from explosive sounds are likely to be similar to reactions studied for other impulsive sounds, such as those produced by air guns and impact pile driving. Data on behavioral responses to impulsive sound sources are limited across all marine mammal groups, with only a few studies available for mysticetes and odontocetes. Most data have come from seismic surveys that occur over long durations (e.g., on the order of days to weeks), and typically utilize large multi-air gun arrays that fire repeatedly. While seismic data provide the best available science for assessing behavioral responses to impulsive sounds by marine mammals, it is likely that these responses represent a worst-case scenario compared to responses to explosives used in Navy activities, which would typically consist of single impulses or a cluster of impulses, rather than long-duration, repeated impulses.

Few data are available on odontocete responses to impulsive sound sources, with only a few studies on responses to seismic surveys, pile driving and construction activity available. However, odontocetes appear to be less sensitive to impulsive sound than mysticetes, with responses occurring at much closer distances. This may be due to the predominance of low-frequency sound associated with these sources that propagates long distances and overlaps with the range of best hearing for mysticetes but is below that range for odontocetes. The exception to this is the harbor porpoise, which has been shown to be highly sensitive to most sound sources, avoiding both stationary (e.g., pile driving) and moving (e.g., seismic survey vessels) impulsive sound sources out to approximately 20 km (e.g., Haelters et al., 2014; Pirotta et al., 2014). However, even this response is short term, with porpoises returning to the area within hours after the cessation of the noise.

There are even fewer direct observations of behavioral reactions from marine mammals due to exposure to explosive sounds. Lammers et al. (2017) recorded dolphin detections near naval mine neutralization exercises and found that although the immediate response (within 30 seconds of the explosion) was an increase in whistles relative to the 30 seconds before the explosion, there was a reduction in daytime acoustic activity during the day of and the day after the exercise within 6 km. However, the nighttime activity did not seem to be different than that prior to the exercise, and two days after there appeared to be an increase in daytime acoustic activity, indicating a rapid return to the area by the dolphins (Lammers et al., 2017).

Ferguson et al. (2006) and Miller et al. (2009) tagged and monitored eight sperm whales in the Gulf of Mexico exposed to seismic air gun surveys. Sound sources were from approximately 2 to 7 NM away from the whales, and received levels were as high as 162 dB SPL re 1 µPa (Madsen et al., 2006). The whales showed no horizontal avoidance, however one whale rested at the water's surface for an extended period of time until air guns ceased firing (Miller et al., 2009). While the remaining whales continued to execute foraging dives throughout exposure, tag data suggested there may have been subtle effects of noise on foraging behavior (Miller et al., 2009). Similarly, Weir (2008) observed that seismic air gun surveys along the Angolan coast did not significantly reduce the encounter rate of sperm whales during the 10-month survey period, nor were avoidance behaviors to air gun impulsive sounds observed. In contrast, Atlantic spotted dolphins did show a significant, short-term avoidance response to air gun impulses within approximately 1 km of the source (Weir, 2008). The dolphins were observed at greater distances from the vessel when the air gun was in use, and when the air gun was not in use they readily approached the vessel to bow ride. Kavanagh et al. (2019) also found that toothed whales were more averse to active air guns, as sightings of several species of odontocetes were reduced by 53 and 29 percent during active and inactive phases of seismic surveys, respectively, compared to baseline surveys. Narwhals exposed to air guns in an Arctic fjord were even more sensitive (Heide-Jorgensen et al., 2021). Even though small and large air gun sources reached ambient noise levels around 3 and 10 km (air gun source levels = 231 and 241 dB re 1 µPa at 1 m), respectively, narwhals still changed their swimming direction away from the source and towards shore when seismic vessels were in line of sight over 11 km away. Swimming speed was context-dependent; whales usually increased speed in the presence of vessels but would reduce speed ("freeze") in response to closely approaching air gun pulses. Other behaviors, like feeding, also ceased when the active air gun noise was less than 10 km away, although received SELs were below 130 dB re 1 µPa^2 s for either air gun at this distance. Due to study research methods and criteria, even these long-distance reactions of narwhals may be conservatively estimating narwhals' range to behavioral response.

Captive bottlenose dolphins sometimes vocalized or were reluctant to return to the test station after exposure to single impulses from a seismic water gun (Finneran et al., 2002). When exposed to multiple impulses from a seismic air gun, some dolphins turned their heads away from the sound source just before the impulse, showing that they could anticipate the timing of the impulses and perhaps reduce the received level (Finneran et al., 2015). During construction (including the blasting of old bastions) of a bridge over a waterway commonly used by the Tampa Bay, Florida stock of bottlenose dolphins, the use of the area by females decreased while males displayed high site fidelity and continued using the area, perhaps indicating differential habitat uses between the sexes (Weaver, 2015).

Harbor porpoises seem to have an avoidance response to seismic surveys. A study using aerial surveys and C-PODs (an autonomous recording device that counts odontocete clicks) found that harbor porpoises appeared to leave the area of the survey, and decreased their foraging activity within 5 to 10 km, as evidenced by both a decrease in vocalizations near the survey and an increase in vocalizations at a distance (Pirotta et al., 2014; Thompson et al., 2013a). However, the animals returned within a day after the air gun operation ceased, and the decrease in occurrence over the survey period was small relative to the observed natural seasonal decrease compared to the previous year.

Harbor porpoises have a similar response to pile driving as well. A similar study using C-PODs at two offshore windfarms to examine differences in harbor porpoises presence and foraging activity between baseline (102 to 104 dB) and construction periods (155 to 161 dB) found decreased presence (8 to 17 percent) and foraging (41 to 62 percent) during construction periods. More porpoises were displaced up to 12 km away from pile driving and 4 km from construction vessels (Benhemma-Le Gall et al., 2021). A number of studies also found strong avoidance responses by harbor porpoises out to 20 km during pile driving; however, animals returned to the area after the activity stopped (Brandt et al., 2011; Dähne et al., 2014; Haelters et al., 2014; Thompson et al., 2010; Tougaard et al., 2005; Tougaard et al., 2009). When bubble curtains were deployed around pile driving, the avoidance distance appeared to be reduced to half that distance (12 km), and the response only lasted about five hours rather than a day before the animals returned to the area (Dähne et al., 2017).

However, not all harbor porpoise behavioral response studies ended in habitat displacement. Bergström et al. (2014) found that although there was a high likelihood of acoustic disturbance during wind farm construction (including pile driving), the impact was short term. In another pile driving study, Graham et al. (2019) found that the distance at which behavioral responses were probable decreased over the course of the construction project, suggesting habituation to pile-driving noise in the local harbor porpoise population. When C-PODs were placed near oil and gas platforms and control sites 15 km away, there was a dose-response effect with the lowest amount of porpoise activity closest to the seismic vessel ($SEL_{single\ shot}$ = 155 dB re 1 µPa^{2}s) and then increasing porpoise activity out to 8 to 12 km, outside of which levels were similar to baseline. Distance to the seismic vessel was a better model predictor of porpoise activity than sound level. Despite these smaller-scale responses, a large-scale response was not detected, and overall porpoise activity in the seismic area was similar to the control stations; this may indicate that the porpoises were moving around the seismic area to avoid the ship, but not leaving the area entirely (Sarnocińska et al., 2020).

According to a 10-year boat-based line-transect survey in an area which included the preconstruction, construction, and postconstruction of offshore wind farm, harbor porpoises were observed throughout the area during all three phases. However, they were not detected within the footprint of the windfarm and were overall less frequent throughout the study area during the construction phase. They returned after the construction was completed at a slightly higher level than in the preconstruction phase. There was no large-scale displacement of harbor porpoises during construction, and their avoidance behavior only occurred out to about 18 km, in contrast to the approximately 25 km avoidance distance found in other windfarm construction and pile driving monitoring efforts.

A five-year study (2015-2020) found that harbor porpoise detections significantly decreased at the beginning of a pile driving project (SEL at 750 m was 160 to 164 dB re 1 μPa^2s) for an oil and gas platform, but detections appeared to return to baseline levels within five months (Todd et al., 2022). The lack of significant trend over years indicated that porpoises did not experience habitat displacement for the entire five-year period. However, it is important to note that the oil and gas platform construction did not take five years, and the type of sources changed over this five-year period.

When exposing a captive harbor porpoise to impact pile driving sounds, Kastelein et al. (2013b) found that above 136 dB re 1 μPa (zero-to-peak) the animal's respiration rates increased, and at higher levels it jumped more frequently. Swim speed, respiration rate, distance from the transducer, and jumping may also increase in response to pile driving sounds, as long as those sounds have higher frequencies present (i.e., above 6 kHz) (Kastelein et al., 2022).

The occurrence of bottlenose dolphins and harbor porpoises over different area and time scales were assessed with and without impact and vibratory pile driving. While there were fewer hours with bottlenose dolphin detections and reduced detection durations within the pile driving area and increased detection durations outside the area, the magnitude of the effects were small, and the reduced harbor porpoise encounter duration was attributed to seasonal changes outside the influence of the pile driving. However, received levels in this area were lower due to propagation effects than in the other areas described above, which may have led to the lack of or reduced response.

Odontocete behavioral responses to impulsive sound sources are likely species- and context-dependent, with most species demonstrating little to no apparent response. Responses might be expected close to a noise source, under specific behavioral conditions such as females with offspring, or for sensitive species such as harbor porpoises.

D.6.5.3 Behavioral Reactions of Pinnipeds

The pinnipeds consist of phocids ("earless" seals) and otariids (sea lions and fur seals), as well as walruses. The below summary will address best available science regarding responses by phocids, followed by otariids. Although not all species are present in the Study Area, information on their responses to acoustic stressors augment the limited knowledge of behavioral responses by pinnipeds.

D.6.5.3.1 Sonar and Other Transducers

Studies of pinniped behavioral responses to sonar and other transducers are limited. Observed responses seem to be mediated by the contextual factors of the exposure, including the characteristics of the signal (e.g., Hastie et al., 2014) and the behavioral state of the animal. However, all studies of pinniped behavioral response to sonars (not including fisheries deterrents) have been conducted in captivity, so application to real-world exposure situations must be done with caution. Based on exposures to other sound sources in the wild (e.g., impulsive sounds and vessels), pinnipeds may only respond strongly to Navy sonar that is near or approaching.

Different responses displayed by captive and wild phocid seals to sound judged to be "unpleasant" or threatening have been reported, including habituation by captive seals (they did not avoid the sound) and avoidance behavior by wild seals (Götz & Janik, 2010). Captive seals received food (reinforcement) during sound playback, while wild seals were exposed opportunistically. These results indicate that motivational state (e.g., reinforcement via food acquisition) can be a factor in whether an animal tolerates or habituates to novel or unpleasant sounds. Another study found that captive hooded seals reacted to 1 to 7 kHz sonar signals, in part with displacement (i.e., avoidance) to the areas of least SPL, at levels between 160 and 170 dB re 1 µPa (Kvadsheim et al., 2010b); however, the animals adapted to the sound and did not show the same avoidance behavior upon subsequent exposures. Captive harbor seals responded differently to three signals at 25 kHz with different waveform characteristics and duty cycles. The seals responded to the frequency modulated signal at received levels over 137 dB re 1 µPa by hauling out more, swimming faster, and raising their heads or jumping out of the water, but did not respond to the continuous wave or combination signals at any received level (up to 156 dB re 1 µPa) (Kastelein et al., 2015c). Low-frequency signals of the Acoustic Thermometry of Ocean Climate sound source centered at 75 Hz, with received levels between 118 and 137 dB re 1 µPa, were not found to overtly affect elephant seal dives (Costa et al., 2003). However, they did produce subtle effects that varied in direction and degree among the individual seals, again illustrating the equivocal nature of behavioral effects and consequent difficulty in defining and predicting them.

Harbor seals exposed to seal scarers (i.e., acoustic harassment devices used to deter seals from fishing nets or salmon farms) did not respond in any biologically significant way in several studies (Kastelein et al., 2015b; Mikkelsen et al., 2017; Morton & Symonds, 2002), but did demonstrate minor responses by occasionally hauling out at 128 to 138 dB re 1 µPa (Kastelein et al., 2015b). Pingers have also been used to deter marine mammals from fishing nets. One study exposed species to novel objects, including a fishing net and anchor with line, both with and without a gillnet pinger. Captive harbor seals, California sea lions and Northern elephant seals avoided a fishing net and anchor with line with a gillnet pinger but did not avoid the same net without a pinger (Bowles & Anderson, 2012). In some cases, pingers on nets lead to the “dinner bell effect,” where the pinger becomes an attractant rather than a deterrent (Carretta & Barlow, 2011).

To better understand otariid responses to tactical mid-frequency sonar, captive California sea lions were exposed to mid-frequency sonar at various received levels (125 to 185 dB re 1 µPa) during a repetitive task (Houser et al., 2013a). Behavioral responses included a refusal to participate, hauling out, an increase in respiration rate, and an increase in the time spent submerged. Young animals (less than two years old) were more likely to respond than older animals. Dose-response curves were developed both including and excluding those young animals. Most responses below 155 dB re 1 µPa were changes in respiration, whereas over 170 dB re 1 µPa more severe responses began to occur (such as hauling out or refusing to participate); many of the most severe responses came from the younger animals. In another study investigating potential deterrent sounds, captive Steller sea lions were exposed to a variety of sounds for two minutes, at a maximum source level of 165 dB re 1 µPa for non-impulsive sounds (Akamatsu et al., 1996). Killer whale vocalizations (whether these were from fish-eating or mammal-eating killer whales is not stated), 1-4 kHz sweeps, and low source level impulses were least effective at causing adults to respond by hauling out, whereas juveniles were more likely to haul out in response to sweeps and low-level impulses. The intermittent pure tone at 8 kHz was most likely to elicit responses in adults and juveniles, although not consistently. The addition of prey items to the test pool greatly reduced the likelihood of hauling out during a sound exposure.

D.6.5.3.2 Vessel Disturbance

Behavioral responses to vessels can be caused my multiple factors. It is difficult for researchers and analysts to separate the effects of vessel noise and vessel presence; therefore, this section will cover both aspects in publications that specifically aim to target response to noise. Pinniped reactions to vessels are variable and reports include a wide spectrum of possibilities including vigilance, avoidance, alerting, and reduced time feeding, resting, or nursing (Martin et al., 2023a; Martin et al., 2022; Mikkelsen et al., 2019; Richardson et al., 1995b). On the opposite end of the spectrum, some pinnipeds demonstrate in-water attraction or a lack of significant reaction when hauled out, suggesting habituation to or tolerance of vessels (Richardson et al., 1995b). Specific case reports in Richardson et al. (1995b) vary based on factors such as routine anthropogenic activity, distance from the vessel, engine type, wind direction, and ongoing subsistence hunting. As with reactions to sound reviewed by Southall et al. (2007), pinniped responses to vessels are affected by the context of the situation and by the animal's experience. Social variables such as animal density and reproductive context may play a role in degree of responsiveness as well. For example, Cape fur seals were less responsive to vessel noise in sites with lower seal abundances compared to a site with a large breeding colony (Martin et al., 2023a).

Increasing numbers of vessels in coastal areas have reduced haul-out time and increase heart rate for harbor seals in certain contexts. The most harbor seal haul outs on Alaskan tidewater glaciers occur during pupping season, and the presence of any vessel reduced this haul out time, with cruise ships and other large vessels having the strongest effect (Blundell & Pendleton, 2015). Another study in Alaska found that hauled out harbor seals were more likely to flush and enter the water when cruise ships approached the ice within 500 m, and were four times more likely to flush when the cruise ship approaches within 100 m (Jansen et al., 2010). Harbor seal heart rates increased when vessels were present during haul out periods and increased further when vessels approached and animals re-entered the water (Karpovich et al., 2015). Harbor seals responded more to vessels passing by haul out sites in areas with less overall vessel activity, and the model best predicting their flushing behavior included the number of boats, type of boats, and distance to boats. More flushing occurred to non-motorized vessels (e.g., kayaks), likely because they tended to occur in groups rather than as single vessels, and tended to pass closer (25 to 184 m) to the haul out sites than motorized vessels (55 to 591 m) (Cates & Acevedo-Gutiérrez, 2017).

Other behaviors not associated with haul-out time and flushing are impacted by vessel disturbance as well. Long-term biologgers (DTAGs) were attached to harbor seals and grey seals to opportunistically examine behaviors over several weeks (Mikkelsen et al., 2019). The data showed that seals were exposed to vessel noise between 2.2 and 20.5 percent of their time in water. Potential responses to vessels, coinciding with increasing or peak vessel noise on the tags, included interruption of resting and foraging behaviors. Although there were no behavioral differences between hauled-out wild cape fur seals exposed to low (60-64 dB re 20 µPa RMS SPL), medium (64-70 dB) and high-level (70-80 dB) vessel noise playbacks, mother-pup pairs spent less time nursing (15-to 31 percent) and more time awake (13 to 26 percent), vigilant (7 to 31 percent), and mobile (2to 4 percent) during boat noise conditions compared to control conditions (Martin et al., 2022).

Impact to pinnipeds may differ based on the location or species, as some populations may be more tolerant to vessel disturbance or have a lower degree of overlap with boat traffic. Walrus reaction to vessel noise in the Arctic remains inconclusive (Taylor et al., 2023). Grey seal reactions to increasing vessel traffic off Ireland's coast in association with construction activities suggest that the number of vessels had an indeterminate effect on the seals' presence (Anderwald et al., 2013). Modeling of harbor seals and grey seals in the UK found that they were most likely to overlap with vessel traffic within 50 km of the coast, which included around half of the seals' Special Areas of Conservation (Jones et al. (2017). While there was no evidence of reduced population size in any of these high overlap areas, estimated received levels of shipping noise and maximum daily M-weighted cumulative SEL values ranged from 170 to 189 dB, with the upper confidence intervals of those estimates sometimes exceeding TTS values.

D.6.5.3.3 Aircraft Disturbance

Richardson et al. (1995b) noted that responsiveness to aircraft overflights generally was dependent on the range (altitude and distance) of the aircraft, the abruptness of the associated aircraft sound, and life cycle stage (breeding, molting, etc.). Pinnipeds may startle, orient towards the sound source, increase vigilance, or briefly re-enter the water, but, in general, they are unresponsive to overflights and typically remain hauled out or immediately return to their haul out location (Blackwell et al., 2004; Gjertz & Børset, 1992). Reactions of walruses on land varied in severity and included minor head raising at a distance of 2.5 km, orienting toward, or entering the water at less than 150 m and 1.3 km in altitude, to full flight reactions at horizontal ranges of less than 1 km at altitudes as high as 1,000–1,500 m (Richardson et al., 1995b).

Helicopters are used in studies of several species of seals hauled out and are considered an effective means of observation (Bester et al., 2002; Gjertz & Børset, 1992), although they have been known to elicit behavioral reactions such as fleeing (Hoover, 1988). For California sea lions and Steller sea lions at a rocky haulout off Crescent City in northern California, helicopter approaches to landing sites typically caused the most severe response of diving into the water (National Oceanic and Atmospheric Administration, 2010). Responses were also dependent on the species, with Steller sea lions being more sensitive and California sea lions more tolerant. Depending on the time between subsequent approaches, animals hauled out in between and fewer animals reacted upon subsequent exposures (National Oceanic and Atmospheric Administration, 2010).

Pinnipeds may respond to unmanned aerial systems, especially those flying at low altitudes, due to their possible resemblance to predatorial birds (Smith et al., 2016), which could lead to flushing behavior (Olson, 2013). Responses may also vary by species, age class, behavior, and habituation to other anthropogenic noise, as well as by the type, size, and configuration of unmanned aerial vehicle used (Pomeroy et al., 2015). However, in general pinnipeds have demonstrated little to no response to unmanned aerial systems, with some orienting towards the vehicle, other alerting behavior, or short-term flushing possible (Laborie et al., 2021; Moreland et al., 2015; Sweeney et al., 2015).

D.6.5.3.4 Impulsive Noise

Pinnipeds may be the least sensitive marine mammal group to noise sources in this document. Some species may be more sensitive than others and are likely to only respond to loud impulsive sound sources at close ranges by startling, jumping into the water when hauled out, or ceasing foraging, but only for brief periods before returning to their previous behavior (e.g., Southall et al., 2007). Pinnipeds may even experience hearing effects before exhibiting a behavioral response (Southall et al., 2007). A review of behavioral reactions by pinnipeds to impulsive noise can be found in Richardson et al. (1995b) and Southall et al. (2007).

Blackwell et al. (2004) observed that ringed seals exhibited little or no reaction to pile-driving noise with mean underwater levels of 157 dB re 1 µPa and in-air levels of 112 dB re 20 µPa, suggesting that the seals had habituated to the noise. On the other hand, harbor seals were displaced from areas surrounding wind farm pile driving (average pile driving duration 6 hours) at estimated received levels between 166 and 178 dB re 1 µPa SPL (peak to peak), with presence returning to baseline within two hours of cessation of pile driving (Russell et al., 2016). Similarly, harbor and grey seals avoided a seismic air gun by rapidly swimming away and ceasing foraging, then returned to normal behavior afterwards (Thompson et al. 1998, cited in Gordon et al., 2003).

Captive California sea lions avoided sounds from an underwater impulsive source at levels of 165 to 170 dB re 1 µPa (Finneran et al., 2003b). However, few responses were observed by New Zealand fur seals to a towed air gun array operating at full power; rather, when responses were observed it seemed to be to the physical presence of the vessel and tow apparatus, and these only occurred when the vessel was within 200 m and sometimes as close as 5 m (Lalas & McConnell, 2016). Captive Steller sea lions were exposed to a variety of tonal, sweep, impulsive and broadband sounds to determine what might work as a deterrent from fishing nets (Akamatsu et al., 1996). An impulsive sound at a source level of 210 dB re 1 µPa at 1 m was more likely to cause both adults and juveniles to haul out and refuse to eat fish presented in a net compared to other exposures. Fewer instances of juvenile haul outs and no adult haul outs were observed in response to the same impulse sound at a source level of 165 dB re 1 µPa, including with and without the food item in the test pool. Steller sea lions exposed to in-air explosive blasts increased their activity levels and often re-entered the water when hauled out (Demarchi et al., 2012). However, these responses were short-lived and, within minutes, the animals had hauled out again, and there were no lasting behavioral impacts in the days following the blasts.

Hastie et al. (2021) studied how the number and severity of avoidance events may be an outcome of marine mammal cognition and risk assessment. Five captive grey seals were given the option to forage in a high- or low-density prey patch while continuously exposed to silence, pile driving or tidal turbine playbacks (148 dB re 1 µPa at 1 m). One prey patch was closer to the speaker, so had a higher received level in experimental exposures. Overall, seals avoided both anthropogenic noise playback conditions with higher received levels when the prey density was limited but would forage successfully and for as long as control conditions when the prey density was higher, demonstrating that noise has the potential to impact seal foraging decisions if the level is high enough. Experimentally, Götz and Janik (2011) tested underwater startle responses to a startling sound (sound with a rapid rise time and a 93 dB sensation level [the level above the animal's hearing threshold at that frequency]) and a non-startling sound (sound with the same level, but with a slower rise time) in wild-captured gray seals. The animals exposed to the startling treatment avoided a known food source, whereas animals exposed to the non-startling treatment did not react or habituated during the exposure period. The results of this study highlight the importance of the characteristics of the acoustic signal in an animal's response of habituation.

D.6.5.4 Behavioral Reactions of Sirenians

D.6.5.4.1 Sonar and Other Transducers

Few data exist on manatee responses to sonar; however, there has been some work using side-scan and fish-finding sonar to detect manatees (Gonzalez-Socoloske & Olivera-Gomez, 2012; Gonzalez-Socoloske et al., 2009; Niezrecki, 2010). These are typically very-high-frequency systems, with frequencies over 200 kHz, although in some cases frequencies of 50 kHz were used. The response of the manatees to the sonar was not the focus of these studies, but, when reported, the authors stated that no response was observed. Studies have also been conducted on the efficacy of using pingers to warn manatees about the presence of vessels or fishing gear. Bowles et al. (2001) observed brief startle responses to pingers sweeping 10 to 80 kHz in two of nine manatees tested. However, the captive manatees appeared to be attracted to the fishing gear, and did not avoid it even in the presence of pingers (Bowles et al., 2001). Dugongs in Australia were exposed to 3.5 and 10 kHz pingers with source levels around 133 dB re 1 µPa, with no significant responses observed and continued foraging throughout the experiment (Hodgson & Marsh, 2007). In contrast, wild dugongs in Thailand exposed to 3.5 kHz tones at 141 dB re 1 µPa did not approach the source within 100 m, while playbacks of dugong calls elicited approaches within 10 m (Ichikawa et al., 2009).

These limited data may indicate that sirenians are relatively robust to sonar and other active acoustic sources; however, with the lack of focused studies on these sound sources it is difficult to draw any conclusions.

D.6.5.4.2 Vessel Disturbance

Behavioral responses to vessels can be caused my multiple factors. It is difficult for researchers and analysts to separate the effects of vessel noise and vessel presence; therefore, this section will cover both aspects in publications that specifically aim to target response to noise. The West Indian manatee responds to vessel movement via acoustic and possibly visual cues by moving away from the approaching vessel, increasing its swimming speed, and moving toward deeper water (Miksis-Olds et al., 2007; Nowacek et al., 2004b). When vessels pass within 10 m, manatees respond by fluking, changing their heading or depth, or rolling (Rycyk et al., 2018). The degree of response varies with the individual manatee and may be more pronounced in deeper water, where they are more easily able to determine the direction of the approaching vessel (Nowacek et al., 2004b). Similar responses were observed for slow- and fast-moving vessels (Rycyk et al., 2018). However, they were more likely to change their behavior to boat passes of longer durations, and the longer they had to change their behavior (e.g., slower moving boats) that behavior change occurred earlier relative to the boat's closest approach. In other words, slower moving vessels allowed manatees a greater opportunity to move out of the way of the vessel. This disturbance is a temporary response to the approaching vessel. West Indian manatees have also been shown to seek out areas with a lower density of vessels (Buckingham et al., 1999). West Indian manatees exhibit a clear behavioral response to vessels within distances of 25 to 50 m, but it is unclear at what distance the manatees first detect the presence of vessels (Nowacek et al., 2004b). Vessel traffic and recreation activities that disturb West Indian manatees may cause them to leave preferred habitats and may alter biologically important behaviors, such as feeding, suckling, or resting (Haubold et al., 2006).

In manatees, call rates and call amplitude were affected by noise that shared dominant frequencies of watercraft, with call rates decreasing during feeding and socializing. Differential effects were also seen on call type based on the presence or absence of calves (Miksis-Olds & Tyack, 2009). Similarly, call rates in dugongs did not change in the presence of vessels, but call durations were longer and more harmonics were present when boats passed within 400 m (Ando-Mizobata et al., 2014). These changes in vocalizations varied with the frequency of the noise, the type of call being produced, and the behavioral or social context; taken together, these changes may indicate that responses to vessel noise are dependent on behavioral and environmental contexts.

D.6.5.4.3 Aircraft Disturbance

There are few data on the effects of aircraft overflight on sirenians. Rathbun (1988) studied the reaction of West Indian manatees to both fixed-wing aircraft and helicopters used during census surveys. The manatees did not react to a fixed-wing aircraft moving at approximately 130 km per hour at 160 m altitude. However, animals did react to a helicopter below approximately 100 m moving at speeds of 0 (hovering) to 20 km per hour by startling from rest and diving to deeper waters. This again demonstrates that distance to the aircraft impacts if and how an animal may respond.

Hodgson et al. (2013) conducted a pilot study to conduct aerial surveys of dugongs using an unmanned aerial vehicle flown at altitudes of 500, 750 and 1,000 ft.; no behavioral responses were mentioned but noise levels were much lower than for a typical fixed-wing aircraft. Similarly, manatees were not disturbed by a fixed-wing unmanned vehicle flying at 100 m (Jones IV et al., 2006; Smith et al., 2016).

D.6.5.4.4 Vessel Disturbance

Sea otters that live far inshore and may be exposed to noise from recreational boats and commercial and military ships transiting in and out of port areas. Sea otters have similar in-air hearing sensitivities as pinnipeds (Miksis-Olds et al., 2007; Nowacek et al., 2004b), and may react in a similar fashion when approached by vessels. However, underwater hearing sensitivities are significantly reduced compared to pinnipeds (Ghoul & Reichmuth, 2014a, 2014b). While reactions to underwater vessel noise may occur, they will have lower overall severity to those of pinnipeds. Sea otters in Monterey, CA that were living in areas of disturbance from human activity such as recreational boating spent more time engaged in travel than resting (Curland, 1997). Sea otters in undisturbed areas spent 5 percent of their time travelling; otters in areas of disturbance due to vessels were shown to spend 13 percent of their time travelling (Curland, 1997). While this may not appear to be a large change in behavior, sea otter dives are very costly and require twice the metabolic energy that phocid seals need to dive; therefore sea otters may not dive or travel far in response to disturbance, as they already require long periods of rest at the surface to counterbalance the high cost of foraging at sea (Yeates et al., 2007). For example, when a single air gun vessel passed a large raft of otters, several otters were mildly alarmed (e.g., rolled over on their sides or bellies and looked intently at the vessel as it approached) but did not leave the raft. However, they reacted to the vessel every time it passed, even though the air gun was only operational for two of the four passes. This indicates that otters were either responding to the loud airborne sounds of the boat engines and compressor, or to the close approach of the vessel itself, rather than the seismic sounds (Reidman, 1983). However, sea otters may habituate quickly. Even when purposefully harassed in an effort to cause a behavioral response, sea otters generally moved only a short distance (100 to 200 m) before resuming normal activity, and nearby boats, nets, and floating oil containment booms were sometimes an attractant (Davis et al., 1988). Although Barrett (2019) found that sea otters have a high metabolic rate and are at risk of increased energetic costs when disturbed, there was less than a 10 percent chance of disturbance when small vessels were more 54 m away from sea otters.

D.6.5.4.5 Aircraft Disturbance

Sea otters spend approximately 80 percent of their time on the surface of the water (Curland, 1997) with their heads above the surface, and will most likely be exposed to noise from aircraft. Recordings of underwater noise produced by helicopter overflights did not appear to affect sea otter foraging behavior, foraging success, or daily activity patterns when projected underwater 1 to 1.5 km from a group of otters in Lobos Cove (Reidman, 1983). Sea otters have similar in-air hearing sensitivities as pinnipeds (Ghoul & Reichmuth, 2014a, 2014b), and may react in a similar fashion when exposed to aircraft noise. Pinnipeds in general are unresponsive but may react depending on the altitude of the aircraft or the abruptness of the associated sound (Richardson et al., 1995b), with reactions ranging from unresponsiveness to flushing into the water location (Blackwell et al., 2004; Gjertz & Børset, 1992). Sea otters may dive below the surface of the water or flush into the water to avoid aircraft noise. However, sea otter dives are very costly and require twice the metabolic energy that phocid seals need to dive; therefore sea otters may not dive or travel so readily in response to disturbance, as they already require long periods of rest at the surface to counterbalance the high cost of foraging at sea (Yeates et al., 2007). So far, there has been no evidence that any aircraft has had adverse effects on a well-monitored translocated colony of sea otters at San Nicolas Island, which has a landing field operated by the U.S. Navy (U.S. Fish and Wildlife Service, 2012, 2015).

D.6.5.4.6 Impulsive Noise

There are few available studies on responses of sea otters to impulsive sounds. A playback study of multiple and single air guns had no significant impact on sea otters in California. During the multiple air gun exposures, otters rested 1 percent more and foraged 1 percent less. They were successful at obtaining prey during 84 percent of their foraging dives when the air gun vessel was 50 NM away, and the success rate only decreased by 5 percent when the multiple air gun vessel moved closer (0.5 NM away). Overall, foraging and dive behaviors remained undisturbed, as did the density and distribution of sea otters in the area. This study caveats that the data were collected under rough weather conditions which could have affected the otters’ perception of the seismic sounds. In addition, otters kept close to shore in relatively sheltered coves (Reidman, 1983).

During the single air gun experiment, the air gun ship approached a raft of otters (at a minimum of 730 m), and several otters were mildly alarmed (e.g., rolled over on their sides or bellies and looked intently at the vessel as it approached) but did not leave the raft. Of the four times the vessel passed the group of otters, the air gun was operational during only two of the transects. However, the otters reacted to the vessel every time it passed, indicating that otters were either responding to the loud airborne sounds of the boat engines and compressor, or to the close approach of the vessel itself, rather than the seismic sounds (Reidman, 1983).

In a follow-up study, Riedman (1984) monitored sea otter reactions to drilling platform sounds and air gun firing projected from a source vessel 0.9 to 1.6 km away from groups of sea otters. No behavioral reactions or movements were observed in 14 days of observations with 15–38 individual sea otters present on any given day. Sound pressure levels from the air gun were reported as 166 dB re 1 µPa at 1.1 km, which means that two otters may have been subjected to levels greater than this at ranges of 900 m on the one day the pair foraged closer to the air gun ship for one hour. Most of the otters would have been subjected to just under this level, since the majority of otters foraged 1.3 to 1.6 m away from the sound sources, and propagation loss due to distance and the kelp environment needs to be considered. In a survey of the local coastline, no change in numbers of sea otters was evident between just prior to the sound stimuli and on day 10 of the emissions. No changes in feeding dive times or feeding success was seen during the study either.

When conducting impact and vibratory pile driving for the Parsons Slough estuarine restoration, the Elkhorn Slough National Estuarine Research Reserve (2011) recorded the abundance and behavior of sea otters in the area. Disturbances within 30 m of the pile driving site included otters raising their heads, swimming away without startling, or startle diving. Usually only single adult males with an established territory that included the construction site traveled within 30 m. Otters further away (> 180 m) were observed swimming away with startling, including mother-pup pairs. However, sea otter behavioral disturbances 30–180 m away from the pile driving site were difficult to tease apart from the impacts of pedestrian vessels and other construction activities.

Sea otters spend approximately 80 percent of their time on the surface of the water (Curland, 1997) with their heads above the surface, which reduces their exposure to underwater sounds. They require long periods of undisturbed rest at the surface to counterbalance high metabolic costs associated with forging at sea (Yeates et al., 2007). If reactions to Navy impulsive noise were to occur, they may be similar to those of pinnipeds, which show temporary avoidance responses or cessation of foraging behavior (Gordon et al., 2003; Thompson et al., 1998). However, underwater hearing sensitivities are significantly reduced in sea otters when compared to pinnipeds (Ghoul & Reichmuth, 2014a, 2014b), so reactions may not be as strong, if they occur at all.

D.6.6 PHYSIOLOGICAL RESPONSE

The growing field of conservation physiology relies in part on the ability to monitor stress hormones in populations of animals, particularly those that are threatened or endangered. Physiological stress is an adaptive process that helps an animal cope with changing conditions. The ability to make predictions from stress hormones about impacts on individuals and populations exposed to various forms of stressors, natural and human-caused, relies on understanding the linkages between changes in stress hormones and resulting physiological impacts. Currently, the sound characteristics that correlate with specific stress responses in marine mammals are poorly understood, as are the ultimate consequences of these changes. Navy-funded efforts have improved the understanding of and the ability to predict how stressors ultimately affect marine mammal populations (e.g., King et al., 2015; New et al., 2013a; Pirotta et al., 2015a; Pirotta et al., 2022b). This includes not only determining how and to what degree various types of anthropogenic sound cause stress in marine mammals, but what factors can mitigate those responses. Factors potentially affecting an animal's response to a stressor include the mammal's life history, sex, age, reproductive status, overall physiological and behavioral plasticity, and whether they are naïve or experienced with the sound (e.g., prior experience with a stressor may result in a reduced response due to habituation)(Finneran & Branstetter, 2013; St. Aubin & Dierauf, 2001). Because there are many unknowns regarding the occurrence of acoustically induced stress responses in marine mammals, any physiological response (e.g., hearing loss or injury) or significant behavioral response is assumed to be associated with a stress response.

Marine mammals naturally experience stressors within their environment and as part of their life histories. Changing weather and ocean conditions, exposure to disease and naturally occurring toxins, lack of prey availability, and interactions with predators all contribute to the stress a marine mammal experiences (Atkinson et al., 2015). Breeding cycles, periods of fasting, social interactions with members of the same species, and molting (for pinnipeds) are also stressors, although they are natural components of an animal's life history. Anthropogenic activities have the potential to provide additional stressors beyond those that occur naturally (Fair et al., 2014; Meissner et al., 2015; Rolland et al., 2012). Anthropogenic stressors potentially include such things as fishery interactions, pollution, tourism, and ocean noise.

Relatively little information exists on the linkage between anthropogenic sound exposure and stress in marine mammals, and even less information exists on the ultimate consequences of sound-induced stress responses (either acute or chronic). Most studies to date have focused on acute responses to sound either by measuring catecholamines or heart rate as an assumed proxy for an acute stress response.

D.6.6.1 Heart Rate Response

Increases in heart rate were observed in captive bottlenose dolphins to which known calls of other dolphins were played, although no increase in heart rate was observed when background tank noise was played back (Miksis et al., 2001). Unfortunately, it cannot be determined whether the increase in heart rate was due to stress or social factors, such as expectation of an encounter with a known conspecific. Similarly, a young captive beluga's heart rate increased during exposure to noise, with increases dependent upon the frequency band of noise and duration of exposure, and with a sharp decrease to normal or below normal levels upon cessation of the exposure (Lyamin et al., 2011). Spectral analysis of heart rate variability corroborated direct measures of heart rate (Bakhchina et al., 2017). This response might have been in part due to the conditions during testing, the young age of the animal, and the novelty of the exposure; a year later the exposure was repeated at a slightly higher received level and there was no heart rate response, indicating the beluga whale had potentially habituated to the noise exposure.

Kvadsheim et al. (2010a) measured the heart rate of captive hooded seals during exposure to sonar signals and found an increase in the heart rate of the seals during exposure periods versus control periods when the animals were at the surface. When the animals dove, the normal dive-related heart rate decrease was not impacted by the sonar exposure. Similarly, Thompson et al. (1998) observed a rapid but short-lived decrease in heart rates in wild harbor and grey seals exposed to seismic air guns (cited in Gordon et al., 2003).

Two captive harbor porpoises showed significant bradycardia (reduced heart rate), below that which occurs with diving, when they were exposed to pinger-like sounds with frequencies between 100-140 kHz (Teilmann et al., 2006). The bradycardia was found only in the early noise exposures and the porpoises acclimated quickly across successive noise exposures. Elmegaard et al. (2021) also found that initial exposures to sonar sweeps produced bradycardia but did not elicit a startle response in captive harbor porpoises. As with Teilmann et al. (2006), the cardiac response disappeared over several repeat exposures suggesting rapid acclimation to the noise. In the same animals, 40-kHz noise pulses induced startle responses but without a change in heart rate. Bakkeren et al. (2023) found no change in the heart rate of a harbor porpoise during exposure to masking noise (1/3rd octave band noise, centered frequency of 125 kHz, maximum received level of 125 dB re 1 µPa) during an echolocation task but showed significant bradycardia while blindfolded for the same task. The authors attributed the change in heart rate to sensory deprivation, although no strong conclusions about acoustic masking could be made since the animal was still able to perform the echolocation task in the presence of the masking noise.

Williams et al. (2022) observed periods of increased heart rate variability in narwhals during seismic air gun impulse exposure, but profound bradycardia was not noted. Conversely, Williams et al. (2017) found that a profound bradycardia persisted in narwhals, even though exercise effort increased dramatically as part of their escape response following release from capture and handling.

Limited evidence across several different species suggests that increased heart rate might occur as part of the acute stress response of marine mammals that are at the surface. However, the decreased heart rate typical of diving marine mammals can be enhanced in response to an acute stressor, suggesting that the context of the exposure is critical to understanding the cardiac response. Furthermore, in instances where a cardiac response was noted, there appears to be rapid habituation when repeat exposures occur. Additional research is required to understand the interaction of dive bradycardia, noise-induced cardiac responses, and the role of habituation in marine mammals.

D.6.6.2 Stress Hormone and Immune Response

What is known about the function of the various stress hormones is based largely upon observations of the stress response in terrestrial mammals. The endocrine response of marine mammals to stress may not be the same as that of terrestrial mammals because of the selective pressures marine mammals faced during their evolution in an ocean environment (Atkinson et al., 2015). For example, due to the necessity of breath-holding while diving and foraging at depth, the physiological role of epinephrine and norepinephrine (the catecholamines) might be different in marine versus other mammals.

Catecholamines increase during breath-hold diving in seals, co-occurring with a reduction in heart rate, peripheral vasoconstriction (constriction of blood vessels), and an increased reliance on anaerobic metabolism during extended dives (Hance et al., 1982; Hochachka et al., 1995; Hurford et al., 1996); the catecholamine increase is not associated with increased heart rate, glycemic release, and increased oxygen consumption typical of terrestrial mammals. Captive belugas demonstrated no catecholamine response to the playback of oil drilling sounds (Thomas et al., 1990b) but showed a small but statistically significant increase in catecholamines following exposure to impulsive sounds produced from a seismic water gun (Romano et al., 2004). A captive bottlenose dolphin exposed to the same sounds did not demonstrate a catecholamine response, but did demonstrate a statistically significant elevation in aldosterone (Romano et al., 2004), however, the increase was within the normal daily variation observed in this species (St. Aubin et al., 1996) and was likely of little biological significance. Aldosterone has been speculated to not only contribute to electrolyte balance, but possibly also the maintenance of blood pressure during periods of vasoconstriction (Houser et al., 2011). In marine mammals, aldosterone is thought to play a role in mediating stress (St. Aubin & Dierauf, 2001; St. Aubin & Geraci, 1989).

Yang et al. (2021) measured cortisol concentrations in two captive bottlenose dolphins and found significantly higher concentrations after exposure to 140 dB re 1 µPa impulsive noise playbacks. Two out of six tested indicators of immune system function underwent acoustic dose-dependent changes, suggesting that repeated exposures or sustained stress response to impulsive sounds may increase an affected individual's susceptibility to pathogens. Unfortunately, absolute values of cortisol were not provided, and it is not possible from the study to tell if cortisol rose to problematic levels (e.g., see normal variation and changes due to handling in Houser et al. (2021) and Champagne et al. (2018)). Exposing dolphins to a different acoustic stressor yielded contrasting results. Houser et al. (2020) measured cortisol and epinephrine obtained from 30 captive bottlenose dolphins exposed to simulated U.S. Navy mid-frequency sonar and found no correlation between SPL and stress hormone levels, even though sound exposures were as high as 185 dB re 1 µPa. In the same experiment (Houser et al., 2013b), behavioral responses were shown to increase in severity with increasing received SPLs. These results suggest that behavioral reactions to sonar signals are not necessarily indicative of a hormonal stress response.

Whereas a limited amount of work has addressed the potential for acute sound exposures to produce a stress response, almost nothing is known about how chronic exposure to acoustic stressors affects stress hormones in marine mammals, particularly as it relates to survival or reproduction. In what is probably the only study of chronic noise exposure in marine mammals associating changes in a stress hormone with changes in anthropogenic noise, Rolland et al. (2012) compared the levels of cortisol metabolites in North Atlantic right whale feces collected before and after September 11, 2001. Following the events of September 11, shipping was significantly reduced in the region where fecal collections were made, and regional ocean background noise declined. Fecal cortisol metabolites significantly decreased during the period of reduced ship traffic and ocean noise (Rolland et al., 2012). Rolland et al. (2017) also compared acute (death by ship strike) to chronic (entanglement or live stranding) stressors in North Atlantic right whales and found that whales subject to chronic stressors had higher levels of glucocorticoid stress hormones (cortisol and corticosterone) than either healthy whales or those killed by ships. It was presumed that whales subjected to acute stress may have died too quickly for increases in fecal glucocorticoids to be detected.

Considerably more work has been conducted in an attempt to determine the potential effect of vessel disturbance on smaller cetaceans, particularly killer whales (Bain, 2002; Erbe, 2002; Lusseau, 2006; Noren et al., 2009; Pirotta et al., 2015b; Read et al., 2014; Rolland et al., 2012; Williams et al., 2009; Williams et al., 2014a; Williams et al., 2014b; Williams et al., 2006b). Most of these efforts focused primarily on estimates of metabolic costs associated with altered behavior or inferred consequences of boat presence and noise but did not directly measure stress hormones. However, Ayres et al. (2012) investigated Southern Resident killer whale fecal thyroid hormone and cortisol metabolites to assess two potential threats to the species' recovery: lack of prey (salmon) and impacts from exposure to the physical presence of vessel traffic (but without measuring vessel traffic noise). Ayres et al. (2012) concluded from these stress hormone measures that the lack of prey overshadowed any population-level physiological impacts on Southern Resident killer whales due to vessel traffic. Lemos et al. (2022) investigated the potential for vessel traffic to affect gray whales. By assessing gray whale fecal cortisol metabolites across years in which vessel traffic was variable, Lemos et al. (2022) found a direct relationship between the presence/density of vessel traffic and fecal cortisol metabolite levels. Unfortunately, no direct noise exposure measurements were made on any individual making it impossible to tell if other natural and anthropogenic factors could also be related to the results. Collectively, these studies indicate the difficulty in determining which factors are primarily influence the secretion of stress hormones, including the separate and additive effects of vessel presence and vessel noise. While vessel presence could contribute to the variation in fecal cortisol metabolites in North Atlantic right whales and gray whales, there are other potential influences on fecal hormone metabolites, so it is difficult to establish a direct link between ocean noise and fecal hormone metabolites.

D.6.7 DIRECT INJURY

D.6.7.1 Injury due to Sonar

An object exposed to its resonant frequency will tend to amplify its vibration at that frequency, a phenomenon called acoustic resonance. Acoustic resonance has been proposed as a mechanism by which a sonar or sources with similar operating characteristics could damage tissues of marine mammals. In 2002, NMFS convened a panel of government and private scientists to investigate the potential for acoustic resonance to occur in marine mammals (National Oceanic and Atmospheric Administration, 2002). They modeled and evaluated the likelihood that Navy mid-frequency sonar caused resonance effects in beaked whales that eventually led to their stranding. The conclusion of the group was that resonance in air-filled structures did not likely cause the Bahamas stranding in 2000. The frequency at which resonance was predicted to occur in the animals' lungs was 50 Hz, well below the frequencies used by the mid-frequency sonar systems associated with the Bahamas event. Furthermore, air cavity vibrations, even at resonant frequencies, were not considered to be of sufficient amplitude to cause tissue damage, even under the unrealistic scenario in which air volumes would be undamped (unrestrained) by surrounding tissues and the amplitude of the resonant response would be greatest. These same conclusions would apply to other training activities involving acoustic sources. Therefore, the Action Proponents conclude that acoustic resonance would not occur under real training conditions. The potential impact of acoustic resonance is not considered further in this analysis.

D.6.7.1.1 Acoustically Induced Bubble Formation

A suggested cause of injury to marine mammals is rectified diffusion (Crum & Mao, 1996), the process of increasing the size of a microscopic gas bubble by exposing it to a sound field. The process is dependent upon several factors including the SPL and duration. Under this hypothesis, microscopic bubbles assumed to exist in the tissues of marine mammals may experience one of three things: (1) bubbles grow to the extent they become emboli or cause localized tissue trauma, (2) bubbles develop to the extent that a complement immune response is triggered or the nervous tissue is subjected to enough localized pressure that pain or dysfunction occurs (a stress response without injury), or (3) the bubbles are cleared by the lung without negative consequence to the animal.

Rectified diffusion is facilitated if the environment in which the ensonified bubbles exist is supersaturated with gas. As discussed above, repetitive diving by marine mammals can cause the blood and some tissues to become supersaturated (Ridgway & Howard, 1979). The dive patterns of some marine mammals (e.g., beaked whales) are predicted to induce greater supersaturation (Houser et al., 2001b). If rectified diffusion were possible in marine mammals exposed to high-level sound, conditions of tissue supersaturation could theoretically speed the rate and increase the size of bubble growth. Subsequent effects due to tissue trauma and emboli would presumably mirror those observed in humans suffering from decompression sickness.

It is unlikely that the short duration of sonar pulses would be long enough to drive bubble growth to any substantial size, if such a phenomenon occurs. However, an alternative but related hypothesis has also been suggested: stable microbubbles could be destabilized by high-level sound exposures such that bubble growth then occurs through static diffusion of gas out of supersaturated tissues. In such a scenario, the marine mammal would need to be in a gas-supersaturated state for a long enough time for bubbles to become a problematic size. The phenomena of bubble growth due to a destabilizing exposure was shown by Crum et al. (2005) by exposing highly supersaturated ex vivo bovine tissues to a 37 kHz source at 214 dB re 1 µPa. Although bubble growth occurred under the extreme conditions created for the study, these conditions would not exist in the wild because the levels of tissue supersaturation in the study (as high as 400 to 700 percent) are substantially higher than model predictions for marine mammals (Fahlman et al., 2009; Fahlman et al., 2014; Houser et al., 2001b; Saunders et al., 2008), and such high exposure levels would only occur in very close proximity to the most powerful sonars. For these reasons, it is improbable that this mechanism is responsible for stranding events or traumas associated with beaked whale strandings.

There has been considerable disagreement among scientists as to the likelihood of this phenomenon (Evans & Miller, 2003; Piantadosi & Thalmann, 2004). Although it has been argued that traumas from beaked whale strandings are consistent with gas emboli and bubble-induced tissue separations (Fernandez et al., 2005; Jepson et al., 2003), nitrogen bubble formation as the cause of the traumas has not been verified. The presence of bubbles postmortem, particularly after decompression, is not necessarily indicative of bubble pathology (Bernaldo de Quiros et al., 2012; Bernaldo de Quiros et al., 2013a; Bernaldo de Quiros et al., 2013b; Dennison et al., 2012; Moore et al., 2009), and other mechanisms by which bubble emboli might occur once animals are rapidly stranded (e.g., cardiovascular collapse preventing tissue off-gassing) have not been ruled out (Houser et al., 2009).

D.6.7.2 Behaviorally Mediated Injury

Marine mammals mitigate nitrogen gas accumulation in their blood and other tissues, which is caused by gas exchange from the lungs under conditions of increased hydrostatic pressure during diving, through anatomical, behavioral, and physiological adaptations (Hooker et al., 2012).

Although not an injury caused by the interaction of sound with tissues, variations in marine mammal diving behavior or avoidance responses in response to sound exposure have been hypothesized to result in the off-gassing of nitrogen super-saturated tissues, possibly to the point of deleterious vascular and tissue bubble formation (Hooker et al., 2012; Jepson et al., 2003; Saunders et al., 2008) with resulting symptoms similar to decompression sickness (also known as “the bends”).

Whether marine mammals can produce deleterious gas emboli has been under debate in the scientific community (Hooker et al., 2012; Saunders et al., 2008), although various lines of evidence have been presented in support of the phenomenon. Some of these postulations are described below.

- Analyses of bycaught animals demonstrated that nitrogen bubble formation occurs in drowned animals when they are brought to the surface (Bernaldo de Quiros et al., 2013b; Moore et al., 2009). Since gas exchange with the lungs no longer occurs once drowned, tissues become supersaturated with nitrogen due to the reduction in hydrostatic pressure near the surface. This demonstrates that the phenomenon of bubble formation is at least physically possible.
- The presence of osteonecrosis (bone death due to reduced blood flow) in deep-diving sperm whales has been offered as evidence of impacts due to chronic nitrogen supersaturation and a lifetime of decompression insults (Moore & Early, 2004).
- Dennison et al. (2012) investigated dolphins stranded in 2009–2010. Using ultrasound, they identified gas bubbles in kidneys of 21 of the 22 live-stranded dolphins and in the liver of two of the 22. The authors postulated that stranded animals were unable to recompress by diving, and thus retained bubbles that would have otherwise re-absorbed in animals that continued to dive. However, the researchers concluded that the minor bubble formation observed could be tolerated since most stranded dolphins released did not re-strand.
- A fat embolic syndrome (out-of-place fat particles, typically in the bloodstream) was identified by Fernandez et al. (2005) coincident with the identification of bubble emboli in stranded beaked whales. The fat embolic syndrome was the first pathology of this type identified in marine mammals and was thought to possibly arise from the formation of bubbles in fat bodies, which subsequently resulted in the release of fat emboli into the blood stream.

- Findings of gas and fat emboli in a few stranded Risso's dolphin, and in which sonar exposure was ruled out as a cause of stranding, suggested that other factors, in this case struggling with a prey item, might cause significant variations in dive behavior such that emboli formation could occur (Fernandez et al., 2017).

Only one study has attempted to find vascular bubbles in a freely diving marine mammal (Houser et al., 2009). In that study, no vascular bubbles were imaged by ultrasound in a bottlenose dolphin that repeatedly dove to a 100 m depth and maintained a dive profile meant to maximize nitrogen gas uptake. Thus, although lines of evidence suggest that marine mammals manage excessive nitrogen gas loads, most of the evidence for the formation of bubble and fat emboli come from stranded animals in which physiological compromise due to the stranding event is a potential confounding factor. To validate decompression sickness observations in certain stranded cetaceans found coincident with naval activities, a study used rabbits as an experimental pathological model and found that rabbit mortalities during or immediately following decompression showed systematically distributed gas bubbles (microscopic and macroscopic), as well as emphysema and hemorrhages in multiple organs, similar to observations in the stranded cetacean mortalities (Velazquez-Wallraf et al., 2021). Similar findings were not found in almost half the rabbits that survived at least one hour after decompression, revealing individual variation has an essential role in this condition.

Researchers have examined how dive behavior affects tissue supersaturation conditions that could put an animal at risk of gas bubble embolism. An early hypothesis was that if exposure to a startling sound elicits a rapid ascent to the surface, tissue gas saturation sufficient for the evolution of nitrogen bubbles might result (Fernandez et al., 2005; Jepson et al., 2003). However, modeling suggested that even unrealistically rapid rates of ascent from normal dive behaviors are unlikely to result in supersaturation to the extent that bubble formation would be expected in beaked whales (Zimmer & Tyack, 2007). Instead, emboli observed in animals exposed to mid-frequency active sonar (Fernandez et al., 2005; Jepson et al., 2003) could stem from a behavioral response that involves repeated dives, shallower than the depth of lung collapse (Aguilar de Soto et al., 2006; Hooker et al., 2012; Tyack et al., 2006; Zimmer & Tyack, 2007). Longer times spent diving at mid-depths above lung collapse would allow gas exchange from the lungs to continue under high hydrostatic pressure conditions, increasing potential for supersaturation; below the depth of lung collapse, gas exchange from the lungs to the blood would likely not occur (Costidis & Rommel, 2016; Fahlman et al., 2014). To estimate risk of decompression sickness, Kvadsheim et al. (2012) modeled gas exchange in the tissues of sperm, pilot, killer, and beaked whales based on actual dive behavior during exposure to sonar in the wild. Results predicted that venous supersaturation would be within the normal range for these species, which would presumably have naturally higher levels of nitrogen gas loading. Nevertheless, deep-diving whales, such as beaked whales, have also been predicted to have higher nitrogen gas loads in body tissues for certain modeled changes in dive behavior, which might make them more susceptible to decompression sickness (Fahlman et al., 2014; Fernandez et al., 2005; Hooker et al., 2012; Jepson et al., 2003). Bernaldo de Quirós et al. (2019) summarized discussions from a 2017 workshop on potential sonar impacts on beaked whales, suggesting that the effect of mid-frequency active sonar on beaked whales varies among individuals or populations and that predisposing conditions such as previous exposure to sonar and individual health risk factors may contribute to individual outcomes (such as decompression sickness) as well.

Modeling has suggested that the long, deep dives performed regularly by beaked whales over a lifetime could result in the saturation of long-halftime tissues (i.e., tissues that take longer to give off nitrogen, e.g., fat and bone lipid) to the point that they are supersaturated when the animals are at the surface (Fahlman et al., 2014; Hooker et al., 2009; Saunders et al., 2008). Proposed adaptations for prevention of bubble formation under conditions of persistent tissue saturation have been suggested (Fahlman et al., 2006; Hooker et al., 2009), and because of the time it takes for tissue offloading, it is feasible that long-halftime tissues are not a concern for decompression insults under normal ventilation or dive (recompression) conditions. However, for beaked whale strandings associated with sonar use, one proposed hypothesis is that observed bubble formation may be caused by compromised blood flow due to stranding-related cardiovascular collapse. This would reduce the ability to remove nitrogen from tissues following rapid sonar-induced stranding and could preclude typical management of nitrogen in supersaturated, long-halftime tissues (Houser et al., 2009).

Predictive modeling conducted to date has been performed with many unknowns about the respiratory physiology of deep-diving breath-hold animals. For example, Denk et al. (2020) found intra-species differences in the compliance of tracheobronchial structures of post-mortem cetaceans and pinnipeds under diving hydrostatic pressures, which would affect depth of alveolar collapse. Although, as hypothesized by Garcia Parraga et al. (2018) and reviewed in (Fahlman et al., 2021), mechanisms may exist that allow marine mammals to create a pulmonary shunt without the need for hydrostatic pressure-induced lung collapse, i.e., by varying perfusion to the lung independent of lung collapse and degree of ventilation. If such a mechanism exists, then assumptions in prior gas models require reconsideration, the degree of nitrogen gas accumulation associated with dive profiles needs to be re-evaluated, and behavioral responses potentially leading to a destabilization of the relationship between pulmonary ventilation and perfusion should be considered. Costidis and Rommel (2016) suggested that gas exchange may continue to occur across the tissues of air-filled sinuses in deep diving odontocetes below the depth of lung collapse if hydrostatic pressures are high enough to drive gas exchange across into non-capillary veins.

If feasible, kinetic gas models would need to consider an additional gas exchange route that might be functional at great depths within the odontocetes. Other adaptations potentially mitigating and defending against deleterious nitrogen gas emboli have been proposed (Blix et al., 2013). Researchers have also considered the accumulation of carbon dioxide produced during periods of high activity by an animal, theorizing that accumulating carbon dioxide, which cannot be removed by gas exchange below the depth of lung collapse, might also facilitate the formation of bubbles in nitrogen-saturated tissues (Bernaldo de Quiros et al., 2012; Fahlman et al., 2014). In all these cases, the hypotheses have received little in the way of experimentation to evaluate whether they are supported, thus leaving many unknowns as to the predictive accuracy of modeling efforts.

The appearance of extensive bubble and fat emboli in beaked whales was unique to a small number of strandings associated with certain high-intensity sonar events; the phenomenon has not been observed to the same degree in other stranded marine mammals, including other beaked whale strandings not associated with sonar use. It is uncertain as to whether there is some more easily triggered mechanism for this phenomenon specific to beaked whales or whether the phenomenon occurs only following rapidly occurring stranding events (i.e., when whales are not capable of sufficiently decompressing). Nevertheless, based on the rarity of observations of bubble pathology, the potential for nitrogen decompression sickness due to exposure to the Action Proponents' sound sources is considered discountable.

D.6.7.3 Strandings Associated with Sonar

A stranding occurs when a marine mammal is found dead, either ashore or in the water, or is found alive, but is unable to return to the water, needs medical attention, or is unable to return to its natural habitat without assistance. Marine mammals face many threats in their environment, and many of these factors, both natural and anthropogenic, may cause or contribute to a stranding. These include disease, vessel strike, entanglement, marine debris, algal blooms, pollution, starvation, weather events, and oceanographic changes (National Marine Fisheries Service, 2019). Decomposition, buoyancy, scavenging by other marine species, wave damage, and other oceanic conditions complicate the assessment of marine mammal carcasses (Moore et al., 2020). In most instances, even for the more thoroughly investigated strandings involving post-stranding data collection and necropsies, the cause (or causes) for strandings remains undetermined.

Strandings of deep diving odontocetes, specifically beaked whales, have been correlated with naval anti-submarine warfare sonar use. D'Amico et al. (2009) reviewed global beaked whale mass strandings (two or more marine mammals of the same species other than a mother/calf pair) occurring between 1950 and 2004. The review suggested that 12 of 126 of the strandings could be considered to have coincided in space and time with naval activity that may have included mid-frequency active sonar use. Sonar use during exercises involving the U.S. Navy has been identified as a contributing cause or factor in five specific mass stranding events: Greece in 1996; the Bahamas in March 2000; Madeira Island, Portugal in 2000; the Canary Islands in 2002, and Spain in 2006 (Cox et al., 2006; Fernandez, 2006), as described in the Navy's technical report titled *Marine Mammal Strandings Associated with U.S. Navy Sonar Activities* (U.S. Department of the Navy, 2017b). These five mass strandings have resulted in about 40 known cetacean deaths consisting mostly of beaked whales and with close linkages to mid-frequency active sonar activity. Two minke whales also stranded in shallow water after the U.S. Navy training event in the Bahamas in 2000, although these animals were successfully returned to deep water with no physical examinations; therefore, no final conclusions were drawn on whether the sonar led to their stranding (Filadelfo et al., 2009a; Filadelfo et al., 2009b; U.S. Department of Commerce & U.S. Department of the Navy, 2001). Factors that were associated with these strandings included steep bathymetry, multiple hull-mounted platforms using sonar simultaneously, constricted channels, and acoustic propagation conditions that trapped sound near the sea surface (i.e., strong surface ducts). While no other beaked whale strandings have since been correlated to U.S. Navy sonar use, Simonis et al. (2020) claimed a correlation between sonar and beaked whale strandings in the Mariana Islands between 2007 and 2019. This analysis, however, relied on incomplete or inaccurate assumptions about actual U.S. Navy sonar use around the Mariana Islands, such as news reports about Navy activities rather than actual records of sonar use. In a subsequent analysis, the Center for Naval Analysis found no statistically significant correlation of beaked whale strandings around the Mariana Islands with actual use of U.S. Navy sonar based on the complete classified record of all U.S. Navy sonar used (Center for Naval Analysis, 2020).

Sonar was considered a plausible cause in other stranding investigations for other species: coastal bottlenose dolphins in California (Danil et al., 2021) and melon-headed whales in Hawaii (Southall et al., 2006). It should be noted that other factors were considered plausible causes in these investigations, such as a fisheries interaction for the bottlenose dolphins in California or lunar cycles for the melon-headed whales in Hawaii. In Alaska, Savage et al. (2021) suggested that historical Stejneger's beaked whale strandings could have co-occurred with Navy sonobuoy use but present no evidence of correlation.

Multiple hypotheses regarding the relationship between non-impulsive sound exposure and stranding have been proposed (see Bernaldo de Quirós et al., 2019). These range from direct impact of the sound on the physiology of the marine mammal, to behavioral reactions contributing to altered physiology (e.g., "gas and fat embolic syndrome") (Fernandez et al., 2005; Jepson et al., 2003; Jepson et al., 2005), to behaviors directly contributing to the stranding (e.g., beaching of fleeing animals). Unfortunately, without direct observation of not only the event but also the underlying process, and given the potential for artefactual evidence (e.g., chronic condition, previous injury) to complicate conclusions from the post-mortem analyses of stranded animals (Cox et al., 2006), it has not been possible to determine with certainty the exact mechanism underlying these strandings. Based on examination of the above sonar-associated strandings, Bernaldo de Quirós et al. (2019) list diagnostic features, the presence of all of which suggest gas and fat embolic syndrome for beaked whales stranded in association with sonar exposure. Bernaldo de Quirós et al. (2019) observed that, to date, strandings which have a confirmed association with naval exercise have exhibited all seven of the following diagnostic features:

1. Individual or multiple animals stranded within hours or a few days of an exercise in good body condition
2. Food remnants in the first gastric compartment ranging from undigested food to squid beaks
3. Abundant gas bubbles widely distributed in veins (subcutaneous, mesenteric, portal, coronary, subarachnoid veins, etc.) composed primarily of nitrogen in fresh carcasses
4. Gross subarachnoid and/or acoustic fat hemorrhages
5. Microscopic multi-organ gas and fat emboli associated with bronchopulmonary shock
6. Diffuse, mild to moderate, acute, monophasic myonecrosis (hyaline degeneration) with ”disintegration” of the interstitial connective tissue and related structures, including fat deposits, and their replacement by amorphous hyaline material (degraded material) in fresh and well-preserved carcasses
7. Multi-organ microscopic hemorrhages of varying severity in lipid-rich tissues such as the central nervous system, spinal cord, and the coronary and kidney fat when present

Historically, stranding reporting and response efforts have been inconsistent, although they have improved considerably over the time. Although reporting forms have been standardized nationally, data collection methods, assessment methods, detail of reporting, and procedures vary by region and are not yet standardized across the United States. Conditions such as weather, time, location, and decomposition state may also affect the ability to thoroughly examine a specimen (Carretta et al., 2016b; Moore et al., 2013). Because of this, the current ability to interpret long-term trends in marine mammal stranding is limited. While the investigation of stranded animals provides insight into the types of threats marine mammal populations face, investigations are only conducted on a small fraction of the total number of strandings that occur, limiting the understanding of the causes of strandings (Carretta et al., 2016a).

D.6.7.3.1 Direct Injury due to Explosives

Explosive injury to marine mammals would consist of primary blast injury, which refers to those injuries that result from the compression of a body exposed to a blast wave and is usually observed as barotrauma of gas-containing structures (e.g., lung and gut) and structural damage to the auditory system (Greaves et al., 1943; Office of the Surgeon General, 1991; Richmond et al., 1973). The near instantaneous high magnitude pressure change near an explosion can injure an animal where tissue material properties significantly differ from the surrounding environment, such as around air-filled cavities in the lungs or gastrointestinal tract. Large pressure changes at tissue-air interfaces in the lungs and gastrointestinal tract may cause tissue rupture, resulting in a range of injuries depending on degree of exposure. The lungs are typically the first site to show any damage, while the solid organs (e.g., liver, spleen, and kidney) are more resistant to blast injury (Clark & Ward, 1943). Odontocetes can also incur hemorrhaging in the acoustic fats in the melon and jaw (Siebert et al., 2022). Recoverable injuries would include slight lung injury, such as capillary interstitial bleeding, and contusions to the gastrointestinal tract. More severe injuries, such as tissue lacerations, major hemorrhage, organ rupture, or air in the chest cavity (pneumothorax), would significantly reduce fitness and likely cause death in the wild. Rupture of the lung may also introduce air into the vascular system, producing air emboli that can cause a stroke or heart attack by restricting oxygen delivery to critical organs.

If an animal is exposed to an explosive blast underwater, the likelihood of injury depends on the charge size, the geometry of the exposure (distance to the charge, depth of the animal and the charge), and the size of the animal. In general, models predict that an animal would be less susceptible to injury near the water surface because the pressure wave reflected from the water surface would interfere with the direct path pressure wave, reducing positive pressure exposure (Goertner, 1982; Yelverton & Richmond, 1981). This is shown in the records of humans exposed to blast while in the water, which show that the gastrointestinal tract was more likely to be injured than the lungs, likely due to the shallower exposure geometry of the lungs (i.e., closer to the water surface) (Lance et al., 2015). Susceptibility would increase with depth, until normal lung collapse (due to increasing hydrostatic pressure) and increasing ambient pressures again reduce susceptibility (Goertner, 1982).

The only known occurrence of mortality or injury to a marine mammal due to a Navy training event involving explosives occurred in March 2011 in nearshore waters off San Diego, California, at the Silver Strand Training Complex. This area had been used for underwater demolitions training for at least three decades without prior known incident. On this occasion, however, a group of approximately 100 to 150 long-beaked common dolphins entered the mitigation zone surrounding an area where a time-delayed firing device had been initiated on an explosive with a NEW of 8.76 pounds (lb.) (3.97 kg) placed at a depth of 48 ft. (14.6 m). Approximately one minute after detonation, three animals were observed dead at the surface. The Navy recovered those animals and transferred them to the local stranding network for necropsy. A fourth animal was discovered stranded and dead 42 NM to the north of the detonation three days later. It is unknown exactly how close those four animals were to the detonation. Upon necropsy, all four animals were found to have sustained typical mammalian primary blast injuries (Danil & St Leger, 2011).

Relatively little is known about auditory system trauma in marine mammals resulting from explosive exposure, although it is assumed that auditory structures would be vulnerable to blast injuries. Auditory trauma was found in two humpback whales that died following the detonation of a 5,000 kg explosive used off Newfoundland during demolition of an offshore oil rig platform (Ketten et al., 1993), but the proximity of the whales to the detonation was unknown. Eardrum rupture was examined in submerged terrestrial mammals exposed to underwater explosions (Richmond et al., 1973; Yelverton et al., 1973); however, results may not be applicable to the anatomical adaptations for underwater hearing in marine mammals. In this discussion, primary blast injury to auditory tissues is considered gross structural tissue damage distinct from threshold shift or other auditory effects.

Controlled tests with a variety of lab animals (mice, rats, dogs, pigs, sheep, and other species) are the best data sources on actual injury to mammals due to underwater exposure to explosions. In the early 1970s, the Lovelace Foundation for Medical Education and Research conducted a series of tests in an artificial pond at Kirtland Air Force Base, New Mexico, to determine the effects of underwater explosions on mammals, with the goal of determining safe ranges for human divers. The resulting data were summarized in two reports (Richmond et al., 1973; Yelverton et al., 1973). Specific physiological observations for each test animal are documented in Richmond et al. (1973). Gas-containing internal organs, such as lungs and intestines, were the principle damage sites in submerged terrestrial mammals; this is consistent with earlier studies of mammal exposures to underwater explosions in which lungs were consistently the first areas to show damage, with less consistent damage observed in the gastrointestinal tract (Clark & Ward, 1943; Greaves et al., 1943).

In the Lovelace studies, the first positive acoustic impulse was found to be the metric most related to degree of injury, and size of an animal's gas-containing cavities was thought to play a role in blast injury susceptibility. For these shallow exposures of small terrestrial mammals (masses ranging from 3.4 to 50 kg) to underwater detonations, Richmond et al. (1973) reported that no blast injuries were observed when exposures were less than 6 pounds per square inch per millisecond (psi-ms) (40 pascal seconds [Pa-s]), no instances of slight lung hemorrhage occurred below 20 psi-ms (140 Pa-s), and instances of no lung damage were observed in some exposures at higher levels up to 40 psi-ms (280 Pa-s). An impulse of 34 psi-ms (230 Pa-s) resulted in about 50 percent incidence of slight lung hemorrhage. About half of the animals had gastrointestinal tract contusions (with slight ulceration, i.e., some perforation of the mucosal layer) at exposures of 25–27 psi-ms (170-190 Pa-s). Lung injuries were found to be slightly more prevalent than gastrointestinal tract injuries for the same exposure. The anatomical differences between the terrestrial animals used in the Lovelace tests and marine mammals are summarized in Fetherston et al. (2019). Goertner (1982) examined how lung cavity size would affect susceptibility to blast injury by considering both marine mammal size and depth in a bubble oscillation model of the lung; however, the Goertner (1982) model did not consider how tissues surrounding the respiratory air spaces would reflect shock wave energy or constrain oscillation (Fetherston et al., 2019).

Goertner (1982) suggested a peak overpressure gastrointestinal tract injury criterion because the size of gas bubbles in the gastrointestinal tract are variable, and their oscillation period could be short relative to primary blast wave exposure duration. The potential for gastrointestinal tract injury, therefore, may not be adequately modeled by the single oscillation bubble methodology used to estimate lung injury due to impulse. Like impulse, however, high instantaneous pressures may damage many parts of the body, but damage to the gastrointestinal tract is used as an indicator of any peak pressure-induced injury due to its vulnerability.

Because gas-containing organs are more vulnerable to primary blast injury, adaptations for diving that allow for collapse of lung tissues with depth may make animals less vulnerable to lung injury with depth. Adaptations for diving include a flexible thoracic cavity, distensible veins that can fill space as air compresses, elastic lung tissue, and resilient tracheas with interlocking cartilaginous rings that provide strength and flexibility (Ridgway, 1972). Denk et al. (2020) found intra-species differences in the compliance of tracheobronchial structures of post-mortem cetaceans and pinnipeds under diving hydrostatic pressures, which would affect depth of alveolar collapse. Older literature suggested complete lung collapse depths at approximately 70 m for dolphins (Ridgway & Howard, 1979) and 20 to 50 m for phocid seals (Falke et al., 1985; Kooyman et al., 1972). Follow-on work by Kooyman and Sinnett (1982), in which pulmonary shunting was studied in harbor seals and sea lions, suggested that complete lung collapse for these species would be about 170 m and about 180 m, respectively. Evidence in sea lions suggests that complete collapse might not occur until depths as great as 225 m; although the depth of collapse and depth of the dive are related, sea lions can affect the depth of lung collapse by varying the amount of air inhaled on a dive (McDonald & Ponganis, 2012). This is an important consideration for all divers who can modulate lung volume and gas exchange prior to diving via the degree of inhalation and during diving via exhalation (Fahlman et al., 2009); indeed, there are noted differences in pre-dive respiratory behavior, with some marine mammals exhibiting pre-dive exhalation to reduce the lung volume (e.g., phocid seals Kooyman et al., 1973).

D.6.8 POPULATION CONSEQUENCES TO MARINE MAMMALS FROM ACOUSTIC STRESSORS

This section summarizes the best available science on consequences to marine mammal populations from exposure to acoustic sources.

D.6.8.1 Long-Term Consequences to Populations

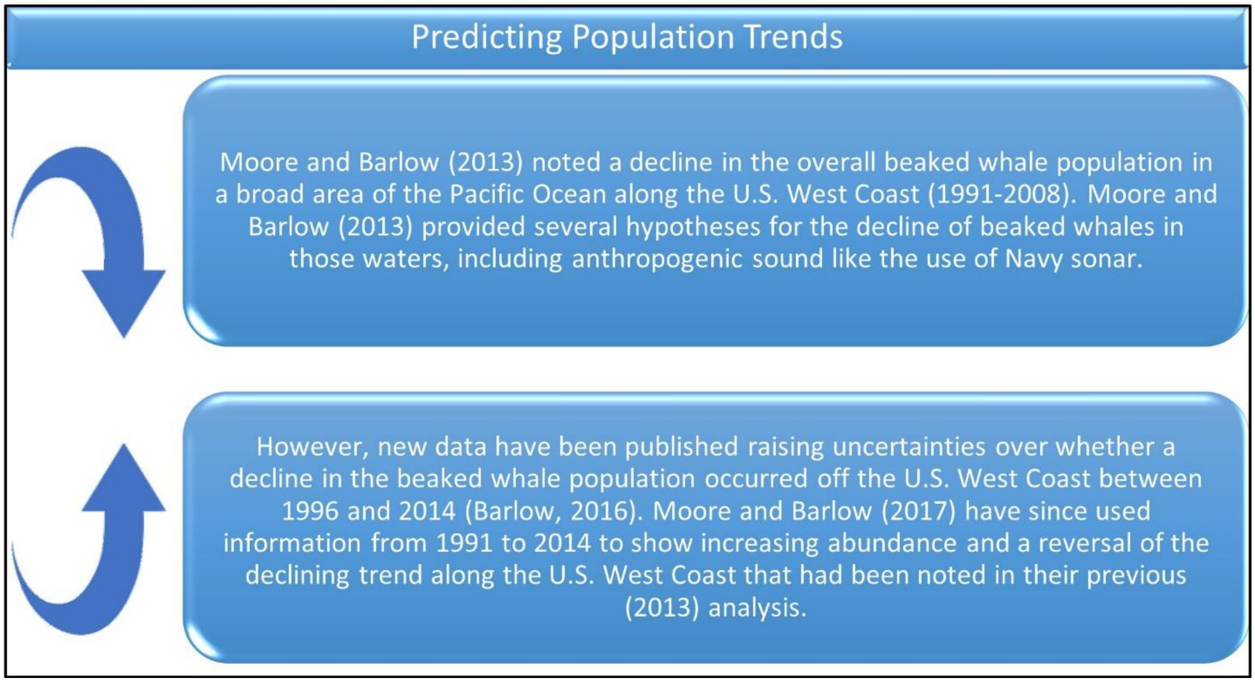
The long-term consequences of disturbance (anthropogenic or environmental), hearing loss, chronic masking, and short-term or chronic physiological stress are difficult to predict because of the different factors experienced by individual animals, such as context of stressor exposure, underlying health conditions, and other environmental or anthropogenic stressors. Linking these non-lethal effects on individuals to changes in population growth rates requires long-term data, which is lacking for many populations.

An important variable to consider is duration of disturbance. Severity scales used to assess behavioral responses to acute sound exposures are not appropriate to apply to sustained or repeated (chronic) exposures, as the focus has shifted from the immediate impacts to an individual to the health of a population over time (Southall et al., 2021). For example, short-term costs experienced over the course of a week by an otherwise healthy individual may be recouped over time after exposure to the stressor ends. These short-term costs would be unlikely to result in long-term consequences to that individual or to that individual’s population. Comparatively, long-term costs accumulated by otherwise healthy individuals over an entire season, year, or throughout a life stage would be less easily recouped and more likely to result in long-term consequences to that individual or population.

Marine mammals exposed to frequent or intense human activities may leave the area, habituate to the activity, or tolerate the disturbance and remain in the area (Wartzok et al., 2003). Highly resident or localized populations may also stay in an area of disturbance because the cost of displacement may be higher than the cost of remaining (Forney et al., 2017). An apparent lack of response (e.g., no displacement or avoidance of a sound source) may not necessarily mean there is no cost to the individual or population, as some resources or habitats may be of such high value that animals may choose to stay, even when experiencing the consequences of stress, masking, or hearing loss (Forney et al., 2017).

Longer term displacement can lead to changes in abundance or distribution patterns of the species in the affected region (Bejder et al., 2006b; Blackwell et al., 2004; Teilmann et al., 2006). For example, gray whales in Baja California abandoned a historical breeding lagoon in the mid-1960s due to an increase in dredging and commercial shipping operations, and only repopulated the lagoon after shipping activities had ceased for several years (Bryant et al., 1984). Mysticetes in the northeast tended to adjust to vessel traffic over a number a of years, trending towards more neutral behavioral responses to passing vessels (Watkins, 1986), indicating that some animals may habituate to high levels of human activity. A study on bottlenose dolphin responses to vessel approaches found that lesser reactions in populations of dolphins regularly subjected to high levels of vessel traffic could be a sign of habituation, or it could be that the more sensitive animals in this population previously abandoned the area of higher human activity (Bejder et al., 2006a).

Population characteristics such as if a population is open or closed to immigration and emigration can influence sensitivity to disturbance as well; closed populations could not withstand a higher probability of disturbance compared to open populations with no limitation on food (New et al., 2020). Still, predicting population trends or long-term displacement patterns due to anthropogenic disturbance is challenging due to limited information and survey data for many species over sufficient temporal and spatial scales, as well as a full understanding of how other factors, such as oceanographic oscillations and climate change, affect presence (e.g., see Figure D.6-4).



Predicting Population Trends

Moore and Barlow (2013) noted a decline in the overall beaked whale population in a broad area of the Pacific Ocean along the U.S. West Coast (1991-2008). Moore and Barlow (2013) provided several hypotheses for the decline of beaked whales in those waters, including anthropogenic sound like the use of Navy sonar.

However, new data have been published raising uncertainties over whether a decline in the beaked whale population occurred off the U.S. West Coast between 1996 and 2014 (Barlow, 2016). Moore and Barlow (2017) have since used information from 1991 to 2014 to show increasing abundance and a reversal of the declining trend along the U.S. West Coast that had been noted in their previous (2013) analysis.

Sources: (Barlow, 2016; Moore & Barlow, 2017; Moore & Barlow, 2013)

Note: Real-world displacement trends are complicated. This example demonstrates how the abundance, and the implied trend of habitat displacement, of beaked whales in an area changed depending on the years analyzed.

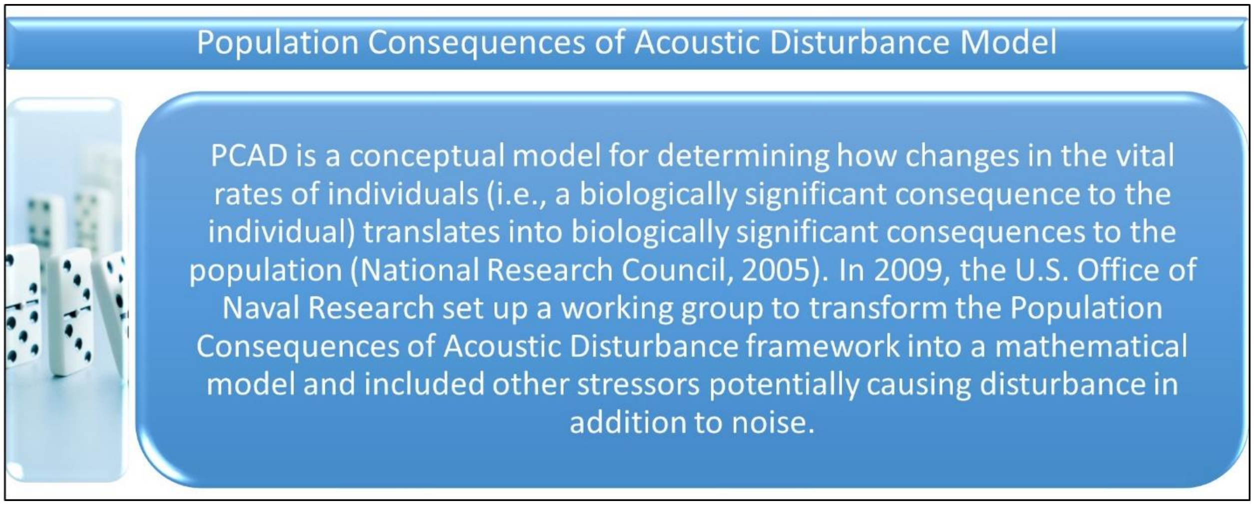
Figure D.6-4: Predicting Population Trends

D.6.8.2 Population Consequences of Disturbance Models

Scientists link short-term effects to individuals from disturbance (anthropogenic impacts or environmental change) to long-term population consequences using population models. Population models accept inputs for the population size and changes in vital rates of the population, such as the mean values for survival age, lifetime reproductive success, and recruitment of new individuals into the population (i.e., raising self-sufficient pups and calves past the weaning stage), to predict changes in population dynamics (e.g., population growth rate). These efforts often rely on bioenergetic models, or energy budget models, which analyze energy intake from food and energy costs for life functions, such as maintenance, growth, and reproduction, either at the individual or population level (Pirotta, 2022). There is high uncertainty around many parameters in these models (Hütt et al., 2023). Model sensitivity analyses have identified the most consequential parameters, including prey characteristics, feeding processes, energy expenditure, body size, energy storage, and lactation capability (Pirotta, 2022).

The National Research Council committee on Characterizing Biologically Significant Marine Mammal Behavior developed an initial conceptual model to link acoustic disturbance to population effects and inform data and research needs (National Research Council, 2005) (Figure D.6-5). This Population Consequences of Acoustic Disturbance, or PCAD, conceptual model linked parameters as illustrated in Figure D.6-6.

In its report, the committee found that the relationships between vital rates and population effects were relatively well understood, but that the relationships between the other components of the model were not well-known or easily observed.



Population Consequences of Acoustic Disturbance Model

PCAD is a conceptual model for determining how changes in the vital rates of individuals (i.e., a biologically significant consequence to the individual) translates into biologically significant consequences to the population (National Research Council, 2005). In 2009, the U.S. Office of Naval Research set up a working group to transform the Population Consequences of Acoustic Disturbance framework into a mathematical model and included other stressors potentially causing disturbance in addition to noise.

Source: (National Research Council, 2005)

Figure D.6-5: Population Consequences of Acoustic Disturbance (PCAD) Model Definition

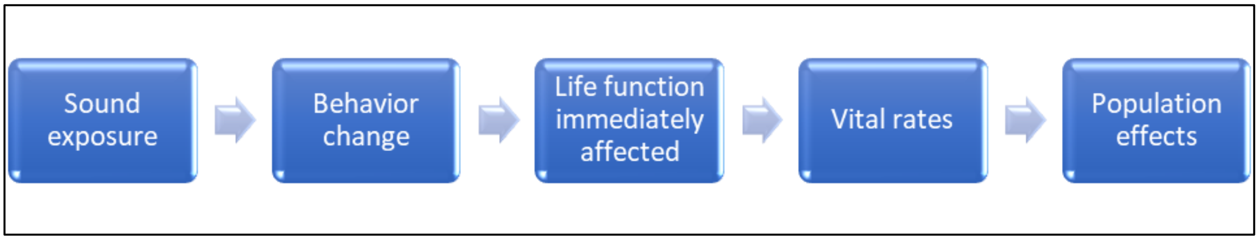
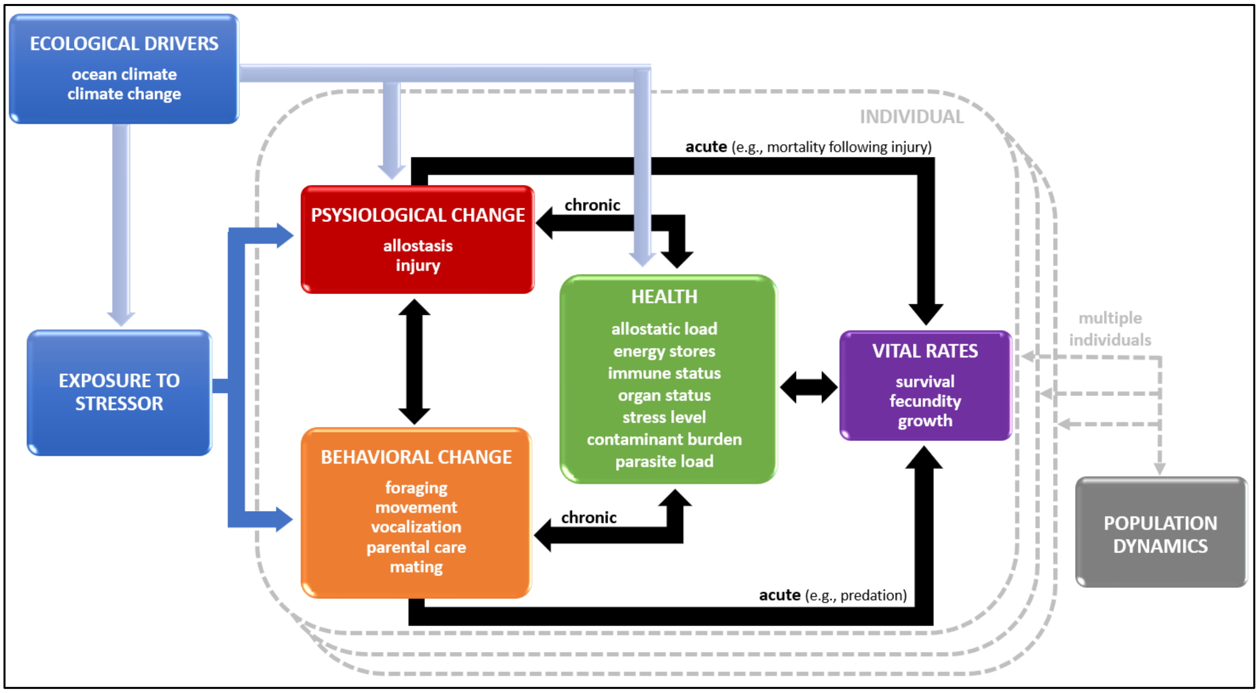


Figure D.6-6: PCAD Model Parameters Flowchart

Building on the PCAD framework, the PcoD conceptual model was developed by an Office of Naval Research working group. The PCOD model considers all types of disturbance, not solely anthropogenic or acoustic, and incorporates physiological changes, such as stress or injury, along with behavioral changes as a direct result of disturbance (National Academies of Sciences Engineering and Medicine, 2017). It also links these changes to both acute effects on vital rates (e.g., survival, fecundity) and chronic effects on health (e.g., energy stores, stress, immunity) (New et al., 2014; Pirotta et al., 2018a). Examples of acute effects include immediate injury, such as vessel strike; immediate health impacts, such as toxic algae exposure; or behavioral responses that increase predation risk (National Academies of Sciences Engineering and Medicine, 2017). Examples of chronic effects include exposure to persistent contaminants and permanent hearing loss (National Academies of Sciences Engineering and Medicine, 2017). These relationships are shown in Figure D.6-7.

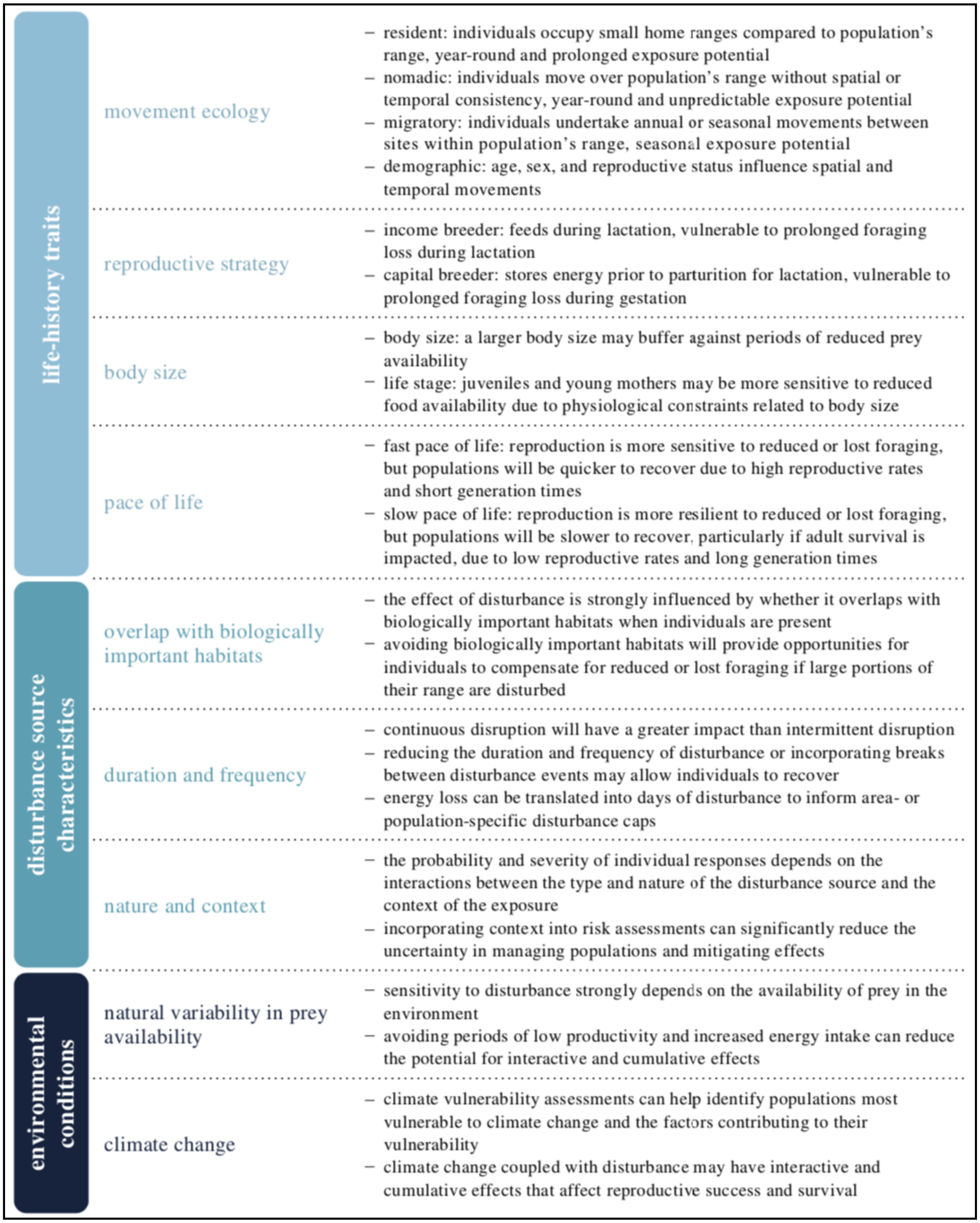


Sources: Adapted from Pirotta et al. (2018a), National Academies of Sciences Engineering and Medicine (2017), New et al. (2014), and Keen et al. (2021)

Figure D.6-7: The Population Consequences of Disturbance Conceptual Model

The Population Consequences of Disturbance (PcoD) model identifies the types of data that would be needed to assess population-level impacts. These data are lacking for many marine mammal species (Booth et al., 2020). Southall et al. (2021) states that future modeling and population simulation studies can help determine population-wide long-term consequences and impact analysis. However, the method to do so is still developing, as there are gaps in the literature, possible sampling biases, and results are rarely ground-truthed, with a few exceptions (Booth et al., 2022; Schwarz et al., 2022). Nowacek et al. (2016) reviewed technologies such as passive acoustic monitoring, tagging, and the use of unmanned aerial vehicles which can improve scientists' abilities to study these model inputs and link behavioral changes to individual life functions and ultimately population-level effects. Relevant data needed for improving analyses of population-level consequences resulting from disturbances will continue to be collected during projects funded by the Navy's marine species monitoring program.

Multiple case studies across marine mammal taxonomic groups have been conducted following the PcoD framework (see Table D.6-11). From these studies, Keen et al. (2021) identified themes and contextual factors relevant to assessing impacts to populations due to disturbance (see Figure D.6-8).



life-history traits	movement ecology	– resident: individuals occupy small home ranges compared to population's range, year-round and prolonged exposure potential – nomadic: individuals move over population's range without spatial or temporal consistency, year-round and unpredictable exposure potential – migratory: individuals undertake annual or seasonal movements between sites within population's range, seasonal exposure potential – demographic: age, sex, and reproductive status influence spatial and temporal movements
	reproductive strategy	– income breeder: feeds during lactation, vulnerable to prolonged foraging loss during lactation – capital breeder: stores energy prior to parturition for lactation, vulnerable to prolonged foraging loss during gestation
	body size	– body size: a larger body size may buffer against periods of reduced prey availability – life stage: juveniles and young mothers may be more sensitive to reduced food availability due to physiological constraints related to body size
	pace of life	– fast pace of life: reproduction is more sensitive to reduced or lost foraging, but populations will be quicker to recover due to high reproductive rates and short generation times – slow pace of life: reproduction is more resilient to reduced or lost foraging, but populations will be slower to recover, particularly if adult survival is impacted, due to low reproductive rates and long generation times
disturbance source characteristics	overlap with biologically important habitats	– the effect of disturbance is strongly influenced by whether it overlaps with biologically important habitats when individuals are present – avoiding biologically important habitats will provide opportunities for individuals to compensate for reduced or lost foraging if large portions of their range are disturbed
	duration and frequency	– continuous disruption will have a greater impact than intermittent disruption – reducing the duration and frequency of disturbance or incorporating breaks between disturbance events may allow individuals to recover – energy loss can be translated into days of disturbance to inform area- or population-specific disturbance caps
	nature and context	– the probability and severity of individual responses depends on the interactions between the type and nature of the disturbance source and the context of the exposure – incorporating context into risk assessments can significantly reduce the uncertainty in managing populations and mitigating effects
environmental conditions	natural variability in prey availability	– sensitivity to disturbance strongly depends on the availability of prey in the environment – avoiding periods of low productivity and increased energy intake can reduce the potential for interactive and cumulative effects
	climate change	– climate vulnerability assessments can help identify populations most vulnerable to climate change and the factors contributing to their vulnerability – climate change coupled with disturbance may have interactive and cumulative effects that affect reproductive success and survival

Source: Table from Keen et al. (2021)

Figure D.6-8: Emerging Themes in PcoD Models that Should Be Considered When Assessing the Likelihood and Duration of Exposure and the Sensitivity of a Population to Disturbance Identified by Keen et al. (2021)

Table D.6-11: Published Models using the Population Consequences of Disturbance Framework

Species	*Disturbance/ PcoD Variables[1]*	*Findings*
Minke whale	Whale watching activities/ M, R, D	Whale watching interactions decreased (42%) feeding and increased (7%) non-feeding activity, but cumulative bioenergetic cost remained low (88,018 kJ) even for the most exposed whale which resulted in a minor decrease in body condition safely below the threshold which would impact fetal growth. Impacts would be larger if vessels interacted with whales significantly more during the feeding season (Christiansen & Lusseau, 2015).
Blue whale	Simulated seismic survey/ M, B, D, N	Migrating blue whales are more likely to go long periods without exposure but are more likely to be exposed to seismic during seasonal presence, like in the California Current feeding grounds. Time and proportion of whales exposed increased (< 19%) as stressor increased (Costa et al., 2016a).
	Five scenarios of natural (El Nino or unprecedented change) and unspecified anthropogenic disturbances modeled as lost foraging time (i.e., 0%, habitat displacement, or 50%)/ M, R, B, D, N, V	Short environmental changes like El Nino reduced calf recruitment a little, but unprecedented climate changes impacted fecundity much more (i.e., increased abortions). Weak anthropogenic disturbances over a diffuse area (e.g., ship traffic, whale watching) had little effect on fecundity. Impact from intense, continuous noise (e.g., seismic, pile driving) depended on females' response. If they stayed in the area, body condition decreased and rate of abortions and starvation increased; if they moved to feed elsewhere there was no long-term effect (Pirotta et al., 2018b).
	Natural and unspecified anthropogenic disturbances modeled as lost foraging time/ M, R, S, P, B, D, N, V, C	Blue whale model above was expanded to encompass females' entire lives. Increased frequency of climate change decreased fecundity gradually (e.g., calves weaned early). High levels of anthropogenic disturbance only impacted vital rates if disturbance occurred for 1 year in all locations of the home range, or if disturbance was localized in summer feeding grounds. Repeated disturbance decreased reproductive success and survival of young whales (Pirotta et al., 2019).
	Sonar/ M, N, V	Activity budgets, lunging rates and ranging pattern caused variability in the predicted cost of sonar disturbance. With disturbance, whale reproductive strategies resulted in lower fitness (Pirotta et al., 2021).
Gray whale	Unspecified "anthropogenic disturbance"/ M, R, P, D	Western gray whales had greater energetic requirements during the longer migration to Baja California and China, compared to the shorter migration of Eastern gray whales, so were more sensitive to energy lost through disturbance (Villegas-Amtmann et al., 2017).
	Seismic surveys/ M, R, S, P, B, N, V	Direct disturbance or displacement from nearshore (less energy-rich) areas had little impact on population abundance, but females deprived access to energy-rich offshore summer feeding grounds decreased reproductive success and adult survival, leading to long-term consequences on population abundance (McHuron et al., 2021).
Long-finned pilot whale	Unspecified disturbance modeled as "lost foraging days" for mother-calf pairs (e.g., habitat displacement)/ R, S, P, D, V	Short disturbances increased mortality of calves born to young mothers, and longer disturbances increased calf mortality (born to older mothers) and decreased the life expectancy for mothers, including starvation during lactation periods. Disturbance impacted whales faster in winter (5+ days) than in summer (20+ days) (Hin et al., 2019).

Table D.6-11: Published Models using the Population Consequences of Disturbance Framework (continued)

Species	*Disturbance/ PcoD Variables*[1]	*Findings*
	Unspecified "anthropogenic disturbance"/ R, S, P, D, V	Modeled disturbance decreased reproductive strategies and fitness. When resources were not evenly distributed, cautious strategies and knowledge of resource variation was advantageous (Pirotta et al., 2020).
	Unspecified disturbance modeled as "lost foraging days" (e.g., habitat displacement)/ R, S, P, D, V	Disturbance decreased population density (e.g., young lactating females) and increased prey availability, which resulted in improved body condition in the population overall and no net impact on lifetime reproductive output, suggesting that fitness markers may not indicate population effects (Hin et al., 2021).
Humpback whale	Simulated seismic survey/ M, B, D, N	Whale populations that foraged for krill over wide areas (West Antarctic Peninsula) were exposed to seismic less, resulting in less disturbed foraging behavior. In contrast, Bering Sea humpback whales hunted fish over a much smaller/ localized area, and have a limited range for foraging where more whales (90%) were exposed to seismic and interrupted while foraging (Costa et al., 2016a).
	Seismic surveys/ M, R, S, P, N	PcoD models can be used for predicting population consequences or making management decisions, depending if forwards or backward approach is used (Dunlop et al., 2021).
North Atlantic Right Whale	Fishing gear entanglements/ D, N, V, C	Entanglement and limited prey availability can be considered continuous stressors (e.g., prey density changes throughout range and entanglement level), and compounded impacts as entanglement decreases foraging success. When there isn't enough empirical information, a mechanistic model can be used to simulate the interaction between varying levels of entanglement, feeding rate and maximum prey intake (Pirotta et al., 2022b).
Beaked whale	Unspecified "non-lethal" disturbance/ R, S, P, B, V	Different assumptions for duration of gestation and lactation can alter model results for mother and calf mortality. Six beaked whale species were very sensitive, Baird's had a quick time to weaning, and Longman's needed higher quality habitat. Consistent long-term disturbance with minor reduction in energy intake may have same effect as strong, short-term disturbance that halts energy intake. Many conservative assumptions were used for this model since many parameters were unknown for 21 beaked whale species (New et al., 2013b).
	Sonar/ M, B, D, N	Beaked whales at SOAR and AUTEC ranges exposed to MFA navy sonar could have outcomes ranging from slight increase in population abundance to population extinction, depending on the interaction of sonar use, habitat quality, and the whales' behavioral response to sonar (i.e., displacement, cessation of feeding, both, or no response) (Hin et al., 2023)
Killer whale	Vessel strike, vessel noise, polychlorinated biphenyls contamination/ R, P, B, V	Both Northern and Southern killer whale populations were impacted by the interaction of low prey abundance with vessel strike, vessel noise, and contaminants, but more research is needed to validate the mechanisms of all non-prey variables (Murray et al., 2021).
Harbor porpoise	Wind turbine noise, ship noise/ M, R, S, P, N	Even assuming a 10% reduction in population size, if prey is impacted up to two days, the presence of ships and wind turbines did not deplete the population (Nabe-Nielsen et al., 2014).

Table D.6-11: Published Models using the Population Consequences of Disturbance Framework (continued)

Species	*Disturbance/ PcoD Variables*[1]	*Findings*
	Pile driving/ M, R, S, P, D, V	Predicted a < 0.5% decline in harbor porpoise population size from wind farm construction in worst case scenario (King et al., 2015).
	Seismic surveys/ M, R, S, P, N, V	Seismic activity in May had less impact on porpoise health and reproduction, and seismic in September had more impact (Gallagher et al., 2021).
Sperm whale	Oil spill, seismic survey/ M, R, S, P, B, D, N	10-year model projected population reductions from the oil spill and further declines when compounded with exposure to seismic surveys. Amount of additional population decline due to seismic noise depended on modeling method (i.e., single step-functions had more impacts than functions with multiple steps and frequency weighting). Resilient populations (e.g., able to make up reserves through increased foraging) mediate impacts from both disturbances (Farmer et al., 2018a).
	Unspecified "anthropogenic disturbance" associated with reduced foraging efficiency/ R, S, P, D	Mothers with calves were most vulnerable to foraging disruptions due to high energetic cost of lactation (Farmer et al., 2018b).
Bottlenose dolphin	Climate change, ship noise, fisheries bycatch, epizootic (morbillivirus)/ R, S, P, D, B, V, C	5-year model predicted that epizootic and climate change scenarios would have the largest impact on population size and fecundity. Fisheries interactions and shipping noise disturbance had little overall impact on population abundances in either location, even in the most extreme impact scenarios modeled (Reed et al., 2020).
Northern elephant seal	Unspecified "environmental change" or "anthropogenic disturbance"/ M, R, P, D	Predicted that populations of elephant seals are relatively robust even with > 50% reduction in foraging trips (only a 0.4% population decline in the following year) (New et al., 2014).
	Continuous acoustic disturbance/ M, R, N	Elephant seals would be less impacted than California sea lions since their foraging range and transit area is more expansive. Negligible impacts on reproduction and pup survival rates (Costa et al., 2016b).
Harbor seal	Pile driving/ M, R, S, P, D	Worst-case scenario PCAD model predicted that the 18% of harbor seals with PTS from wind farm construction noise exposure could translate to higher mortality rates or lower reproductive rates for the population (Thompson et al., 2013b).
California sea lion	Continuous acoustic disturbance/ M, R, N	California sea lions were disturbed for a longer period than elephant seals because the sea lions' range (foraging and transit area) is more limited. However, even animals exposed for the longest periods had negligible modeled impacts on their reproduction and pup survival rates (Costa et al., 2016b).
	Generalized disturbance/ M, R, S, P, D	Very short duration disturbances/responses led to little change, particularly if the disturbance was a single event, and changes in the timing of the event in the year had little effect. Relatively short disturbances or mild responses, when a disturbance was modeled as recurring, resulted in a fewer number of adults and pups. The effects weren't noticeable for several years, as the impacts on pup survival did not affect the population until those pups were mature (McHuron et al., 2018a).

Table D.6-11: Published Models using the Population Consequences of Disturbance Framework (continued)

Species	*Disturbance/ PcoD Variables*[1]	*Findings*
11 mysticete and odontocete species[2]	Sonar/ M, S, P, V	Short-term energetic cost was influenced more by lost foraging opportunities than increased locomotor effort during avoidance. Mysticetes incurred more energetic costs that odontocetes, even during mild behavioral responses to mid-frequency active sonar (Czapanskiy et al., 2021).

[1] If an anthropogenic disturbance was modeled it is included, along with the variables included in the PcoD model, such as life-history traits (M= movement ecology, R= reproductive strategy, S= body size, P= pace of life), disturbance of source characteristics (B= overlap with biologically important habitats, D= duration and frequency, N= nature and context), and environmental conditions (V= natural variability in prey, C= climate change). Notation adapted from Keen et al. (2021).

[2] 11 species studied: harbor porpoise, Risso's dolphin, Bainville's beaked whale, short-finned pilot whale, long-finned pilot whale, goose-beaked whale, minke whale, sperm whale, humpback whale, fin whale, and blue whale (Czapanskiy et al., 2021).

Notes: % = percent; > = greater than; kJ = kilojoule; PCAD = Population Consequences of Acoustic Disturbance; PcoD = Population Consequences of Disturbance; PTS = permanent threshold shift

D.6.8.3 Movement Ecology

A population's movement ecology determines the potential for temporal and spatial overlap with a disturbance. Resident populations or populations that rely on spatially limited habitats for critical life functions, such as foraging or breeding, would be at greater risk of repeated or chronic exposure to disturbances than populations that are wide-ranging relative to the footprint of a disturbance (Keen et al., 2021). Even for the same species, differences in habitat use between populations can result in different potential for repeated exposure to individuals for a similar stressor (Costa et al., 2016a). The location and radius of disturbance can impact how many animals are exposed and for how long (Costa et al., 2016b). While some models have shown the advantages of populations with larger ranges, namely the decreased chance of being exposed (Costa et al., 2016b), it's important to consider that for some species, the energetic cost of a longer migration could make a population more sensitive to energy lost through disturbance (Villegas-Amtmann et al., 2017). In addition to ranging patterns, a species' activity budgets and lunging rates can cause variability in their predicted cost of disturbance as well (Pirotta et al., 2021).

D.6.8.4 Resource Dependence

Bioenergetics frameworks that examine the impact of foraging disruption on body reserves of individual whales found that rates of daily foraging disruption can predict the number of days to terminal starvation for various life stages (Farmer et al., 2018b). Similarly, when a population is displaced by a stressor, and only has access to areas of poor habitat quality (i.e., low prey abundance) for relocation, bioenergetic models may be more likely to predict starvation, longer recovery times, or extinction (Hin et al., 2023). There is some debate over the use of blubber thickness as a metric of cetacean energy stores and health, as marine mammals may not use their fat stores in a similar manner to terrestrial mammals (Derous et al., 2020).

Resource limitation can impact population growth rate regardless of additional anthropogenic disturbance. Stochastic Dynamic Programming models have been used to explore the impact declining prey species has on focal marine mammal predators (McHuron et al., 2023a; McHuron et al., 2023b). A Stochastic Dynamic Programming model determined that a decrease in walleye pollock availability increased the time and distance northern fur seal mothers had to travel offshore, which negatively impacted pup growth rate and wean mass, despite attempts to compensate with longer recovery time on land (McHuron et al., 2023b).

Prey is an important factor in long-term consequence models for many species of marine mammals. In disturbance models that predict habitat displacement or otherwise reduced foraging opportunities, populations are being deprived of energy dense prey or “high quality” areas which can lead to long-term impacts on fecundity and survival (Czapanskiy et al., 2021; Hin et al., 2019; McHuron et al., 2023a; New et al., 2013b).

Prey density limits the energy available for growth, reproduction, and survival. Some disturbance models indicate that the immediate decrease in a portion of the population (e.g., young lactating mothers) is not necessarily detrimental to a population, since as a result, prey availability increases and the population’s overall improved body condition reduces the age at first calf (Hin et al., 2021).

The timing of a disturbance with seasonally available resources is important. If a disturbance occurs during periods of low resource availability, the population-level consequences are greater and occur faster than if the disturbance occurs during periods when resource levels are high (Hin et al., 2019). When resources are not evenly distributed, populations with cautious strategies and knowledge of resource variation have an advantage (Pirotta et al., 2020).

Even when modeled alongside several anthropogenic sources of disturbance (e.g., vessel strike, vessel noise, chemical contaminants, sonar), several species of marine mammals are most influenced by lack of prey (Czapanskiy et al., 2021; Murray et al., 2021). Some species like killer whales are especially sensitive to prey abundance due to their limited diet (Murray et al., 2021). The short-term energetic cost of eleven species of cetaceans and mysticetes exposed to mid-frequency active sonar was influenced more by lost foraging opportunities than increased locomotor effort during avoidance (Czapanskiy et al., 2021). Additionally, the model found that mysticetes incurred more energetic cost than odontocetes, even during mild behavioral responses to sonar. These results may be useful in the development of future Population Consequences of Multiple Stressors and Population Consequences of Disturbance models since they should seek to qualify cetacean health in a more ecologically relevant manner.

D.6.8.5 Harbor Porpoises and Non-Military Disturbance Consequences

Studies have investigated the potential consequences of fasting for harbor porpoises because their high metabolic rate may leave them especially vulnerable to disturbances that prevent them from feeding. Four stranded harbor porpoises were able to consume 85–100 percent of their daily food mass intake in a short time period with no physical problems, suggesting that they can compensate for periods of missed feeding if food is available (Kastelein et al., 2019c). Harbor porpoises are also capable of recovering from lost foraging opportunities, largely because of their varied diet, high foraging rates, and high prey capture success (Booth, 2019). By modeling their foraging behavior and known prey species and sizes, the porpoises’ generalist feeding behavior, in most scenarios, would enable them to obtain more than 100 percent of their energetic needs through typical foraging behavior, and therefore would largely be robust to short-term disturbances to foraging.

Seasonality is an important predictor of disturbance for harbor porpoises. Movement and foraging behavior were modeled in seasons, and seismic activity in May had a much smaller impact on harbor porpoise health and reproduction, due to the porpoises having greater energy stores that time of year and females having already weaned their calves (Gallagher et al., 2021). In contrast, seismic surveys in September had a much greater impact due to lower energy reserves at that time, while females were lactating and possibly pregnant as well.

Different stressors and models have generated different long-term consequences within the same species. Even when high and frequent exposure levels are included, some harbor porpoise models result in few long-term consequences from sound exposure (e.g., wind farms, pile driving), but have costly results in others (e.g., pile driving, seismic surveys) (King et al., 2015). For example, the impact of noise from wind farms on harbor porpoises predicted that even when assuming a 10 percent reduction in population size if prey is impacted up to two days, the presence of ships and wind turbines did not deplete the population. Similarly, even under the worst case scenarios, King et al. (2015) model of wind farm impacts on harbor porpoises predicted less than a 0.5 percent decline in harbor porpoise populations. De Silva et al. (2014) analyzed the long-term impacts of a different stressor (pile driving and construction noise) on harbor porpoises and bottlenose dolphins. Despite including the extreme and unlikely assumptions that 25 percent of animals that received PTS would die, and that behavioral displacement from an area would lead to breeding failure, the model only found short-term impacts on the population size and no long-term effects on population viability. In contrast, Heinis et al. (2015) used the Population Consequences of Disturbance framework to estimate impacts from both pile driving and seismic exploration on harbor porpoises and found a 23 percent decrease in population size over six years, with an increased risk for further reduction with additional disturbance days. These seemingly contradictory results demonstrate that refinements to models need to be investigated to improve consistency and interpretation of model results.

D.6.8.6 Multiple Stressors and Cumulative Effects

Population consequences of disturbance models have been used to assess the impacts of multiple and recurring stressors. A marine mammal population that is already subject to chronic stressors like climate change will likely be more vulnerable to acute disturbances. Models that have looked at populations of cetaceans who are exposed to multiple stressors over several years have found that even one major chronic stressor (e.g., climate change, epizootic disease, oil spill) has severe impacts on population size. A layer of one or more stressor (e.g., seismic surveys) in addition to a chronic stressor (like an oil spill) can yield devastating impacts on a population. These results may vary based on species and location, as one population may be more impacted by chronic shipping noise, while another population may not. However, just because a population doesn’t appear to be impacted by one chronic stressor (e.g., shipping noise), does not mean they aren’t affected by others, such as climate change or disease (Reed et al., 2020). Recurring or chronic stressors can impact population abundance even when instances of disturbance are short and have minimal behavioral impact on an individual (Farmer et al., 2018a; McHuron et al., 2018b; Pirotta et al., 2019). Some changes to response variables like pup recruitment (survival to age one) aren’t noticeable for several years, as the impacts on pup survival does not affect the population until those pups are mature but impacts to young animals will ultimately lead to population-wide declines. The severity of the repeated disturbance can also impact a population’s long-term reproductive success. Scenarios with severe repeated disturbance (e.g., 95 percent probability of exposure, with 95 percent reduction in feeding efficiency) can severely reduce fecundity and calf survival, while a weaker disturbance (25 percent probability of exposure, with 25 percent reduction in feeding efficiency) had no population-wide effect on vital rates (Pirotta et al., 2019). An expanded version of the Population Consequences of Multiple Stressors framework in Figure D.6-7 would include multiple “exposure to stressor” buttons to signify the many stressors an individual and population faces, as well as multiple layers of physiological and behavioral responses per individual (National Academies of Sciences Engineering and Medicine, 2017).

The study that modeled an oil spill led to chronic declines in a sperm whale population over 10 years, and if models included even one more stressor (i.e., behavioral responses to air guns), the population declined even further (Farmer et al., 2018a). However, the amount of additional population decline due to acoustic disturbance depended on the way the dose-response of the noise levels were modeled. A single step-function led to higher impacts than a function with multiple steps and frequency weighting. In addition, the amount of impact from both disturbances was mediated when the metric in the model that described animal resilience was changed to increase resilience to disturbance (e.g., able to make up reserves through increased foraging).

Not all stressors have the same impact for all species and all locations. Another model analyzed the effect of a number of chronic disturbances on two bottlenose dolphin populations in Australia over 5 years (Reed et al., 2020). Results indicated that disturbance from fisheries interactions and shipping noise had little overall impact on population abundances in either location, even in the most extreme impact scenarios modeled. At least in this area, epizootic and climate change scenarios had the largest impact on population size and fecundity.

Recurring stressors can impact population abundance even when individual instances of disturbance are short and have minimal behavioral impact on an individual. A model on California sea lions introduced a generalized disturbance at different times throughout the breeding cycle, with their behavior response being an increase in the duration of a foraging trip by the female (McHuron et al., 2018b). Very short duration disturbances or responses led to little change, particularly if the disturbance was a single event, and changes in the timing of the event in the year had little effect. However, with even relatively short disturbances or mild responses, when a disturbance was modeled as recurring there were resulting reductions in population size and pup recruitment (survival to age one). Often, the effects weren’t noticeable for several years, as the impacts on pup survival did not affect the population until those pups were mature.

D.6.8.7 PcoD Models as Tools for Management

PcoD models may also have application for species management. One model used for migrating humpback whale mother-calf pair responses to seismic surveys used both a forwards and backward approach (Dunlop et al., 2021). While a typical forwards approach can determine if a stressor would have population-level consequences, authors demonstrated that working backwards through a Population Consequences of Disturbance model can be used to assess the worst-case scenario for an interaction of a target species and stressor. This method may be useful for future management goals when appropriate data becomes available to fully support the model.

D.6.8.8 Long-Term Consequences on Navy Ranges

D.6.8.8.1 Blue Whales on Navy Ranges in Southern California

The U.S. Navy funds research on blue whale sonar disturbance on Navy ranges. Pirotta et al. (2018b) modeled one reproductive cycle of a female North Pacific blue whale, starting with leaving the breeding grounds off Baja California to begin migrating north to feeding grounds off California, and ending with her returning to the breeding grounds, giving birth, and lactating. They modeled this scenario with no disturbance and found 95 percent calf recruitment (the successful growth and weaning of a calf); under a “normal” environmental perturbation (El Niño-Southern Oscillation) there was a very small reduction in recruitment, and, under an “unprecedented” environmental change, recruitment was reduced to 69 percent. An intense, localized anthropogenic disturbance was modeled (although the duration of the event was not provided); if the animals were not allowed to leave the area, they did not forage, and recruitment dropped to 63 percent. However, if animals could leave the area of the disturbance, then there was almost no change to the recruitment rate. A weak but broader spatial disturbance, where foraging was reduced by 50 percent, caused only a small decrease in calf recruitment to 94 percent.

Pirotta et al. (2022a) investigated the potential long-term effects of changing environmental conditions and military sonar by modeling vital rates of Eastern North Pacific blue whales. Previous work from Pirotta et al. (2021) was used as a foundation for incorporating the best available science into the 2022 vital rate model. Using data and underlying models of behavioral patterns, energy budgets, body condition, contextual responses to noise, and prey resources, the model predicted female vital rates including survival (age at death), and reproductive success (number of female calves). The model simulation results showed that environmental changes were more likely to affect vital rates, "while the current regime of sonar activities was not " (Pirotta et al., 2022a). The case study used an annual sonar regime in Southern California Range Complex based on the description of the action in the Navy's 2018 Hawaii-Southern California Training and Testing EIS/OEIS. Additional military sonar scenarios were modeled, and only a ten-fold increase in sonar activity combined with a shift in geographical location to overlap with main feeding areas of blue whales resulted in a moderate decrease in lifetime reproductive success (Cohen's $d = 0.47$), but there was no effect on survival (Cohen's $d = 0.05$).

D.6.8.8.2 Beaked Whales on Navy Ranges

The Navy has funded sonar research on three instrumented ranges that contribute to understanding long-term effects on beaked whale populations exposed to sonar: Southern California Anti-Submarine Warfare Range, Atlantic Undersea Test and Evaluation Center, and the Pacific Missile Range Facility. Long-term impacts to sensitive beaked whale populations on Navy testing and training grounds is a heavily researched topic, and the residency on the range may play a role. Studies on the AUTEC instrumented range in the Bahamas have shown that some Blainville's beaked whales may be residents during all or part of the year in the area. Individuals may move off the range for several days during and following a sonar event but return within a few days (Joyce et al., 2019; McCarthy et al., 2011; Tyack et al., 2011).

A study by Benoit-Bird et al. (2020) demonstrated that differences in prey distribution could be a substantial factor for beaked whale habitat preference in the Bahamas. Photo-identification studies in the SOCAL Range Complex have identified approximately 100 individual goose-beaked whales, with 40 percent having been seen in one or more prior years and re-sightings up to seven years apart (Falcone & Schorr, 2014; Falcone et al., 2009). These results indicate long-term residency by individuals in an intensively used Navy training area, which may suggest a lack of long-term consequences from exposure to Navy training activities but could also be indicative of high-value resources that exceed the cost of remaining in the area. Long-term residency by itself does not mean there has been no impact on population growth rates and there are no data on the reproductive rates of populations inhabiting the Navy range area around San Clemente Island compared to beaked whales from other areas. In that regard however, results from photo-identification efforts can provide critically needed calving and weaning rate data for resident animals on the Navy's Southern California range. Three adult females that had been sighted with calves in previous years were again sighted in 2016, one of these was associated with her second calf, and a fourth female that was first identified in 2015 without a calf, was sighted in 2016 with a calf (Schorr et al., 2017). Resident females documented with and without calves from year to year will provide data on growth rate for this population.

Beaked whales may routinely move hundreds of kilometers as part of their normal pattern. While at least some beaked whales are residents of a particular area, more than three beaked whales in the SOCAL Range Complex have been documented traveling hundreds of kilometers after being tagged (Falcone & Schorr, 2012, 2014). Out of eight goose-beaked whales, five made journeys of approximately 250 km from their tag deployment location, and one of these five made an extra-regional excursion over 450 km south to Mexico and back again (Schorr et al., 2014).

D.6.8.8.3 Ongoing Research and Monitoring

The best assessment of long-term consequences from Navy training activities will be to monitor the populations over time within the Study Area. A U.S. workshop on Marine Mammals and Sound (Fitch et al., 2011) indicated a critical need for baseline biological data on marine mammal abundance, distribution, habitat, and behavior over sufficient time and space to evaluate impacts from human-generated activities on long-term population survival. The Navy has implemented comprehensive monitoring plans since 2009 for protected marine mammals on Navy ranges with the goal of assessing the impacts of training activities on marine species and the effectiveness of the Navy's mitigation measures. The results of this long-term monitoring are continually being compiled and analyzed for trends in occurrence or abundance over time (e.g., Martin et al., 2017).

Preliminary results of this analysis at Pacific Missile Range Facility off Kauai, Hawaii indicate no changes in detection rates for several species over the past decade, demonstrating that Navy activities may not be having long-term population-level impacts. This type of analysis can be expanded to the other Navy ranges, such as in the Pacific Northwest. Continued analysis of this 15-year dataset and additional monitoring efforts over time are necessary to fully understand the long-term consequences of exposure to military readiness activities.

It should be noted that, in all the population consequence models discussed above, many assumptions were made, and many input variables were unknown and so were estimated using data when available. It is not possible to estimate long-term or population-level effects from individual short-term behavioral responses alone.

D.7 REPTILES

D.7.1 HEARING AND VOCALIZATION

Sea turtle ears are adapted for hearing underwater and in air, with auditory structures that may receive sound via bone conduction (Lenhardt et al., 1985), resonance of the middle ear cavity (Willis et al., 2013), or the standard tympanic middle ear path (Hetherington, 2008). In-water hearing in sea turtles is typically between 50 and 1,600 Hertz (Hz). Maximum hearing sensitivity is between 100 and 400 Hz, and sensitivity rapidly drops off at higher frequencies (Bartol & Ketten, 2006; Martin et al., 2012; Piniak et al., 2012b; Piniak et al., 2016). Sea turtles are also limited to low-frequency hearing in-air, with juveniles hearing between 50 and 800 Hz, with a maximum hearing sensitivity around 300–400 Hz (Bartol & Ketten, 2006; Piniak et al., 2016). Hearing abilities have primarily been studied with sub-adult, juvenile, and hatchling subjects in four sea turtle species, including green (Bartol & Ketten, 2006; Ketten & Moein-Bartol, 2006; Piniak et al., 2016; Ridgway et al., 1969; Yudhana et al., 2010), olive ridley (Bartol & Ketten, 2006), loggerhead (Bartol et al., 1999; Lavender et al., 2014; Martin et al., 2012), and leatherback (Piniak et al., 2012a). Only one study examined the auditory capabilities of an adult sea turtle (Martin et al., 2012); the hearing range of the adult loggerhead turtle was similar to other measurements of juvenile and hatchling sea turtle hearing ranges.

The role of underwater hearing in sea turtles is unclear. Sea turtles may use acoustic signals from their environment as guideposts during migration and as cues to identify their natal beaches (Lenhardt et al., 1983). However, they may rely more on other senses, such as vision and magnetic orientation, to interact with their environment (Avens, 2003; Lohmann & Lohmann, 2019; Narazaki et al., 2013; Putman et al., 2015). Hearing may also be used for intra-specific communication in water (Charrier et al., 2022) and in air, including hatching synchronization and nest emergence (Cook & Forrest, 2005; Ferrara et al., 2014; Ferrara et al., 2019; McKenna et al., 2019; Mrosovsky, 1972).

All best-available underwater sea turtle AEP and behavioral hearing threshold data from the scientific literature were considered to develop a composite sea turtle audiogram for underwater hearing (Figure D.7-1). An overview of the data used, and the methods to develop a composite sea turtle audiogram for underwater hearing are described in the *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase IV) technical report* (U.S. Department of the Navy, 2024a).

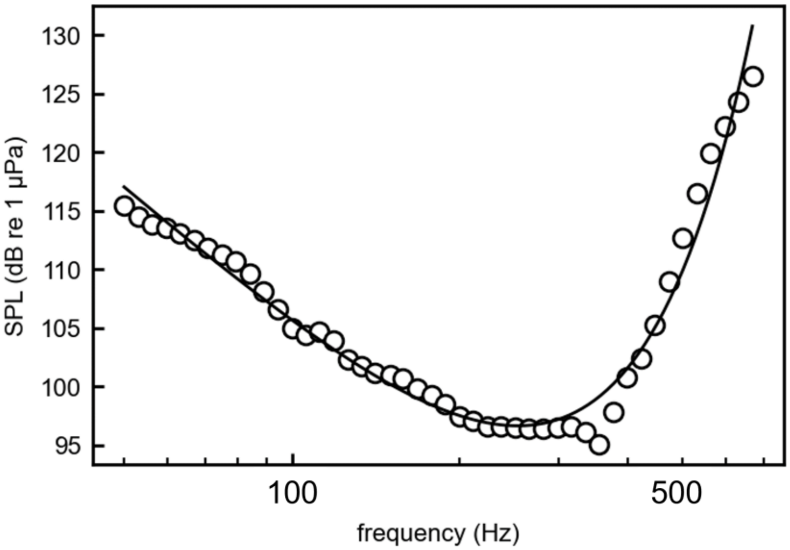


Figure D.7-1: Composite Audiogram used in Sea Turtle Criteria and Thresholds (U.S. Department of the Navy, 2024a)

Some in-air sounds have been recorded during nesting activities ashore, including belch-like sounds and sighs, exhale/inhales, gular pumps, and grunts by female leatherback turtles, and low-frequency pulsed and harmonic sounds by hawksbill, Olive Ridley, Kemp's Ridley, leatherback, and green sea turtle embryos in eggs and hatchlings (Cook & Forrest, 2005; Ferrara et al., 2014; Ferrara et al., 2019; McKenna et al., 2019; Mrosovsky, 1972). Underwater vocalizations from juvenile green turtles have been characterized as pulses, low amplitude calls, frequency modulated sounds, and squeaks (Charrier et al., 2022). Croaks and squeaks have components that are outside the known frequency bandwidth of green turtle hearing and may only be partially detectable (Charrier et al., 2022). These vocalizations were not associated with a specific behavior or the presence of another sea turtle, but there are similarities in vocalizations from freshwater turtles and hatchling Kemp's ridley turtles (Ferrara et al., 2019; Giles et al., 2009).

Crocodilians (e.g., crocodiles and alligators), like other amphibious species, have both in-air and underwater hearing capabilities. However, crocodilians appear to be structurally adapted for detection of airborne sound based on the similarities between crocodilian and avian ear morphology and the corresponding auditory brainstem structures (Gleich & Manley, 2000). Crocodilians detect airborne sound via the tympanic membrane, while sounds in water appear to be detected via bone conduction (Higgs et al., 2002). Crocodilians have external muscular flaps both above and below the opening of the external auditory canal that reflexively close to seal off the canal when submerged and relax to open above/out of the water (Saunders et al., 2000; Shute & Bellairs, 1955).

Crocodilian hearing is most sensitive at low frequencies, both in air and in water. Hearing abilities have not been studied for adult crocodilians but have been studied in juveniles. Best hearing range in air was found between 1 and 1.5 kilohertz (kHz), with poor sensitivity above 2 kHz (Higgs et al., 2002; Wever, 1971). In water, best sensitivity was observed at 800 Hz, with no responses to exposures at 4 kHz (Higgs et al., 2002). The hearing range for crocodilians was observed to extend to higher frequencies in air than in water (Higgs et al., 2002). Crocodilians use hearing for prey detection and social communication, but also rely on good vision, scent, and touch for interacting with their environment (Grigg & Gans, 1993; Wever, 1971). Regarding sound production, crocodilian calls are typically low-frequency (i.e., below 1 kHz), short, and repetitive. Adult calls include courtship bellows at the air-water interface with a notable in-water component, grunts, hisses during threat displays, and coughs (Garrick et al., 1978; Vergne et al., 2009; Vliet, 1989). Hatchling and juvenile American alligators have a more restricted communication repertoire (Higgs et al., 2002). Sound production includes contact calls, hisses, and snarls (Bierman et al., 2014).

No definitive research is available to ascertain how terrapins use sound in the environment. Hearing may be used to locate food or mates, avoid predators, navigate, or communicate (Lester, 2012). Lester et al. (2012) determined that diamondback terrapins can hear a limited range of low-frequency tones less than 1,000 Hz. Terrapins responded to in-air sounds from 100 to 1,000 Hz, with the range of best hearing from 400 to 600 Hz with mean lowest threshold of 64 dB re 20 µPa SPL (Lester, 2012). In-water, terrapins responded to sounds from 50 to 800 Hz with mean lowest threshold of 86 dB re 1 µPa SPL (Lester, 2012).

Sea turtles, crocodilians, and terrapins have similar hearing capabilities, mechanisms, and likely usage. Therefore, the types of impacts to crocodilians and terrapins are assessed to be comparable to those for sea turtles.

D.7.2 HEARING LOSS AND AUDITORY INJURY

A Working Group organized under the ANSI-Accredited Standards Committee S3, Subcommittee 1, Animal Bioacoustics, developed sound exposure guidelines for fishes and sea turtles (Popper et al., 2014), hereafter referred to as the *ANSI Sound Exposure Guideline Technical Report*. The guidelines do not include numeric sound exposure thresholds for auditory effects on sea turtles rather, they qualitatively estimate that sea turtles are less likely to incur TTS or AINJ with increasing distance from various sound sources. Sea turtle hearing is most sensitive around 100–400 Hz in-water and is limited over 1 kHz (Bartol & Ketten, 2006; Martin et al., 2012; Piniak et al., 2012b; Piniak et al., 2016). Therefore, sound exposures from most mid-frequency and all high-frequency sound sources are not anticipated to affect sea turtle hearing, and sea turtles are likely only susceptible to auditory impacts when exposed to very high levels of sound within their limited hearing range. No studies have measured TTS or AINJ in sea turtles, however, TTS in freshwater turtles has been examined (Salas et al., 2023, 2024). Onset values of TTS for freshwater turtles (Salas et al., 2023, 2024) were extrapolated to determine a TTS onset level for non-impulsive sources in sea turtles (U.S. Department of the Navy, 2024a). Consistent with methods from prior analyses, (U.S. Department of the Navy, 2017c) TTS onset levels for non-impulsive sources were used to determine AINJ for non-impulsive sound sources, and onset levels for impulsive sources (U.S. Department of the Navy, 2024a).

D.7.3 MASKING

Reptiles likely use their hearing to detect broadband low-frequency sounds in their environment so the potential for masking would be limited to sound exposures that have similar characteristics (i.e., frequency, duration, and amplitude). Continuous and near-continuous human-generated sounds that have a significant low-frequency component, are not brief, and are of sufficient received level, are most likely to result in masking (e.g., proximate vessel noise and high-duty cycle or continuous active sonar). Other intermittent, short-duration sound sources with low-frequency components (e.g., air guns, pile driving, aircraft noise, weapons noise, and explosives) would have limited potential for masking.

Because reptiles may rely primarily on senses other than hearing for interacting with their environment, any effect of masking may be mediated by reliance on other environmental inputs.

D.7.4 BEHAVIORAL REACTIONS

Behavioral responses fall into two major categories: Alterations in natural behavioral patterns and avoidance. These types of reactions are not mutually exclusive, and reactions may be combinations of behaviors or a sequence of behaviors. The response of a sea turtle to an anthropogenic sound would likely depend on the frequency, duration, temporal pattern, and amplitude of the sound as well as the animal's prior experience with the sound and the context in which the sound is encountered (i.e., what the animal is doing at the time of the exposure) (Ellison et al., 2011; Southall et al., 2021; Wartzok et al., 2003). Distance from the sound source and whether it is perceived as approaching or moving away may also affect a sea turtle's response.

In the *ANSI Sound Exposure Guideline Technical Report* (Popper et al., 2014), qualitative risk factors were developed to assess the potential for sea turtles to respond to various underwater sound sources. The guidelines state that there is a low likelihood that sea turtles would respond within tens of meters of low-frequency sonars, and that it is highly unlikely that sea turtles would respond to mid-frequency sources. The risk that sea turtles would respond to other broadband sources, such as shipping, is considered high within tens of meters of the sound source, but moderate to low at farther distances.

D.7.4.1 Behavioral Reactions to Sonar and Other Transducers

Studies of reptile responses to sonar and other transducers are limited and all data are from studies with sea turtles. Lenhardt (1994) used very low-frequency vibrations (less than 100 Hz) coupled to a shallow tank to elicit swimming behavior responses by two loggerhead sea turtles in which turtles swam to the surface and remained at the surface or slightly submerged. The limitations of conducting acoustic experiments in shallow tanks are discussed in Section D.1.5, Acoustic Propagation in Small Tanks. Watwood et al. (2016) tagged green sea turtles with acoustic transponders and monitored them using acoustic telemetry arrays in Port Canaveral, Florida. Sea turtles were monitored before, during, and after a routine pier-side submarine sonar test that utilized typical source levels, signals, and duty cycle. The authors concluded that no significant long-term displacement was exhibited by the sea turtles in this study. The authors note that Port Canaveral is an urban marine habitat and that resident sea turtles may be less likely to respond than naïve populations.

Kastelein et al. (2023) exposed two green and two hawksbill sea turtles to a wide variety of potential acoustic deterrent signals (> 200 Hz) including Helicopter Long-Range Active Sonar (HELRAS) down sweeps (1.3 to 1.44 kHz). The authors concluded that no behavioral responses were observed to the HELRAS, pure tones, impulsive sounds, or killer whale vocalizations, at levels of approximately 173 dB re 1 µPa. Behavioral responses were observed to eighteen different sounds with various spectro-temporal characteristics, duty cycles and received levels. Of those, four sound types with Navy-relevant signal characteristics (frequency modulated and upsweep). However, no consistent relationship between signal level and behavioral response was observed, and contextual factors appeared to explain some of these responses. The baseline behavioral state of the sea turtle appeared to influence the likelihood of a response, with bottom-resting sea turtles exhibiting little to no responses. The reverberant, shallow environment of the testing pool, minimal controls in the experimental design, and absence of behavioral responses to impulsive sounds suggests that the results of this study should be interpreted with caution, and do not necessitate any changes to the criterion for sonar.

According to the qualitative risk factors developed in the *ANSI Sound Exposure Guideline Technical Report* (Popper et al., 2014), the likelihood of sea turtles responding to low- and mid-frequency sonar is low and highly unlikely, respectively. Based on the limited behavioral response data discussed above, behavioral responses to non-impulsive sounds could consist of temporary avoidance, increased swim speed, or no observable response.

D.7.4.2 Behavioral Reactions to Vessel Noise

There is limited information on reptile behavioral responses to vessel noise. Diaz et al. (2023) quantified the behavioral responses of free-ranging green turtles to vessel noise using audio, video, and positional data from devices mounted to the carapace. Data were collected in the presence and absence of vessel noise while turtles were either traveling or resting on the sea floor. During exposures to vessel noise, existing behaviors were amplified, and the time spent traveling or at the sea floor increased. In addition, more time was spent scanning during traveling when vessel noise was present, which may indicate increased vigilance to detect potential threats. This supports the findings from Hazel et al. (2007) in which turtles avoided vessels more quickly when there was good visibility. In contrast, the amount of time spent scanning while at the sea floor did not significantly increase when vessel noise was present. While at the sea floor vessels may not be perceived as an immediate threat or vessel noise may not be detectable.

Based on the limited behavioral response data discussed above, behavioral responses to vessel noise could include amplification of existing behaviors, increased vigilance, or no observable response.

D.7.4.3 Behavioral Reactions to Aircraft Noise

Behavioral reactions due to aircraft noise, including hovering helicopters, are likely to be brief and minor, if they occur at all. Reptile reactions to aircraft noise have not been studied like marine mammals. For marine mammals, aircraft noise would cause only small temporary changes in behavior. Since reptile hearing is less sensitive than marine mammals, conservatively, it is likely that reptiles could exhibit temporary changes in behavior to aircraft noise as well.

D.7.4.4 Behavioral Reactions to Impulsive Sound Sources

There are limited studies of reptile responses to sounds from impulsive sound sources, and all data come from sea turtles exposed to seismic air guns. These exposures consist of multiple air gun shots, either in close proximity or over long durations, so it is likely that observed responses may over-estimate responses to single or short-duration impulsive exposures. Studies of responses to air guns are used to inform reptile responses to other impulsive sounds (e.g., weapon noise and explosions).

O'Hara and Wilcox (1990) attempted to create a sound barrier at the end of a canal using seismic air guns. They reported that loggerhead turtles kept in a 300 m by 45 m enclosure in a 10-m deep canal and maintained a minimum standoff range of 30 m from air guns fired simultaneously at intervals of 15 seconds with strongest sound components in the 25 to 1,000 Hz frequency range. McCauley et al. (2000a) estimated that the received SPL at which turtles avoided sound in the O'Hara and Wilcox (1990) experiment was 175 to 176 dB re 1 µPa.

Moein Bartol et al. (1995) investigated the use of air guns to repel juvenile loggerhead sea turtles from hopper dredges. Sound frequencies of the air guns ranged from 100 to 1,000 Hz at three source SPLs: 175, 177, and 179 dB re 1 µPa at 1 m. The turtles avoided the air guns during the initial exposures (mean range of 24 m), but additional exposures on the same day and several days afterward did not elicit avoidance behavior that was statistically significant. They concluded that this was likely due to habituation.

McCauley et al. (2000a) exposed a caged green and a caged loggerhead sea turtle to an approaching-departing single air gun to gauge behavioral responses. The trials showed that above a received SPL of 166 dB re 1 µPa, the turtles noticeably increased their swimming activity compared to nonoperational periods, with swimming time increasing as air gun SPLs increased during approach. Above 175 dB re 1 µPa, behavior became more erratic, possibly indicating the turtles were in an agitated state. The authors noted that the point at which the turtles showed more erratic behavior and exhibited possible agitation would be expected to approximate the point at which active avoidance to air guns would occur for unrestrained turtles.

No obvious avoidance reactions by free-ranging sea turtles, such as swimming away, were observed during a multi-month seismic survey using air gun arrays, although fewer sea turtles were observed when the seismic air guns were active than when they were inactive (Weir, 2007). Weir (2007) noted that sea state and the time of day affected both air gun operations and sea turtle surface basking behavior, making it difficult to draw conclusions from the data. However, DeRuiter and Doukara (2012) noted several possible startle or avoidance reactions to a seismic air gun array in the Mediterranean by loggerhead turtles that had been motionlessly basking at the water surface.

Based on the limited behavioral response data discussed above, reptile behavioral responses to impulsive sounds could consist of temporary avoidance, increased swim speed, or changes in depth; or there may be no observable response.

D.7.5 PHYSIOLOGICAL RESPONSE

A stress response is a suite of physiological changes meant to help an organism mitigate the impact of a stressor. If the magnitude and duration of the stress response is too great or too long, then it can have negative consequences to the animal (e.g., decreased immune function, decreased reproduction). Physiological stress is typically analyzed by measuring stress hormones, other biochemical markers, or vital signs. Physiological stress (e.g., corticosterone, glucose, total white blood cell count, and heterophil/lymphocyte ratio) has been measured for sea turtles during nesting (Arango et al., 2022; Flower et al., 2015; Valverde et al., 1999; Vasquez-Bultron et al., 2021), capture and handling (Flower et al., 2015; Gregory & Schmid, 2001; Usategui-Martin et al., 2021), transport (Hunt et al., 2019; Hunt et al., 2020), rehabilitation (Caliani et al., 2019), and when caught in entanglement nets (Hoopes et al., 2000; Miguel et al., 2020; Snoddy et al., 2009) and trawls (Stabenau et al., 1991). However, the stress caused by acoustic exposure has not been studied for sea turtles. Therefore, the stress response in sea turtles in the Study Area due to acoustic exposures is considered to be consistent with general knowledge about physiological stress responses described in the Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities.

Marine animals naturally experience stressors within their environment and as part of their life histories. Changing weather and ocean conditions, exposure to diseases and naturally occurring toxins, lack of prey availability, social interactions with members of the same species, nesting, and interactions with predators all contribute to stress (Atkinson et al., 2015). Anthropogenic sound-producing activities have the potential to provide additional stressors beyond those that naturally occur (Fair et al., 2014; Meissner et al., 2015; Rolland et al., 2012).

Due to the limited information about acoustically induced stress responses for reptiles, the Action Proponents conservatively assume in its effects analysis that any physiological response (e.g., hearing loss or injury) or significant behavioral response is also associated with a stress response.

D.7.6 DIRECT INJURY DUE TO SONAR

The high peak pressures close to some non-impulsive underwater sound sources may be injurious, although there are no reported instances of injury to reptiles caused by these sources. Lacking any data on non-auditory sea turtle injuries due to sonar, *ANSI Sound Exposure Guideline Technical Report* (Popper et al., 2014) estimated the risk to sea turtles from low-frequency sonar to be low and mid-frequency sonar to be non-existent. Additionally, sea turtle carapaces (i.e., shells) may protect against non-auditory injury due to exposures to high peak pressures (Popper et al., 2014), which can also be assumed for terrapins.

Mechanisms for non-auditory injury due to acoustic exposure have been hypothesized for diving breath-hold animals. Acoustically induced bubble formation, rectified diffusion, and acoustic resonance of air cavities are considered for their similarity to pathologies observed in marine mammals stranded coincident with sonar exposures but were found to not be likely causal mechanisms, and findings are applicable to reptiles.

Nitrogen decompression due to modifications to dive behavior in response to sonar exposures has never been observed in sea turtles. Sea turtles are thought to deal with nitrogen loads in their blood and other tissues, caused by gas exchange from the lungs under conditions of high ambient pressure during diving, through anatomical, behavioral, and physiological adaptations (Lutcavage & Lutz, 1997). Although diving sea turtles experience gas supersaturation, gas embolism has only been observed in sea turtles bycaught in fisheries, including loggerhead sea turtles (Garcia-Parraga et al., 2014), as well as leatherback, green, and olive ridley sea turtles (Crespo-Picazo et al., 2020). Therefore, nitrogen decompression due to changes in diving behavior in response to sonar exposures is not considered a potential consequence to diving reptiles.

D.7.7 DIRECT INJURY DUE TO EXPLOSIVES

Data on observed injuries to reptiles from explosions is generally limited to animals found following explosive removal of offshore structures (Viada et al., 2008), which can attract reptiles for feeding or shelter (Klima et al., 1988; Viada et al., 2008). Klima et al. (1988) observed a turtle mortality subsequent to an oil platform removal blast, although sufficient information was not available to determine the animal's exposure. Klima et al. (1988) also placed small sea turtles (less than 7 kg) at varying distances from piling detonations. Some of the turtles were immediately knocked unconscious or exhibited vasodilation (i.e., expansion of blood vessels), but others at the same exposure distance exhibited no effects. Vasodilation was present around the throat and flippers for 2-3 weeks and the increase in blood flow helped to repair damaged cells and tissue. Unconsciousness renders a turtle more susceptible to predation and may result in sinking to the bottom. Although resting turtles can remain submerged for hours, the effects of submergence on stunned turtles are unknown. These data also verified that explosions could result in both near- and far- field injuries to turtles.

Incidental injuries to sea turtles due to military explosions have been documented in a few instances. In one incident, a single 1,200 lb. trinitrotoluene (TNT) underwater charge was detonated off Panama City, Florida, in 1981. The charge was detonated at a mid-water depth of 120 ft. Although details are limited, the following were recorded: at a distance of 500–700 ft., a 400 lb. sea turtle was killed; at 1,200 ft., a 200 to 300 lb. sea turtle experienced "minor" injury; and at 2,000 ft. a 200 to 300 lb. sea turtle was not injured (O'Keeffe & Young, 1984). In another incident, two "immature" green sea turtles (size unspecified) were killed when 100 to 150 ft. away from detonation of 20 lb. of C-4 in a shallow-water environment. This illustrates that the likelihood and types of injuries from underwater explosives depends on the charge size, the geometry of the exposure (distance to the charge, depth of the animal and the charge), and the size of the animal.

D.7.8 LONG-TERM CONSEQUENCES

For reptiles present in the Study Area, long-term consequences to individuals and populations due to acoustic exposures have not been studied. For this analysis it is assumed that long term-consequences to reptiles are consistent with general knowledge about long-term consequences to other marine species.

Long-term consequences to reptile populations due to disturbances, whether anthropogenic or environmental, are difficult to assess. Linking non-lethal effects on individuals to changes in population growth rates requires long-term data, which is lacking for many populations. The long-term consequences of hearing loss, chronic masking, and short-term or chronic physiological stress are especially difficult to predict because of the different factors experienced by individual animals, such as context of stressor exposure, underlying health conditions, and other environmental or anthropogenic stressors.

An important variable to consider is duration of disturbance. Severity scales used to assess behavioral responses to acute sound exposures are not appropriate to apply to sustained or repeated (chronic) exposures, as the focus has shifted from the immediate impacts to an individual to the health of a population over time (Southall et al., 2021). For example, short-term costs experienced over the course of a week by an otherwise healthy individual may be recouped over time after exposure to the stressor ends. These short-term costs would be unlikely to result in long-term consequences to that individual or to that individual's population. Comparatively, long-term costs accumulated by otherwise healthy individuals over an entire season, year, or throughout a life stage would be less easily recouped and more likely to result in long-term consequences to that individual or population.

Reptiles exposed to frequent or intense human activities may leave the area, habituate to the activity, or tolerate the disturbance and remain in the area (Wartzok et al., 2003). Highly resident or localized populations may also stay in an area of disturbance because the cost of displacement may be higher than the cost of remaining (Forney et al., 2017). An apparent lack of response (e.g., no displacement or avoidance of a sound source) may not necessarily mean there is no cost to the individual or population, as some resources or habitats may be of such high value that animals may choose to stay, even when experiencing the consequences of stress, masking, or hearing loss (Forney et al., 2017).

Longer term displacement can lead to changes in abundance or distribution patterns of the species in the affected region (Bejder et al., 2006b; Blackwell et al., 2004; Teilmann et al., 2006). Predicting population trends or long-term displacement patterns due to anthropogenic disturbance is challenging due to limited information and survey data for many species over sufficient temporal and spatial scales, as well as a full understanding of how other factors, such as oceanographic oscillations and climate change, affect presence.

D.8 BIRDS AND BATS

This section presents a review of existing literature on acoustic impacts to birds and bats. Assessing whether sounds may disturb or injure an animal involves understanding the characteristics of the acoustic sources and the effects that sound may have on the physiology and behavior of those animals. Impacts could depend on other factors besides the received level of sound, such as the animal's physical condition and prior experience with the sound. Additional explanation of the acoustic terms and sound energy concepts used in this section is found in Section D.1 (Acoustic and Explosive Concepts/Primer).

Following a review of recent literature, the data on acoustic impacts to birds has not changed appreciably. As such, the information presented in the 2018 Final EIS/OEIS Section 3.9.3.1 (Acoustic Stressors) remains valid. Studies since that time generally support conclusions and those have been added to the sections below as appropriate.

D.8.1 HEARING AND VOCALIZATION

D.8.1.1 Birds

Although hearing range and sensitivity has been measured for many land birds, fewer studies have focused on seabird hearing and most published literature on bird hearing focuses on terrestrial birds and their ability to hear in air. A review of 32 terrestrial and marine species indicates that birds generally have the greatest hearing sensitivity between 1 and 4 kHz (Beason, 2004; Dooling, 2002). Very few can hear below 20 Hz, most have an upper frequency hearing limit of 10 kHz, and none exhibit hearing at frequencies higher than 15 kHz (Dooling, 2002; Dooling & Popper, 2000). Hearing capabilities have been studied for several seabirds (Beason, 2004; Beuter et al., 1986; Crowell et al., 2015; Johansen et al., 2016; Thiessen, 1958; Wever et al., 1969); these studies show that seabird hearing ranges and sensitivity in air are consistent with what is known about bird hearing in general. Several studies of seabird hearing have been published since the 2018 Final EIS/OEIS and support previous work. These newer studies of long-tailed duck, common eider, and surf scoter (McGrew et al., 2022); and Atlantic puffin and common murre (Mooney et al., 2019) support previous conclusions that birds generally have greatest hearing sensitivity between 1 and 4 kHz. Two field studies (Mooney et al., 2020; Mooney et al., 2019) of wild captured Atlantic puffin produced auditory curves between 0.5 and 6 kHz, similar to measurements for other seabirds. Smith et al. (2023) measured the same range in marbled murrelet. Hansen et al. (2017) studied great cormorants and found maximum sensitivity at 2 kHz and Larsen et al. (2020) found that great cormorants have underwater hearing sensitivity that is at least as good as their aerial sensitivity along with anatomical adaptations to underwater hearing (thickened eardrum).

Crowell et al. (2015) also compared the vocalizations of the same ten diving bird species to the region of highest sensitivity of in-air hearing. Of the birds studied, vocalizations of only eight species were obtained due to the relatively silent nature of two of the species. The peak frequency of the vocalizations of seven of the eight species fell within the range of highest sensitivity of in-air hearing. Crowell et al. (2015) suggested that the colonial nesters tested had relatively reduced hearing sensitivity because they relied on individually distinctive vocalizations over short ranges. Additionally, Crowell et al. (2015) observed that the species with more sensitive hearing were those associated with freshwater habitats, which are quieter compared to marine habitats with wind and wave noise.

Although important to seabirds in air, it is unknown if seabirds use hearing or vocalizations underwater for foraging, communication, predator avoidance or navigation (Crowell, 2016; Dooling & Therrien, 2012).

Diving birds may not hear as well underwater, compared to other (non-avian) species, based on adaptations to protect their ears from pressure changes (Dooling & Therrien, 2012). Because reproduction and communication with conspecifics occurs in air, adaptations for diving may have evolved to protect in-air hearing ability and may contribute to reduced sensitivity underwater (Hetherington, 2008). Hansen et al. (2020) observed that common murres (*Uria aalge*) consistently reacted to sounds produced by underwater sound bursts and mid-frequency sonar signals. There are many anatomical adaptations in diving birds that may reduce sensitivity both in air and underwater. Anatomical ear adaptations include cavernous tissue in the meatus and middle ear that may fill with blood during dives to compensate for increased pressure on the tympanum, active muscular control of the meatus to prevent water entering the ear, and interlocking feathers to create a waterproof outer covering (Crowell et al., 2015; Rijke, 1970; Sade et al., 2008). Zeyl et al. (2022) used nano-CT scanning of the ears of 127 bird species to measure the morphological adaptations to aerial and underwater hearing. Pursuit and deep diving species have heavily modified middle ears including smaller tympanic membranes and columella footplate of the middle ear middle ear; shorter extrastapedius, as well as reduced cranial air volume and connectivity. These likely facilitating hearing underwater and provide baroprotection, while potentially constraining the sensitivity of aerial hearing.

The northern gannet, a plunge diver, has unique adaptations to hitting the water at high speeds, including additional air spaces in the head and neck to cushion the impact and a thicker tympanic membrane than similar sized birds (Crowell et al., 2015). All these adaptations could explain the measured higher thresholds of diving birds.

D.8.1.2 Bats

The hearing range of insect-eating bats in North America is 10 to 100 kHz. The most sensitive frequency band is 20 to 50 kHz, where bats can detect sounds at approximately 20 dB re 20 µPa (Bohn et al., 2006; Koay et al., 1997). Bats are generally unable to hear frequencies below 500 Hz. While hearing is the primary sense used by echolocating bats to forage and avoid obstacles, they use a combination of auditory and visual landmark recognition (Denzinger & Schnitzler, 2013; Gonzalez-Terrazas et al., 2016; Jensen et al., 2005; Schnitzler et al., 2003), magneto-reception (Holland et al., 2008; Holland et al., 2006; Wang et al., 2007), and spatial memory for long-distance navigation (Barchi et al., 2013; Ulanovsky & Moss, 2008, 2011; William & Williams, 1970; Williams et al., 1966). The variety of vocalizations produced by bats can be separated into two general categories: ultrasonic echolocation sounds and communication sounds. Echolocation is used while foraging, in which bats listen for received echoes from insect targets. Sound detection levels are somewhat dependent on ambient noise, and bats increase the loudness of their calls when they encounter noise (Hage et al., 2013; Hotchkin & Parks, 2013; Luo & Wiegrebe, 2016).

D.8.2 HEARING LOSS AND AUDITORY INJURY

Exposure to intense sound may result in hearing loss which persists after cessation of the noise exposure. Hearing loss may be temporary or permanent, depending on factors such as the sound's frequency, received SPL, and duration. Hearing loss could impair a bird's or a bat's ability to hear biologically important sounds within the affected frequency range. Biologically important sounds include communication with social groups, potential mates, offspring, and parents as well as environmental sounds and sounds made by prey and predators.

Studies of in-air noise induced hearing loss and recovery are limited (e.g., quail, budgerigars, canaries, and zebra finches) (Ryals et al., 1999). There are no studies of bird hearing loss from underwater sound exposures. The frequencies affected by hearing loss would vary depending on the exposure frequency. The limited data on hearing loss in birds show that the frequency of exposure is the hearing frequency most likely to be affected (Saunders & Dooling, 1974).

Hearing loss can result from tissue damage. Tissue damage can include damage to the auditory hair cells and their underlying support cells. Hair cell damage has been observed in birds exposed to long-duration sounds that resulted in initial threshold shifts greater than 40 dB (Niemiec et al., 1994; Ryals et al., 1999). Birds can regenerate hair cells in the ear, usually resulting in considerable anatomical, physiological, and behavioral recovery within several weeks (Rubel et al., 2013; Ryals et al., 1999). Still, intense exposures are not always fully recoverable, even over periods up to a year after exposure, and damage and subsequent recovery vary significantly by species (Ryals et al., 1999). Birds may be able to protect themselves against damage from sustained sound exposures by reducing middle ear pressure, an ability that may protect ears while in flight (Ryals et al., 1999) and from injury due to pressure changes during diving (Dooling & Therrien, 2012).

Studies in mammals have revealed that noise exposures resulting in high levels of TTS (greater than 40 dB) may also result in neural injury without any permanent hearing loss (Kujawa & Liberman, 2009; Lin et al., 2011). It is unknown if a similar effect would be observed for birds.

D.8.2.1 Hearing Loss due to Non-Impulsive Sounds

D.8.2.1.1 Birds

Behavioral studies of threshold shift in birds within their frequencies of best hearing (between 2 and 4 kHz) due to long-duration (30 minutes to 72 hours) continuous, non-impulsive, high-level sound exposures in air have shown that susceptibility to hearing loss varies substantially by species, even in species with similar auditory sensitivities, hearing ranges, and body size (Niemiec et al., 1994; Ryals et al., 1999; Saunders & Dooling, 1974). For example, Ryals et al. (1999) conducted the same exposure experiment on quail and budgerigars, which have very similar audiograms. A 12-hour exposure to a 2.86 kHz tone at 112 dB re 20 µPa SPL (cumulative SEL of 158 dB re 20 $\mu Pa^2 s$) resulted in a 70 dB threshold shift measured after 24 hours of recovery in quail, but a substantially lower 40 dB threshold shift measured after just 12 hours of recovery in budgerigars which recovered to within 10 dB of baseline after three days and fully recovered by one month (Ryals et al., 1999). Whereas the 158 dB re 20 $\mu Pa^2 s$ SEL tonal exposure to quail discussed above caused 20 dB of PTS (Ryals et al., 1999), a shorter (four-hour) tonal exposure to quail with similar SEL (157 dB re 20 $\mu Pa^2 s$) caused 65 dB of threshold shift that fully recovered within two weeks (Niemiec et al., 1994).

Data on threshold shift in birds due to relatively short-duration sound exposures that could be used to estimate the onset of threshold shift is limited. Saunders and Dooling (1974) provide the only threshold shift growth data measured for birds. Saunders and Dooling (1974) exposed young budgerigars to four levels of continuous 1/3-octave band noise (76, 86, 96, and 106 dB re 20 µPa) centered at 2.0 kHz and measured the threshold shift at various time intervals during the 72-hour exposure. The earliest measurement found 7 dB of threshold shift after approximately 20 minutes of exposure to the 96 dB re 20 µPa SPL noise (127 dB re 20 $\mu Pa^2 s$ SEL). Generally, onset of TTS in other species has been considered 6 dB above measured threshold (Finneran, 2015), which accounts for natural variability in auditory thresholds. The Saunders and Dooling (1974) budgerigar data is the only bird data showing low levels of threshold shift. Because of the observed variability of threshold shift susceptibility among bird species and the relatively long duration of sound exposure in Saunders and Dooling (1974), the observed onset level cannot be assumed to represent the SEL that would cause onset of TTS for other bird species or for shorter-duration exposures (i.e., a higher SEL may be required to induce TTS for shorter-duration exposures).

Since the goal of most bird hearing studies has been to induce hair cell damage to study regeneration and recovery, exposure durations were purposely long. Studies with other non-avian species have shown that long-duration exposures tend to produce more threshold shift than short-duration exposures with the same SEL (e.g., see Finneran (2015). The SELs that induced TTS and PTS in these studies likely over-estimate the potential for hearing loss due to any short-duration sound of comparable SEL that a bird could encounter outside of a controlled laboratory setting. In addition, these studies were not designed to determine the exposure levels associated with the onset of any threshold shift or to determine the lowest SEL that may result in PTS.

With insufficient data to determine PTS onset for birds due to a non-impulsive exposure, data from other taxa are considered. Studies of terrestrial mammals suggest that 40 dB of threshold shift is a reasonable estimate of where PTS onset may begin (Southall et al., 2009). Similar amounts of threshold shift have been observed in some bird studies with no subsequent PTS. Of the birds studied, the budgerigars showed intermediate susceptibility to threshold shift; the budgerigars exhibited threshold shifts in the range of 40 dB to 50 dB after 12-hour exposures to 112 dB and 118 dB re 20 µPa SPL tones at 2.86 kHz (158 to 164 dB re 20 $\mu Pa^2 s$ SEL), which recovered to within 10 dB of baseline after three days and fully recovered by one month (Ryals et al., 1999). These experimental SELs are a conservative estimate of the SEL above which PTS may be considered possible for birds.

All of the above studies were conducted in air. There are no studies of hearing loss to diving birds due to underwater exposures.

D.8.2.1.2 Bats

Bats exposed to loud noise have not been shown to exhibit TTS (Hom et al., 2016; Simmons et al., 2015; Simmons et al., 2016). Recently, Hom et al. (2016) exposed four big brown bats (*Eptesicus fuscus*) to intense broadband noise (10 to 100 kHz with SEL 152 dB re 20 μPa^{2}s over one hour) and found no effect on the bats' vocalizations (which could indicate a change in hearing) or psychophysical thresholds 20 minutes, 24 hours, or 48 hours after exposure (Hom et al., 2016; Simmons et al., 2016). Another study on the Japanese house bat (*Pipistrellus abramus*) measured physiological (auditory brainstem response) thresholds immediately after a noise exposure (10 to 80 kHz, 90 dB re 20 μPa SPL, 30-minute duration) and also did not find evidence of TTS (Simmons et al., 2015). This may be because bats are adapted to hear in an acoustic environment where they are likely to experience loud sounds (110 to 140 dB re 20 μPa SPL) continuously for several hours while hunting near other bats that are also echolocating (Jakobsen et al., 2013; Simmons et al., 2001). It is also possible that the stimuli used in these experiments were not loud enough to induce TTS or that measurements of hearing sensitivity took place outside the time window where TTS might be observed.

D.8.2.2 Hearing Loss due to Impulsive Sounds

The only measure of hearing loss in a bird due to an impulsive noise exposure was conducted by Hashino et al. (1988), in which budgerigars were exposed to the firing of a pistol with a received level of 169 dB re 20 μPa peak SPL (two gunshots per each ear); SELs were not provided. While the gunshot frequency power spectrum had its peak at 2.8 kHz, threshold shift was most extensive below 1 kHz. Threshold shift recovered at frequencies above 1 kHz, while a 24 dB PTS was sustained at frequencies below 1 kHz. Studies of hearing loss in diving birds exposed to impulsive sounds underwater do not exist.

Because there is only one study of hearing loss in birds due to an impulsive exposure and no studies of hearing loss in bats due to an impulsive exposure, the few studies of hearing loss in birds and bats due to exposures to non-impulsive sound are the only other data upon which to assess bird and bat susceptibility to hearing loss from an impulsive sound source. Data from other taxa (U.S. Department of the Navy, 2017a) indicate that, for the same SEL, impulsive exposures are more likely to result in hearing loss than non-impulsive exposures. This is due to the high peak pressures and rapid pressure rise times associated with impulsive exposures.

D.8.3 MASKING

Masking occurs when one sound interferes with the detection or recognition of another sound. The quantitative definition of masking is the amount in decibels an auditory detection or discrimination threshold is raised in the presence of a masker (Erbe et al., 2016). As discussed in Section 3.0.3.7.1 (Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities), masking can effectively limit the distance over which an animal can communicate and detect biologically relevant sounds. Masking only occurs in the presence of the masking noise and does not persist after the cessation of the noise.

D.8.3.1 Birds

Critical ratios are the lowest ratio of signal-to-noise at which a signal can be detected. When expressed in decibels, critical ratios can easily be calculated by subtracting the noise level (in dB re 1 μPa^2/Hz) from the signal level (in dB re 1 μPa) at detection threshold. A signal must be received above the critical ratio at a given frequency to be detectable by an animal. Critical ratios have been determined for a variety of bird species (e.g., Crowell, 2016; Dooling, 1980; Dooling & Popper, 2000; Noirot et al., 2011) and interspecies variability is evident. Some birds exhibit low critical ratios at certain vocal frequencies, perhaps indicating that hearing evolved to detect signals in noisy environments or over long distances (Dooling & Popper, 2000).

The effect of masking is to limit the distance over which a signal can be perceived. An animal may attempt to compensate in several ways, such as by increasing the source level of vocalizations (the Lombard effect), changing the frequency of vocalizations, or changing behavior (e.g., moving to another location, increasing visual display). Birds have been shown to shift song frequencies in the presence of a tone at a similar frequency (Goodwin & Podos, 2013), and in continuously noisy urban habitats, populations have been shown to have altered song duration and shift to higher frequencies (Slabbekoorn & den Boer-Visser, 2006). Changes in vocalization may incur energetic costs and hinder communication with conspecifics, which, for example, could result in reduced mating opportunities. These effects are of long-term concern in constant noisy urban environments (Patricelli & Blickley, 2006) where masking conditions are prevalent.

D.8.3.2 Bats

Bats can experience masking during echolocation and communication from a variety of sources such as other bats and jamming of their echolocation signal by prey species (Bates et al., 2011; Chiu et al., 2008; Conner & Corcoran, 2012; Corcoran et al., 2009; Griffin et al., 1962; Simmons et al., 1988; Ulanovsky et al., 2004). They have many strategies to compensate for masking, such as dynamically changing the duration, spectrum, aim, and pattern of their echolocation (Bates et al., 2011; Moss et al., 2011; Petrites et al., 2009; Simmons et al., 2001; Wheeler et al., 2016).

Like other animals, bats increase the amplitude of their vocalizations in response to an increase in background noise level, which is known as the Lombard effect (Hage et al., 2013). It is estimated that a broadband signal of 65 dB re 20 µPa SPL would begin masking most bats' echolocation from targets beyond 1.5 m away (Arnett et al., 2013). Bats have been shown to shift the frequency of their calls when a stimulus was within 2 to 3 kHz of their preferred frequency (Bates et al., 2008).

Behavioral and psychophysical experiments show that the flexibility of bat vocalizations allows for perceptual rejection of masking due to clutter in the surroundings (Bates et al., 2011; Hiryu et al., 2010; Warnecke et al., 2015) or other sources of noise (Bates et al., 2008; Miller et al., 2004; Ulanovsky et al., 2004).

Overall, bats seem to avoid areas with high levels of noise—especially when the noise frequency spectrum overlaps with frequencies important for hunting (20 to 90 kHz). In a controlled laboratory experiment, Schaub et al. (2008) found that, when given a choice, bats spent 10 percent less time foraging in the compartment with noise (traffic, wind, and broadband white noise) as compared to the silent control chamber. Additionally, hunting in the noisy compartment yielded 10 percent fewer successful prey interceptions. Bats spent significantly less time and were significantly less successful as noise conditions increased in bandwidth and overall exposure levels. The greater the noise overlap with frequencies being attended to by the bat, the greater the disturbance to the bats' foraging behavior. However, this experiment was conducted on a small spatial scale, and with the absence of other sensory cues (light). Although laboratory research has shown that noise can decrease hunting success (Siemers & Schaub, 2011), and field and laboratory studies show that foraging bats avoid noise (Berthinussen & Altringham, 2012; Schaub et al., 2008), no studies provide direct evidence from playback experiments in the field that commuting or migrating bats are disturbed by sound.

D.8.4 BEHAVIORAL REACTIONS

Hansen et al. (2020) exposed two common murres to broadband sound bursts and mid-frequency active sonar (3.5 to 4.1 kHz frequency range) playback during an underwater foraging task and found that both birds exhibited behavioral reactions to both stimuli as compared to no reactions in control trials. One subject exhibited stronger behavioral reactions to the noise bursts, and the other to the sonar. This effect was found for received levels between 110 and 137 dB re 1 µPa rms and the birds tended to turn or swim away from the sound source. This research suggests that anthropogenic noise within the birds' hearing range may cause behavioral disturbance while foraging underwater, and that they may exhibit avoidance responses. Sørensen et al. (2020) demonstrated that Gentoo penguins (*Pygoscelis papua*) react to noise bursts (0.2 to 6 kHz) by exposing seven individuals while underwater in a pool. Individual penguins received levels between 100 and 120 dB re 1 µPa, but a dose-response relationship between behavioral responses and SPL could not be established from the data. Variability both within and between individuals was observed. The data suggest that Gentoo penguins, a species adapted for pursuing prey underwater, are likely to react to received levels of 120 dB re 1 µPa and higher.

Numerous studies have documented that birds and other wild animals respond to human-made noise, including aircraft overflights, weapons firing, and explosions (Larkin et al., 1996; National Park Service, 1994; Plumpton, 2006). The way an animal responds to noise could depend on several factors, including life history characteristics of the species, characteristics of the noise source, sound source intensity, onset rate, distance from the noise source, presence or absence of associated visual stimuli, food and habitat availability, and previous exposure (see Section 3.0.3.7.1, Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities). Researchers have documented a range of bird behavioral responses to noise, including no response, head turn, alert behavior, startle response, flying or swimming away, diving into the water, and increased vocalizations (Brown et al., 1999; Larkin et al., 1996; National Park Service, 1994; Plumpton, 2006; Pytte et al., 2003; Stalmaster & Kaiser, 1997). Bat behavioral studies have shown reactions in response to acoustic interference such as reduced activity, area avoidance, and modifying the duration or frequency of calls (Arnett et al., 2013; Bates et al., 2008; Baxter et al., 2006). Some behavioral responses may be accompanied by physiological responses, such as increased heart rate or short-term changes in stress hormone levels (Partecke et al., 2006).

Behavioral responses may depend on the characteristics of the noise, and whether the noise is similar to biologically relevant sounds, such as alarm calls by other birds and predator sounds. For example, European starlings (*Sturnus vulgaris*) took significantly longer to habituate to repeated bird distress calls than white noise or pure tones (Johnson et al., 1985). Starlings may have been more likely to continue to respond to the distress because it is a more biologically meaningful sound. Starlings were also more likely to habituate in winter than summer, possibly meaning that food scarcity or seasonal physiological conditions may affect intensity of behavioral response (Johnson et al., 1985).

D.8.4.1 Behavioral Reactions to Impulsive Sound Sources

Studies regarding behavioral responses by non-nesting birds to impulsive sound sources are limited. Seismic surveys had no noticeable impacts on the movements or diving behavior of long-tailed ducks undergoing wing molt, a period in which flight is limited and food requirements are high (Lacroix et al., 2003). The birds may have tolerated the seismic survey noise to stay in preferred feeding areas.

Responses to aircraft sonic booms are informative of responses to single impulsive sounds. Responses to sonic booms are discussed below in Section D.8.4.3 (Behavioral Responses to Aircraft).

D.8.4.2 Behavioral Reactions to Sonar and Other Active Acoustic Sources

There are no studies of bird responses underwater to sonars, but the effect of pingers on fishing nets has been examined. Fewer common murres (*Uria aalge*) were entangled in gillnets when the gillnets were outfitted with 1.5 kHz pingers with a source level of 120 dB re 1 µPa; however, there was no significant reduction in rhinoceros auklet (*Cerorhinca monocerata*) bycatch in the same nets (Melvin et al., 2011; Melvin et al., 1999). It was unknown whether the pingers elicited a behavioral response by the birds or decreased prey availability.

D.8.4.3 Behavioral Responses to Aircraft

There are multiple possible factors involved in behavioral responses of birds and bats to aircraft overflights, including the noise stimulus as well as the visual stimulus.

Observations of tern colonies responses to balloon overflights suggest that visual stimulus is likely to be an important component of disturbance from overflights (Brown, 1990). Although it was assumed nesting colonial waterbirds may be more likely to flush or exhibit a mob response when disturbed, observations of nesting black skimmers and nesting least, gull-billed, and common terns showed they did not modify nesting behavior in response to military fixed-wing aircraft engaged in low-altitude tactical flights and rotary-wing overflights (Hillman et al., 2015). Maximum behavioral responses by crested tern (*Sterna bergii*) to aircraft noise were observed at SEL greater than 85 dBA re 20 µPa. However, herring gulls (*Larus argentatus*) significantly increased their aggressive interactions within the colony and their flights over the colony during overflights with received SPLs of 101 to 116 dBA re 20 µPa (Burger, 1981).

Raptors and wading birds have responded minimally to jet (110 dBA re 20 µPa) and propeller plane (92 dBA re 20 µPa) overflights, respectively (Ellis, 1981). Jet flights greater than 1,640 feet (ft.) distance from raptors were observed to elicit no response (Ellis, 1981). The impacts of low-altitude military training flights on wading bird colonies in Florida were estimated using colony distributions and turnover rates. There were no demonstrated impacts of military activity on wading bird colony establishment or size (Black et al., 1984). Fixed-winged jet aircraft disturbance did not seem to adversely affect waterfowl observed during a study in coastal North Carolina (Conomy et al., 1998); however, harlequin ducks were observed to show increased agonistic behavior and reduced courtship behavior up to one to two hours after low-altitude military jet overflights (Goudie & Jones, 2004).

It is possible that birds could habituate and no longer exhibit behavioral responses to aircraft noise, as has been documented for some impulsive noise sources (Ellis, 1981; Russel et al., 1996) and aircraft noise (Conomy et al., 1998). Ellis (1981) found that raptors would typically exhibit a minor short-term startle response to simulated sonic booms, and no long-term effect to productivity was noted.

Near-total failure of sooty tern nesting in the Dry Tortugas in the Key West Range Complex was reported in 1969 during a period when the birds were regularly exposed to sonic booms (Austin et al., 1970). In previous seasons, the birds were reported to react to the occasional sonic booms by rising immediately in a “panic flight,” circling over the island, and then usually settling down on their eggs again. Researchers had no evidence that sonic booms caused physical damage to the sooty tern eggs but hypothesized that the strong booms occurred often enough to disturb the sooty terns’ incubating rhythm and cause nest desertion. The 1969 sooty tern nesting failure also prompted additional research to test the hypothesis that sonic booms could cause bird eggs to crack or otherwise affect bird eggs or embryos. However, the findings of the additional research determined that aircraft overflight and sonic booms were not a cause of the failure, and neither were panic flights, predators, weather, inadequate food supplies, or tick infestation (Bowles et al., 1991; Bowles et al., 1994a; Teer & Truett, 1973; Ting et al., 2002). That same year, the colony also contained approximately 2,500 brown noddies, whose young hatched successfully. While it was impossible to conclusively determine the cause of the 1969 sooty tern nesting failure, actions were taken to curb planes breaking the sound barrier within range of the Tortugas, and much of the excess vegetation was cleared (another hypothesized contributing factor to the nesting failure). Similar nesting failures have not been reported since the 1969 failure.

D.8.5 PHYSIOLOGICAL RESPONSE

Chronic stress due to disturbance may compromise the general health and reproductive success of birds (Kight et al., 2012), but a physiological stress response is not necessarily indicative of negative consequences to individual birds or to populations (Larkin et al., 1996; National Park Service, 1994). The reported behavioral and physiological responses of birds to noise exposure can fall within the range of normal adaptive responses to external stimuli, such as predation, that birds face on a regular basis. These responses can include activation of the neural and endocrine systems, causing changes such as increased blood pressure, available glucose, and blood levels of corticosteroids (Manci et al., 1988). It is possible that individuals would return to normal almost immediately after short-term or transient exposure, and the individual’s metabolism and energy budget would not be affected in the long-term. Studies have also shown that birds can habituate to noise following frequent exposure and cease to respond behaviorally to the noise (Larkin et al., 1996; National Park Service, 1994; Plumpton, 2006). However, the likelihood of habituation is dependent upon a number of factors, including species of bird (Bowles et al., 1991), and frequency of and proximity to exposure. Although Andersen et al. (1990) did not evaluate noise specifically, they found evidence that anthropogenic disturbance is related to changes in home ranges; for example, raptors have been shown to shift their terrestrial home range when concentrated military training activity was introduced to the area. On the other hand, cardinals nesting in areas with high levels of military training activity (including gunfire, artillery, and explosives) were observed to have similar reproductive success and stress hormone levels as cardinals in areas of low activity (Barron et al., 2012).

While physiological responses such as increased heart rate or startle response can be difficult to measure in the field, they often accompany more easily measured reactions like behavioral responses. A startle is a reflex characterized by rapid increase in heart rate, shutdown of nonessential functions, and mobilization of glucose reserves. Habituation keeps animals from expending energy and attention on harmless stimuli, but the physiological component might not habituate completely (Bowles, 1995).

A strong and consistent behavioral or physiological response is not necessarily indicative of negative consequences to individuals or to populations (Bowles, 1995; Larkin et al., 1996; National Park Service, 1994). For example, many of the reported behavioral and physiological responses to noise are within the range of normal adaptive responses to external stimuli, such as predation, that wild animals face on a regular basis. In many cases, individuals would return to homeostasis or a stable equilibrium almost immediately after exposure. The individual’s overall metabolism and energy budgets would not be affected if it had time to recover before being exposed again. If the individual does not recover before being exposed again, physiological responses could be cumulative and lead to reduced fitness. However, it is also possible that an individual would have an avoidance reaction (i.e., move away from the noise source) to repeated exposure or habituate to the noise when repeatedly exposed.

Due to the limited information about acoustically induced stress responses, the Action Proponents conservatively assume in its effects analysis that any physiological response (e.g., hearing loss or injury) or significant behavioral response is also associated with a stress response.

D.8.6 DIRECT INJURY

Auditory structures are susceptible to injury from high levels of impulsive sound. This could include tympanic membrane rupture, disarticulation of the middle ear ossicles, and trauma to the inner ear structures such as the hair cells within the organ of Corti. Auditory trauma differs from auditory fatigue in that the latter involves the overstimulation of the auditory system, rather than direct mechanical damage, which may result in hearing loss (see Section D.8.2, Hearing Loss and Auditory Injury). There are no data on damage to the middle ear structures of birds resulting from acoustic exposures. Because birds are known to regenerate auditory hair cells, studies have been conducted to purposely expose birds to very high SELs to induce hair cell damage in the inner ear. Because damage can co-occur with fatiguing exposures at high SELs, effects to hair cells are discussed above in Section D.8.2.

Because there are no data on non-auditory injury to birds from intense non-explosive sound sources, it may be useful to consider information for other similar-sized vertebrates. The rapid large pressure change near non-explosive impulsive sound sources, such as some large air guns and pile driving, are thought to be potentially injurious to other small animals (fishes and sea turtles). While long duration exposures (i.e., minutes to hours) to high sound levels of sonars are thought to be injurious to fishes, this has not been experimentally observed (Popper et al., 2014). The potential for injury is generally attributed to compression and expansion of body gas cavities, either due to rapid onset of pressure changes or resonance (enhanced oscillation of a cavity at its natural frequency). Because water is considered incompressible and animal tissue is generally of similar density as water, animals would be more susceptible to injury from a high-amplitude sound source in water than in air since waves would pass directly through the body rather than being reflected. Exposures to high-amplitude non-impulsive sounds underwater could be limited by a bird's surfacing response.

In air, the risk of barotrauma would be associated with high-amplitude impulses, such as from explosives (discussed in Section 3.9.3.2, Explosive Stressors). Unlike in water, most acoustic energy reflects off the surface of an animal's body in air. Additionally, air is compressible whereas water is not, allowing energy to dissipate more rapidly. For these reasons, in-air non-explosive sound sources in this analysis are considered to pose little risk of non-auditory injury.

Limited data exist on instances of barotrauma to bats. Studies of the effects of rapid pressure changes from rotating wind turbine blades have demonstrated instances of ruptured tympana (Baerwald et al., 2008; Rollins et al., 2012). Although it is undetermined if these ruptures were the result of pressure changes or from a direct strike, the potential exists for auditory injury as a result of high-amplitude sound exposure.

D.8.7 LONG-TERM CONSEQUENCES

Long-term consequences to birds and bats due to acoustic exposures are considered following the Conceptual Framework for Assessing Effects from Proposed Action Activities (Section 3.0.3.7.1).

Long-term consequences due to individual behavioral reactions and short-term instances of physiological stress are especially difficult to predict because individual experience over time can create complex contingencies. It is more likely that any long-term consequences to an individual would be a result of costs accumulated over a season, year, or life stage due to multiple behavioral or stress responses resulting from exposures to multiple stressors over significant periods of time. Conversely, some birds and bats may habituate to or become tolerant of repeated acoustic exposures over time, learning to ignore a stimulus that in the past did not accompany any overt threat. Most research on long-term consequences to birds due to acoustic exposures has focused on breeding colonies or shore habitats and does not address the brief exposures that may be encountered during migration or foraging at sea. More research is needed to better understand the long-term consequences of human-made noise on birds and bats, although intermittent exposures are assumed to be less likely than prolonged exposures to have lasting consequences.

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