# Appendix F Acoustic and Non-Acoustic Effects Supporting Information

# Environmental Impact Statement/
# Overseas Environmental Impact Statement
# Hawaii-California Training and Testing Activities
# TABLE OF CONTENTS

## List of Figures

# List of Tables

# APPENDIX F Acoustic and Non-Acoustic Effects Supporting Information

## F.1 Biological Resource Methods

The analysis of effects on biological resources focused on the likelihood of encountering the stressor, the primary stimulus, response, and recovery of individual organisms. Where appropriate, the potential of a biological resource to overlap with a stressor was analyzed with consideration given to the specific geographic area (large marine ecosystems, open ocean areas, range complexes, Study Areas, and other training and testing areas) in which the overlap could occur. Additionally, the differential effects of training versus testing activities that introduce stressors to the resource were considered.

### F.1.1 Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities

This conceptual framework describes the potential effects from exposure to acoustic and explosive activities and the accompanying short-term costs to the animal (e.g., expended energy or missed feeding opportunity). It then outlines the conditions that may lead to long-term consequences for the individual if the animal cannot fully recover from the short-term costs and how these in turn may affect the population. Within each biological resource section (e.g., marine mammals, birds, and fishes) the detailed methods to predict effects on specific taxa are derived from this conceptual framework.

An animal is considered "exposed" to a sound if the received sound level at the animal's location is above the background ambient noise level within a similar frequency band. A variety of effects may result from exposure to acoustic and explosive activities.

The categories of potential effects are shown in the box below:

- ***Injury and other non-auditory injury*** *- Injury to organs or tissues of an animal.*
- ***Hearing loss*** *- A noise-induced decrease in hearing sensitivity which can be either temporary or permanent and may be limited to a narrow frequency range of hearing.*
- ***Masking*** *- When the perception of a biologically important sound (i.e., signal) is interfered with by a second sound (i.e., noise).*
- ***Physiological stress*** *- An adaptive process that helps an animal cope with changing conditions; although, too much stress can result in physiological problems.*
- ***Behavioral response*** *- A reaction ranging from very minor and brief changes in attentional focus, changes in biologically important behaviors, and avoidance of a sound source or area, to aggression or prolonged flight.*

Figure F-1 is a flowchart that diagrams the process used to evaluate the potential effects to marine animals exposed to sound-producing activities. The shape and color of each box on the flowchart represent either a decision point in the analysis (green diamonds); specific processes such as responses, costs, or recovery (blue rectangles); external factors to consider (purple parallelograms); and final outcomes for the individual or population (orange ovals and rectangles). Each box is labeled for reference throughout the following sections. For simplicity, sound is used here to include not only sound waves but also blast waves generated from explosive sources. Box A1, the Sound-Producing Activity, is the source of this stimuli and therefore the starting point in the analysis.

The first step in predicting whether an activity is capable of affecting a marine animal is to define the stimuli experienced by the animal. The stimuli include the overall level of activity, the surrounding acoustical environment, and characteristics of the sound when it reaches the animal.

Sounds emitted from a sound-producing activity (Box A1) travel through the environment to create a spatially variable sound field. The received sound by the animal (Box A2) determines the range of possible effects. The received sound can be evaluated in several ways, including number of times the sound is experienced (repetitive exposures), total received energy, or highest sound pressure level (SPL) experienced.

Sounds that are higher than the ambient noise level and within an animal's hearing sensitivity range (Box A3) have the potential to cause effects. There can be any number of individual sound sources in a given activity, each with its own unique characteristics. For example, a Navy training exercise may involve several ships and aircraft using several types of sonar. Environmental factors such as temperature and bottom type impact how sound spreads and attenuates through the environment. Additionally, independent of the sounds, the overall level of activity and the number and movement of sound sources are important to help predict the probable reactions.

The magnitude of the responses is based on the characteristics of the acoustic stimuli and the characteristics of the animal (species, susceptibility, life history stage, size, and past experiences). Very high exposure levels close to explosives have the potential to cause injury. High-level, long-duration, or repetitive exposures may potentially cause some hearing loss. All perceived sounds may lead to behavioral responses, physiological stress, and masking. Many sounds, including sounds that are not detectable by the animal, could have no effect (Box A4).

#### F.1.1.1 Injury

Injury (Box B1) refers to the direct injury of tissues and organs by shock or pressure waves impinging upon or traveling through an animal's body. Marine animals are well adapted to large, but relatively slow, hydrostatic pressure changes that occur with changing depth. However, injury may result from exposure to rapid pressure changes, such that the tissues do not have time to adequately adjust.

Therefore, injury is normally limited to relatively close ranges from explosions. Injury can be mild and fully recoverable or, in some cases, lead to mortality.

Injury includes both auditory and non-auditory injury. Auditory injury is the direct mechanical injury to hearing-related structures, including tympanic membrane rupture, disarticulation of the middle ear ossicles, and injury to the inner ear structures such as the organ of Corti and the associated hair cells.

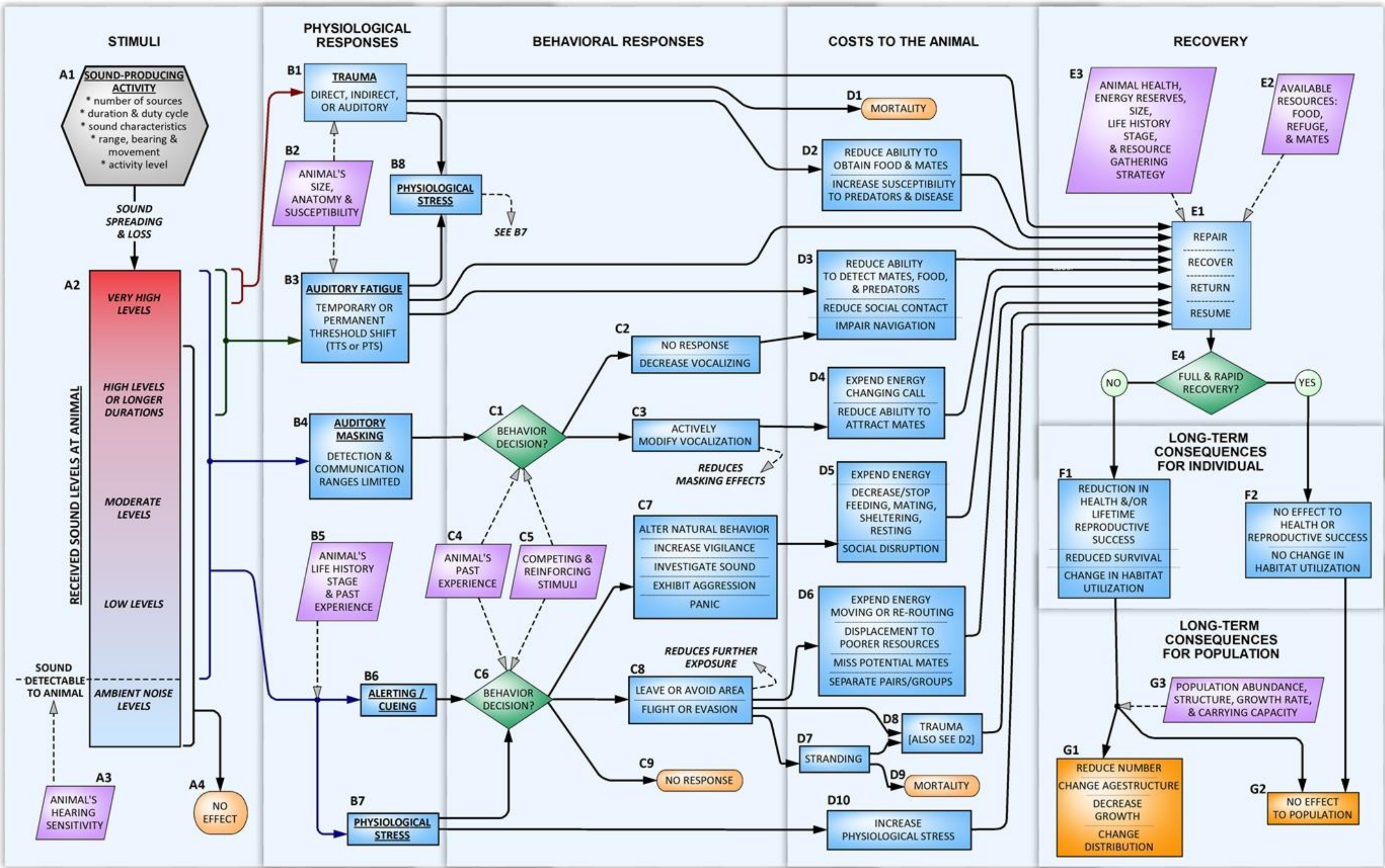



**Figure F-1: Flow Chart of the Evaluation Process of Sound-Producing Activities**

Auditory injury differs from auditory fatigue in that the latter involves the overstimulation of the auditory system at levels below those capable of causing direct mechanical damage. Auditory injury is always injurious but can be temporary. One of the most common consequences of auditory injury is hearing loss.

Non-auditory injury can include hemorrhaging of small blood vessels and the rupture of gas-containing tissues such as the lung, swim bladder, or gastrointestinal tract. After the ear (or other sound-sensing organs), these are usually the organs and tissues most sensitive to explosive injury. An animal's size and anatomy are important in determining its susceptibility to non-auditory injury (Box B2). Larger size indicates more tissue to protect vital organs. Therefore, larger animals should be less susceptible to injury than smaller animals. In some cases, acoustic resonance of a structure may enhance the vibrations resulting from noise exposure and result in an increased susceptibility to injury. The size, geometry, and material composition of a structure determine the frequency at which the object will resonate. Because most biological tissues are heavily damped, the increase in susceptibility from resonance is limited.

Vascular and tissue bubble formation resulting from sound exposure is a hypothesized mechanism of injury to breath-holding marine animals. Bubble formation and growth due to direct sound exposure have been hypothesized (Crum et al., 2005; Crum & Mao, 1996); however, the experimental laboratory conditions under which these phenomena were observed would not be replicated in the wild. Certain dive behaviors by breath-holding animals are predicted to result in conditions of blood nitrogen super-saturation, potentially putting an animal at risk for decompression sickness (Fahlman et al., 2014), although this phenomena has not been observed (Houser et al., 2009). In addition, animals that spend long periods of time at great depths are predicted to have super-saturated tissues that may slowly release nitrogen if the animal then spends a long time at the surface (i.e., stranding) (Houser et al., 2009).

Injury could increase the animal's physiological stress (Box B8), which feeds into the stress response (Box B7) and also increases the likelihood or severity of a behavioral response. Injury may reduce an animal's ability to secure food by reducing its mobility or the efficiency of its sensory systems, making the injured individual less attractive to potential mates, increasing an individual's chances of contracting diseases, falling prey to a predator (Box D2), or increasing an animal's overall physiological stress level (Box D10). Severe injury can lead to the death of the individual (Box D1).

Damaged tissues from mild to moderate injury may heal over time. The predicted recovery of direct injury is based on the severity of the injury, availability of resources, and characteristics of the animal. The animal may also need to recover from any potential costs due to a decrease in resource gathering efficiency and any secondary effects from predators or disease. Severe injuries can lead to reduced survivorship (longevity), elevated stress levels, and prolonged alterations in behavior that can reduce an animal's lifetime reproductive success. An animal with decreased energy stores or a lingering injury may be less successful at mating for one or more breeding seasons, thereby decreasing the number of offspring produced over its lifetime.

#### F.1.1.2 Hearing Loss

Hearing loss, also called a noise-induced threshold shift, is possibly the most studied type of effect from sound exposures to animals. Hearing loss manifests itself as loss in hearing sensitivity across part of an animal's hearing range, which is dependent upon the specifics of the noise exposure. Hearing loss may be either permanent threshold shift (PTS), or temporary threshold shift (TTS). If the threshold shift eventually returns to zero (the animal's hearing returns to pre-exposure value), the threshold shift is a TTS. If the threshold shift does not return to zero but leaves some finite amount of threshold shift, then that remaining threshold shift is a PTS. Figure F-2 shows one hypothetical threshold shift that completely recovers, a TTS, and one that does not completely recover, leaving some PTS.

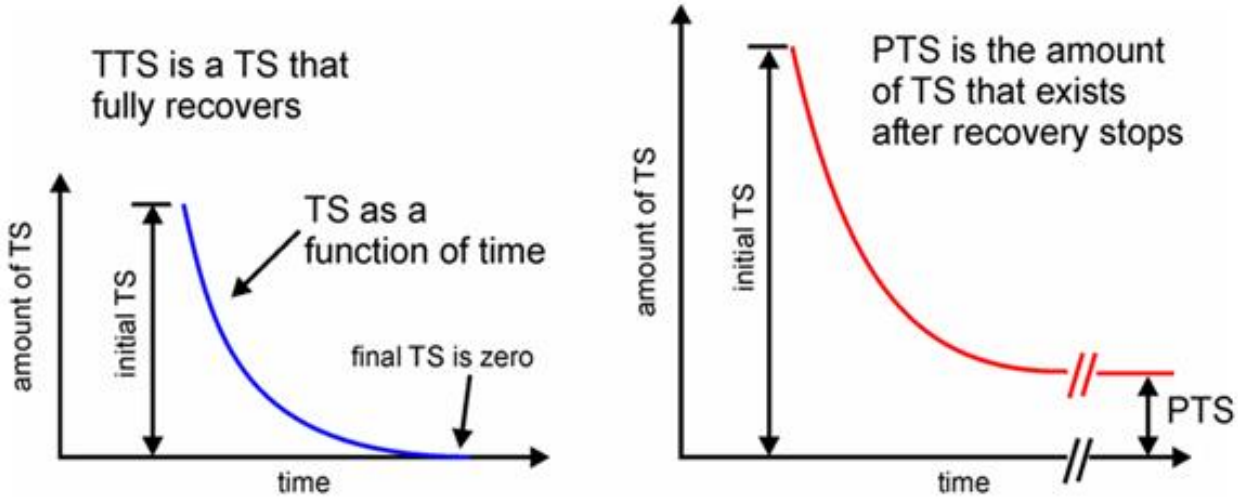



**Figure F-2: Two Hypothetical Threshold Shifts**

The characteristics of the received sound stimuli are used and compared to the animal's hearing sensitivity and susceptibility to noise (Box A3) to determine the potential for hearing loss. The amplitude, frequency, duration, and temporal pattern of the sound exposure are important parameters for predicting the potential for hearing loss over a specific portion of an animal's hearing range. Duration is particularly important because hearing loss increases with prolonged exposure time. Longer exposures with lower sound levels can cause more threshold shift than a shorter exposure using the same amount of energy overall. The frequency of the sound also plays an important role. Experiments show that animals are most susceptible to hearing loss (Box B3) within their most sensitive hearing range. Sounds outside of an animal's audible frequency range do not cause hearing loss.

The mechanisms responsible for hearing loss may consist of a variety of mechanical and biochemical processes in the inner ear, including physical damage or distortion of the tympanic membrane (not including tympanic membrane rupture which is considered auditory injury), physical damage or distortion of the cochlear hair cells, hair cell death, changes in cochlear blood flow, and swelling of cochlear nerve terminals (Henderson et al., 2006; Kujawa & Liberman, 2009). Although the outer hair cells are the most prominent target for fatigue effects, severe noise exposures may also result in inner hair cell death and loss of auditory nerve fibers (Henderson et al., 2006).

The relationship between TTS and PTS is complicated and poorly understood, even in humans and terrestrial mammals, where numerous studies failed to delineate a clear relationship between the two. Relatively small amounts of TTS (e.g., less than 40–50 decibels [dB] measured two minutes after exposure) will recover with no apparent permanent effects; however, terrestrial mammal studies revealed that larger amounts of threshold shift can result in permanent neural degeneration, despite the hearing thresholds returning to normal (Kujawa & Liberman, 2009). The amounts of threshold shift induced by Kujawa and Liberman (2009) were described as being "at the limits of reversibility." It is unknown whether smaller amounts of threshold shift can result in similar neural degeneration, or if effects would translate to other species such as marine animals.

Hearing loss can increase an animal's physiological stress (Box B8), which feeds into the stress response (Box B7). Hearing loss increases the likelihood or severity of a behavioral response and increase an animal's overall physiological stress level (Box D10). Hearing loss reduces the distance over which animals can communicate and detect other biologically important sounds (Box D3). Hearing loss could also be inconsequential for an animal if the frequency range affected is not critical for that animal to hear within, or the hearing loss is of such short duration (e.g., a few minutes) that there are no costs to the individual.

Small to moderate amounts of hearing loss may recover over a period of minutes to days, depending on the amount of initial threshold shift. Severe noise-induced hearing loss may not fully recover, resulting in some amount of PTS. An animal whose hearing does not recover quickly and fully could suffer a reduction in lifetime reproductive success. An animal with PTS may be less successful at mating for one or more breeding seasons, thereby decreasing the number of offspring it can produce over its lifetime.

#### F.1.1.3 Masking

Masking occurs if the noise from an activity interferes with an animal's ability to detect, understand, or recognize biologically relevant sounds of interest (Box B4). In this context noise refers to unwanted or unimportant sounds that mask an animal's ability to hear sounds of interest. Sounds of interest include those from conspecifics such as offspring, mates, and competitors; echolocation clicks; sounds from predators; natural, abiotic sounds that may aid in navigation; and reverberation, which can give an animal information about its location and orientation within the ocean. The probability of masking increases as the noise and sound of interest increase in similarity and the masking noise increases in level. The frequency, received level, and duty cycle of the noise determines the potential degree of auditory masking. Masking only occurs during the sound exposure.

A behavior decision (either conscious or instinctive) is made by the animal when the animal detects increased background noise, or possibly, when the animal recognizes that biologically relevant sounds are being masked (Box C1). An animal's past experiences can be important in determining the behavioral response when dealing with masking (Box C4). For example, an animal may modify its vocalizations to reduce the effects of masking noise. Other stimuli present in the environment can influence an animal's behavior decision (Box C5) such as the presence of predators, prey, or potential mates.

An animal may exhibit a passive behavioral response when coping with masking (Box C2). It may simply not respond and keep conducting its current natural behavior. An animal may also stop calling until the background noise decreases. These passive responses do not present a direct energetic cost to the animal; however, masking will continue, depending on the acoustic stimuli.

An animal may actively compensate for masking (Box C3). An animal can vocalize more loudly to make its signal heard over the masking noise. An animal may also shift the frequency of its vocalizations away from the frequency of the masking noise. This shift can actually reduce the masking effect for the animal and other animals that are listening in the area.

If masking impairs an animal's ability to hear biologically important sounds (Box D3) it could reduce an animal's ability to communicate with conspecifics or reduce opportunities to detect or attract more distant mates, gain information about their physical environment, or navigate. An animal that modifies its vocalization in response to masking could also incur a cost (Box D4). Modifying vocalizations may cost the animal energy, interfere with the behavioral function of a call, or reduce a signaler's apparent quality as a mating partner. For example, songbirds that shift their calls up an octave to compensate for increased background noise attract fewer or less-desirable mates, and many terrestrial species advertise body size and quality with low-frequency vocalizations (Slabbekoorn & Ripmeester, 2007). Masking may also lead to no measurable costs for an animal. Masking could be of short duration or intermittent such that biologically important sounds that are continuous or repeated are received by the animal between masking noise.

Masking only occurs when the sound source is operating; therefore, direct masking effects stop immediately upon cessation of the sound-producing activity. Masking could have long-term consequences for individuals if the activity was continuous or occurred frequently enough.

#### F.1.1.4 Physiological Stress

Marine animals naturally experience physiological stress as part of their normal life histories. The physiological response to a stressor, often termed the stress response, is an adaptive process that helps an animal cope with changing external and internal environmental conditions. Sound-producing activities have the potential to cause additional stress. However, too much of a stress response can be harmful to an animal, resulting in physiological dysfunction.

If a sound is detected (i.e., heard or sensed) by an animal, a stress response can occur (Box B7). The severity of the stress response depends on the received sound level by the animal (Box A2), the details of the sound-producing activity (Box A1), and the animal's life history stage (e.g., juvenile or adult, breeding or feeding season), and past experience with the stimuli (Box B5). An animal's life history stage is an important factor to consider when predicting whether a stress response is likely (Box B5). An animal's life history stage includes its level of physical maturity (i.e., larva, infant, juvenile, sexually mature adult) and the primary activity in which it is engaged such as mating, feeding, or rearing/caring for young. Prior experience with a stressor may be of particular importance because repeated experience with a stressor may dull the stress response via acclimation (St. Aubin & Dierauf, 2001) or increase the response via sensitization. Additionally, if an animal suffers injury or hearing loss, a physiological stress response will occur (Box B8).

The generalized stress response is characterized by a release of hormones (Reeder & Kramer, 2005) and other chemicals (e.g., stress markers) such as reactive oxidative compounds associated with noise-induced hearing loss (Henderson et al., 2006). Stress hormones include norepinephrine and epinephrine (i.e., the catecholamines), which produce elevations in the heart and respiration rate, increase awareness, and increase the availability of glucose and lipid for energy. Other stress hormones are the glucocorticoid steroid hormones cortisol and aldosterone, which are classically used as an indicator of a stress response and to characterize the magnitude of the stress response (Hennessy et al., 1979).

An acute stress response is traditionally considered part of the startle response and is hormonally characterized by the release of the catecholamines. Annoyance type reactions may be characterized by the release of either or both catecholamines and glucocorticoid hormones. Regardless of the physiological changes that make up the stress response, the stress response may contribute to an animal's decision to alter its behavior.

Elevated stress levels may occur whether or not an animal exhibits a behavioral response (Box D10). Even while undergoing a stress response, competing stimuli (e.g., food or mating opportunities) may overcome any behavioral response. Regardless of whether the animal displays a behavioral response, this tolerated stress could incur a cost to the animal. Reactive oxygen compounds produced during normal physiological processes are generally counterbalanced by enzymes and antioxidants; however, excess stress can lead to damage of lipids, proteins, and nucleic acids at the cellular level (Berlett & Stadtman, 1997; Sies, 1997; Touyz, 2004).

Frequent physiological stress responses may accumulate over time increasing an animal's chronic stress level. Each component of the stress response is variable in time, and stress hormones return to baseline levels at different rates. Elevated chronic stress levels are usually a result of a prolonged or repeated disturbance. Chronic elevations in the stress levels (e.g., cortisol levels) may produce long-term health consequences that can reduce lifetime reproductive success.

#### F.1.1.5 Behavioral Reactions

Behavioral responses fall into two major categories: alterations in natural behavior patterns and avoidance. These types of reactions are not mutually exclusive, and many overall reactions may be combinations of behaviors or a sequence of behaviors. Severity of behavioral reactions can vary drastically between minor and brief reorientations of the animal to investigate the sound, to severe reactions such as aggression or prolonged flight. The type and severity of the behavioral response will determine the cost to the animal. The total number of vessels and platforms involved, the size of the activity area, the distance between the animal and activity, and the duration of the activity are important considerations when predicting the initial behavioral responses.

A physiological stress response (Box B7) such as an annoyance or startle reaction, or cueing or alerting (Box B6) may cause an animal to make a behavior decision (Box C6). Any exposure that produces an injury or hearing loss is also assumed to produce a stress response (Box B7) and increase the severity or likelihood of a behavioral reaction. Both an animal's experience (Box C4) and competing and reinforcing stimuli (Box C5) can affect an animal's behavior decision. The decision can result in three general types of behavioral reactions: no response (Box C9), area avoidance (Box C8), or alteration of a natural behavior (Box C7).

An animal’s past experiences can be important in determining what behavior decision it may make when dealing with a stress response (Box C4). Habituation is the process by which an animal learns to ignore or tolerate stimuli over some period and return to a normal behavior pattern, perhaps after being exposed to the stimuli with no negative consequences. Sensitization is when an animal becomes more sensitive to a set of stimuli over time, perhaps as a result of a past, negative experience that could result in a stronger behavioral response.

Other stimuli (Box C5) present in the environment can influence an animal’s behavioral response. These stimuli may be conspecifics or predators in the area or the drive to engage in a natural behavior. Other stimuli can also reinforce the behavioral response caused by acoustic stimuli. For example, the awareness of a predator in the area coupled with the sound-producing activity may elicit a stronger reaction than the activity alone would have.

An animal may reorient, become more vigilant, or investigate if it detects a sound-producing activity (Box C7). These behaviors all require the animal to divert attention and resources, therefore slowing or stopping their presumably beneficial natural behavior. This can be a very brief diversion, or an animal may not resume its natural behaviors until after the activity has concluded. An animal may choose to leave or avoid an area where a sound-producing activity is taking place (Box C8). A more severe form of this comes in the form of flight or evasion. Avoidance of an area can help the animal avoid further effects by avoiding or reducing further exposure. An animal may also choose not to respond to a sound-producing activity (Box C9).

An animal that alters its natural behavior in response to stress or an auditory cue may slow or cease its natural behavior and instead expend energy reacting to the sound-producing activity (Box D5). Natural behaviors include feeding, breeding, sheltering, and migrating. The cost of feeding disruptions depends on the energetic requirements of individuals and the potential amount of food missed during the disruption. Alteration in breeding behavior can result in delaying reproduction. The costs of a brief interruption to migrating or sheltering are less clear.

An animal that avoids a sound-producing activity may expend additional energy moving around the area, be displaced to poorer resources, miss potential mates, or have social interactions affected (Box D6). The amount of energy expended depends on the severity of the behavioral response. Missing potential mates can result in delaying reproduction. Groups could be separated during a severe behavioral response such as flight and offspring that depend on their parents may die if they are permanently separated. Splitting up an animal group can result in a reduced group size, which can have secondary effects on individual foraging success and susceptibility to predators.

Some severe behavioral reactions can lead to stranding (Box D7) or secondary injury (Box D8). Animals that take prolonged flight, a severe avoidance reaction, may injure themselves or strand in an environment for which they are not adapted. Some injury is likely to occur to an animal that strands (Box D8). Injury can reduce the animal's ability to secure food and mates, and increase the animal's susceptibility to predation and disease (Box D2). An animal that strands and does not return to a hospitable environment may die (Box D9).

#### F.1.1.6 Long-Term Consequences

The potential long-term consequences from behavioral responses are difficult to discern. Animals displaced from their normal habitat due to an avoidance reaction may return over time and resume their natural behaviors. This is likely to depend upon the severity of the reaction and how often the activity is repeated in the area. In areas of repeated and frequent acoustic disturbance, some animals may habituate to the new baseline; conversely, species that are more sensitive may not return, or return but not resume use of the habitat in the same manner. For example, an animal may return to an area to feed but no longer rest in that area. Long-term abandonment or a change in the utilization of an area by enough individuals can change the distribution of the population. Frequent disruptions to natural behavior patterns may not allow an animal to recover between exposures, which increase the probability of causing long-term consequences to individuals.

The magnitude and type of effect and the speed and completeness of recovery (i.e., return to baseline conditions) must be considered in predicting long-term consequences to the individual animal (Box E4). The predicted recovery of the animal (Box E1) is based on the cost to the animal from any reactions, behavioral or physiological. Available resources fluctuate by season, location, and year and can play a major role in an animal's rate of recovery (Box E2). Recovery can occur more quickly if plentiful food resources, many potential mates, or refuge or shelter is available. An animal's health, energy reserves, size, life history stage, and resource gathering strategy affect its speed and completeness of recovery (Box E3). Animals that are in good health and have abundant energy reserves before an effect takes place will likely recover more quickly.

Animals that recover quickly and completely are unlikely to suffer reductions in their health or reproductive success, or experience changes in habitat utilization (Box F2). No population-level effects would be expected if individual animals do not suffer reductions in their lifetime reproductive success or change their habitat utilization (Box G2). Animals that do not recover quickly and fully could suffer reductions in their health and lifetime reproductive success; they could be permanently displaced or change how they use the environment; or they could die (Box F1). These long-term consequences to the individual can lead to consequences for the population (Box G1); although, population dynamics and abundance play a role in determining how many individuals would need to suffer long-term consequences before there was an effect on the population.

Long-term consequences to individuals can translate into consequences for populations dependent upon population abundance, structure, growth rate, and carry capacity. Carrying capacity describes the theoretical maximum number of animals of a particular species that the environment can support. When a population nears its carrying capacity, its growth is naturally limited by available resources and predator pressure. If one, or a few animals, in a population are removed or gather fewer resources, then other animals in the population can take advantage of the freed resources and potentially increase their health and lifetime reproductive success. Abundant populations that are near their carrying capacity (theoretical maximum abundance) that suffer consequences on a few individuals may not be affected overall. Populations that exist well below their carrying capacity may suffer greater consequences from any lasting consequences to even a few individuals. Population-level consequences can include a change in the population dynamics, a decrease in the growth rate, or a change in geographic distribution.

## F.1.2 Conceptual Framework for Assessing Effects from Energy-Producing Activities

### F.1.2.1 Stimuli

#### F.1.2.1.1 Magnitude of the Energy Stressor

Regulations do not provide threshold criteria to determine the significance of the potential effects from activities that involve the use of varying electromagnetic frequencies or lasers. Many organisms, primarily marine vertebrates, have been studied to determine their thresholds for detecting electromagnetic fields, as reviewed by Normandeau et al. (2011); however, there are no data on predictable responses to exposure above or below detection thresholds. The types of electromagnetic fields discussed are those from mine neutralization activities (magnetic influence minesweeping). High-energy and low-energy lasers were considered for analysis. Low-energy lasers (e.g., targeting systems, detection systems, laser light detection and ranging) do not pose a risk to organisms (U.S. Department of the Navy, 2010b) and, therefore, will not be discussed further. Radar was also considered for analysis and was determined not to pose a risk to biological resources.

#### F.1.2.1.2 Location of the Energy Stressor

Evaluation of potential energy exposure risks considered the spatial overlap of the resource occurrence and electromagnetic field and high-energy laser use. Wherever appropriate, specific geographic areas of potential effect were identified and the relative location of the resource with respect to the source was considered. For example, the greatest potential electromagnetic energy exposure is at the source, where intensity is greatest and the greatest potential for high energy laser exposure is at the ocean's surface, where high-energy laser intensity is greatest. All light energy, including laser light, entering the ocean becomes absorbed and scattered at a rate that is dependent on the frequency of the light. For most laser applications, the energy is rapidly reduced as the light penetrates the ocean.

#### F.1.2.1.3 Behavior of the Organism

Evaluation of potential energy exposure risk considered the behavior of the organism, especially where the organism lives and feeds (e.g., surface, water column, seafloor). The analysis for electromagnetic devices considered those species with the ability to perceive or detect electromagnetic signals. The analysis for high-energy lasers and radar particularly considered those species known to occur at or above the surface of the ocean.

#### F.1.2.2 Immediate Response and Costs to the Individual

Many different types of organisms (e.g., some invertebrates, fishes, turtles, birds, mammals) are sensitive to electromagnetic fields (Normandeau et al., 2011). An organism that encounters a disturbance in an electromagnetic field could respond by moving toward the source, moving away from it, or not responding at all. The types of electromagnetic devices used in the Proposed Action simulate the electromagnetic signature of a vessel passing through the water column, so the expected response would be similar to that of vessel movement. However, since there would be no actual strike potential, a physiological response would be unlikely in most cases. Recovery of an individual from encountering electromagnetic fields would be variable, but since the physiological response would likely be minimal, as reviewed by Normandeau et al. (2011), any recovery time would also be minimal.

Very little data are available to analyze potential effects on organisms from exposure to high energy lasers. For all but the highest-energy lasers, the greatest laser-related concern for marine species is damage to an organism's ability to see.

#### F.1.2.3 Long-Term Consequences to the Individual and Population

Long-term consequences are considered in terms of a resource's existing population level, growth and mortality rates, other stressors on the resource from the Proposed Action, cumulative effects on the resource, and the ability of the population to recover from or adapt to effects. Effects of multiple or repeated stressors on individuals are cumulative.

### F.1.3 Conceptual Framework for Assessing Effects from Physical Disturbance or Strike

#### F.1.3.1 Stimuli

##### F.1.3.1.1 Size and Weight of the Objects

To determine the likelihood of a strike and the potential effects on an organism or habitat that would result from a physical strike, the size and weight of the striking object relative to the organism or habitat must be considered. For example, most small organisms and early life stages would simply be displaced by the movement generated by a large object moving through, or falling into, the water, whereas a larger organism could potentially be struck by an object since it may not be displaced by the movement of the water. The weight of the object is also a factor that would determine the severity of a strike. A strike by a heavy object would be more severe than a strike by a low-weight object (e.g., a decelerator/parachute, flare end cap, or chaff canister).

##### F.1.3.1.2 Location and Speed of the Objects

Evaluation of potential physical disturbance or strike risk considered the spatial overlap of the resource occurrence and potential striking objects. Analysis of effects from physical disturbance or strike stressors focuses on proposed activities that may cause an organism or habitat to be struck by an object moving through the air (e.g., aircraft), water (e.g., vessels, in-water devices, towed devices), or dropped into the water (e.g., non-explosive practice munitions and seafloor devices). The area of operation, vertical distribution, and density of these items also play central roles in the likelihood of effect. Wherever appropriate, specific geographic areas of potential effect are identified. Analysis of potential physical disturbance or strike risk also considered the speed of vessels as a measure of intensity. Some vessels move slowly, while others are capable of high speeds.

#### F.1.3.1.3 Buoyancy of the Objects

Evaluation of potential physical disturbance or strike risk in the ocean considered the buoyancy of targets or expended materials during operation, which will determine whether the object will be encountered at the surface, within the water column, or on the seafloor.

#### F.1.3.1.4 Behavior of the Organism

Evaluation of potential physical disturbance or strike risk considered where organisms occur and if they occur in the same geographic area and vertical distribution as those objects that pose strike risks.

### F.1.3.2 Immediate Response and Costs to the Individual

Before being struck, some organisms would sense a pressure wave through the water and respond by remaining in place, moving away from the object, or moving toward it. An organism displaced a small distance by movements from an object falling into the water nearby would likely continue on with no response. However, others could be disturbed and may exhibit a generalized stress response. If the object actually hit the organism, direct injury in addition to stress may result. The function of the stress response in vertebrates is to rapidly raise the blood sugar level to prepare the organism to flee or fight. This generally adaptive physiological response can become a liability if the stressor persists and the organism cannot return to its baseline physiological state.

Most organisms would respond to sudden physical approach or contact by darting quickly away from the stimulus. Other species may respond by freezing in place or seeking refuge. In any case, the individual must stop whatever it was doing and divert its physiological and cognitive attention to responding to the stressor. The energy costs of reacting to a stressor depend on the specific situation, but in all cases the caloric requirements of stress reactions reduce the amount of energy available to the individual for other functions such as predator avoidance, reproduction, growth, and metabolism.

The ability of an organism to return to what it was doing following a physical strike (or near miss resulting in a stress response) is a function of fitness, genetic, and environmental factors. Some organisms are more tolerant of environmental or human-caused stressors than others and become acclimated more easily. Within a species, the rate at which an individual recovers from a physical disturbance or strike may be influenced by its age, sex, reproductive state, and general condition. An organism that has reacted to a sudden disturbance by swimming at burst speed would tire after some time; its blood hormone and sugar levels may not return to normal for 24 hours. During the recovery period, the organism may not be able to attain burst speeds and could be more vulnerable to predators. If the individual were not able to regain a steady state following exposure to a physical stressor, it may suffer depressed immune function and even death.

### F.1.3.3 Long-Term Consequences to the Population

Long-term consequences are considered in terms of a resource’s existing population level, growth and mortality rates, other stressors on the resource from the Proposed Action, cumulative effects on the resource, and the ability of the population to recover from or adapt to effects. Effects of multiple or repeated stressors on individuals are cumulative.

### F.1.4 Conceptual Framework for Assessing Effects from Entanglement

#### F.1.4.1 Stimuli

##### F.1.4.1.1 Physical Properties of the Objects

For an organism to become entangled in military expended materials, the materials must have certain properties, such as the ability to form loops and a high breaking strength. Some items could have a relatively low breaking strength on their own, but that breaking strength could be increased if multiple loops were wrapped around an entangled organism.

##### F.1.4.1.2 Physical Features of the Resource

The physical makeup of the organism itself is also considered when evaluating the risk of entanglement. Some species, by their size or physical features, are more susceptible to entanglement than others. For example, more rigid bodies with protruding snouts (e.g., hammerhead shark) or large, rigid fins (e.g., humpback whale) would have an increased risk of entanglement when compared to species with smoother, streamlined bodies such as lamprey or eels.

##### F.1.4.1.3 Location of the Objects

Evaluation of potential entanglement risk considered the spatial overlap of the resource occurrence and military expended materials. Distribution and density of expended items play a central role in the likelihood of effect. Wherever appropriate, specific geographic areas of potential effect are identified.

##### F.1.4.1.4 Behavior of the Organism

Evaluation of potential entanglement risk considered the general behavior of the organism, including where the organism typically occurs (e.g., surface, water column, seafloor). The analysis particularly considered those species known to become entangled in nonmilitary expended materials (e.g., “marine debris”) such as fishing lines, nets, rope, and other derelict fishing gear that often entangle marine organisms.

#### F.1.4.2 Immediate Response and Costs to the Individual

The potential effects of entanglement on a given organism depend on the species and size of the organism. Species that have protruding snouts, fins, or appendages are more likely to become entangled than smooth-bodied organisms. Also, items could get entangled by an organism's mouth, if caught on teeth or baleen, with the rest of the item trailing alongside the organism. Materials similar to fishing gear, which is designed to entangle an organism, would be expected to have a greater entanglement potential than other materials. An entangled organism would likely try to free itself of the entangling object and in the process may become even more entangled, possibly leading to a stress response. The net result of being entangled by an object could be disruption of the normal behavior, injury due to lacerations, and other sublethal or lethal effects.

#### F.1.4.3 Long-Term Consequence to the Individual and Population

Consequences of entanglement could range from an organism successfully freeing itself from the object or remaining entangled indefinitely, possibly resulting in lacerations and other sublethal or lethal effects. Stress responses or infection from lacerations could lead to latent mortality. The analysis will focus on reasonably foreseeable long-term consequences of the direct effect, particularly those that could affect the fitness of an individual. Changes in an individual’s growth, survival, annual reproductive success, or lifetime reproductive success could have population-level effects if enough individuals are affected. This population-level effect would vary among species and taxonomic groups.

## F.1.5 Conceptual Framework for Assessing Effects from Ingestion

### F.1.5.1 Stimuli

#### F.1.5.1.1 Size of the Objects

To assess the ingestion risk from military expended materials, this analysis considered the size of the object relative to the animal's ability to swallow it. Some items are too large to be ingested (e.g., non-explosive practice bombs and most targets) and effects from these items are not discussed further. However, these items may potentially break down into smaller ingestible pieces over time. Items that are of ingestible size when they are introduced into the environment are carried forward for analysis within each resource section where applicable.

#### F.1.5.1.2 Location of the Objects

Evaluation of potential ingestion risk considered the spatial overlap of the resource occurrence and military expended materials. The distribution and density of expended items play a central role in the likelihood of effect. Wherever appropriate, specific geographic areas of potential effect were identified.

#### F.1.5.1.3 Buoyancy of the Objects

Evaluation of potential ingestion risk considered the buoyancy of military expended materials to determine whether the object will be encountered within the water column (including the surface) or on the seafloor. Less buoyant materials, such as solid metal materials (e.g., projectiles or munitions fragments), sink rapidly to the seafloor. More buoyant materials include less dense items (e.g., target fragments and decelerators/parachutes) that may be caught in currents and gyres or entangled in floating kelp. These materials can remain in the water column for an indefinite period of time before sinking. However, decelerators/parachutes are weighted and would generally sink, unless that sinking is suspended, in the scenario described here.

#### F.1.5.1.4 Feeding Behavior

Evaluation of potential ingestion risk considered the feeding behavior of the organism, including where (e.g., surface, water column, seafloor) and how (e.g., filter feeding) the organism feeds and what it feeds on. The analysis particularly considered those species known to ingest nonfood items (e.g., plastic or metal items).

### F.1.5.2 Immediate Response and Costs to the Individual

Potential effects of ingesting foreign objects on a given organism depend on the species and size of the organism. Species that normally eat spiny hard-bodied invertebrates would be expected to have tougher mouths and guts than those that normally feed on softer prey. Materials similar in size and shape to the normal diet of an organism may be more likely to be ingested without causing harm to the animal; however, some general assumptions were made. Relatively small objects with smooth edges, such as shells or small-caliber projectiles, might pass through the digestive tract without causing harm. A small sharp-edged item may cause the individual immediate physical distress by tearing or cutting the mouth, throat, or stomach. If the object is rigid and large (relative to the individual's mouth and throat), it may block the throat or obstruct digestive processes. An object may even be enclosed by a cyst in the gut lining. The net result of ingesting large foreign objects is disruption of the normal feeding behavior, which could be sublethal or lethal.

#### F.1.5.3 Long-Term Consequences to the Individual and Population

The consequences of ingesting nonfood items could be nutrient deficiency, bioaccumulation, uptake of toxic chemicals, compaction, and mortality. The analysis focused on reasonably foreseeable long-term consequences of the direct effect, particularly those that could affect the fitness of an individual. Changes in an individual's growth, survival, annual reproductive success, or lifetime reproductive success could have population-level effects if enough individuals were affected. This population-level effect would vary among species and taxonomic groups.

### F.1.6 Conceptual Framework for Assessing Effects from Secondary Stressors

This conceptual framework describes the potential effects to marine species exposed to stressors indirectly through effects on habitat and prey availability (e.g., sediment or water quality, and physical disturbance). Stressors from Navy training and testing activities could pose indirect effects on marine biological resources via indirect effects to habitat or to prey. These include indirect effects from (1) explosives, explosion byproducts, and unexploded munitions; (2) metals; (3) chemicals; and (4) transmission of disease and parasites. The methods used to determine secondary stressors on marine resources are presented below. Once a category of primary stressor has been analyzed to determine how a marine biological resource is affected, an analysis follows of how a secondary stressor is potentially affecting a marine resource. After the secondary stressors are identified, a determination on the significance of the secondary effect is made. The same criteria to determine the level of significance for primary effects are used for secondary stressors. In addition, it is possible for a significant primary effect to produce a beneficial indirect effect. For example, sinking exercises could generate a significant effect to the seafloor and surrounding habitats, while causing a potential beneficial secondary effect by creating hard-bottom habitat for invertebrates, producing a food source for fishes, and creating structural refuges for other biological resources.

#### F.1.6.1 Secondary Stressors

##### F.1.6.1.1 Effects on Habitat

Primary effects defined in each marine resource section were used to develop a conceptual model to predict the potential secondary stressors on each habitat or resource. This conceptual model incorporated factors such as the co-occurrence of stressors in space and time, the effects or assessment endpoints of individual stressors (e.g., habitat alteration, changes in animal behavior or physiology, injury, mortality, or changes in human use), and the duration and intensity of the effects of individual stressors. For example, a secondary stressor from a munitions strike could be habitat degradation. The primary effect or stressor is the actual strike on the habitat such as the seafloor, with the introduction of military expended materials, munitions, and fragments inducing further habitat degradation.

Secondary stressors can also induce additive effects on habitats. These types of effects are also determined by summing the individual stressors with identical and quantifiable assessment endpoints. For example, if one stressor disturbed 0.25 square nautical miles ($NM^2$) of benthic habitat, a second stressor disturbed 0.5 $NM^2$, and all other stressors did not disturb benthic habitat, then the total benthic habitat disturbed would be 0.75 $NM^2$. For stressors with identical but not quantifiable assessment endpoints, potential additive effects were qualitatively evaluated using available scientific knowledge and best professional judgment. Other habitat effects such as underwater detonations were assessed by size of charge (net explosive weight), charge radius, height above the seafloor, substrate types in the area, and equations linking all these factors. The analysis also considered that effects of underwater explosions vary with the bottom substrate type and that the secondary effects would also be variable among substrate types.

##### F.1.6.1.2 Effects on Prey Availability

Assessing the effects of secondary stressors on prey availability falls into two main areas over different temporal scales: the cost to an individual over a relatively short amount of time (short-term) and the cost to an individual or population over a longer period of time (long-term).

#### F.1.6.2 Immediate Response and Costs to the Individual

After a primary effect was identified, an analysis of secondary stressors on that resource was initiated. This analysis examined whether indirect effects would occur after the initial (primary) effect and at what temporal scale that secondary stressor would affect the resource (short-term or long-term). An assessment was then made as to whether the secondary stressor would affect an individual or a population. For example, an underwater explosion could affect a single resource such as a fish or multiple other species in the food web (e.g., prey species such as plankton). The analysis also took into consideration whether the primary effect affected more than an individual or single species. For example, a prey species that would be directly injured or killed by an explosive blast could draw in predators or scavengers from the surrounding waters that would feed on those organisms, and in turn could be more directly susceptible to being injured or killed by subsequent explosions. For purposes of this analysis, indirect effects on a resource did not require trophic transfer (e.g., bioaccumulation) in order to be observed. It is important to note that the terms “indirect” and “secondary” describe how the effect may occur in an organism or its ecosystem and does not imply reduced severity of environmental consequences.

#### F.1.6.3 Long-Term Consequences to the Individual and Population

Long-term consequences of secondary stressors on an individual or population are often difficult to determine. Once a primary effect is identified, the severity of that effect helps to determine the temporal scale at which the secondary stressor can be measured. For most marine resources, the abundance of prey species near a detonation point would be diminished for a short period (weeks to months) before being repopulated by animals from adjacent waters. In some extreme cases, recovery of the habitat or prey resources could occur over a relatively long-time frame (months to years). It is important to note that indirect effects often differ among resources, spatial, and temporal scales.

## F.2 Sediments and Water Quality

### F.2.1 Explosives and Explosive Byproducts

Explosives may be introduced into the seawater and sediments by the Proposed Action. The explosive fillers contained within the munitions used during training and testing activities and their degradation products can enter the environment through high-order detonations (i.e., the munition functions as intended and the vast majority of explosives are consumed), low-order detonations (i.e., the munition partially functions with only a portion of the explosives consumed), or unexploded munitions (i.e., the munition fails to detonate and explosives remain in the casing). In the case of a successful detonation, only a small or residual amount of explosives may enter the marine environment (U.S. Environmental Protection Agency, 2012). A low-order detonation would result in some residual explosives and some unconsumed explosives remaining in the munitions casing entering the water. In the case of unexploded munitions, the explosives contained in the munition would not be consumed and would remain encased within the munition as it enters the marine environment. The munitions casing may corrode or rupture over time and release explosives into the sediments and water column.

The behavior of explosives and explosives byproducts in marine environments and the extent to which those constituents of explosives have adverse effects are influenced by a number of processes, including the ease with which the explosive dissolves in a liquid such as water (solubility), the degree to which explosives are attracted to other materials in the water (e.g., clay-sized particles and organic matter, sorption), and the tendency of the explosives to evaporate (volatilization). These characteristics, in turn, influence the extent to which the material is subject to biotic (biological) and abiotic (physical and chemical) transformation and degradation (Pennington & Brannon, 2002). The solubility of various explosives is provided in Table F-1. In the table, higher values indicate greater solubility. For example, high melting explosive is virtually insoluble in water. Table salt, which dissolves easily in water, is included in the table for comparison.

**Table F-1: Water Solubility of Common Explosives and Explosive Degradation Products**

| Compound | Water Solubility[1] (mg/L at 20 ˚C) |
|---|---|
| Table salt (sodium chloride)[2] | 357,000 |
| Ammonium perchlorate (O) | 249,000 |
| Picric acid (E) | 12,820 |
| Nitrobenzene (D) | 1,900 |
| Dinitrobenzene (E) | 500 |
| Trinitrobenzene (E) | 335 |
| Dinitrotoluene (D) | 160 |
| TNT (E) | 130 |
| Tetryl (E) | 51 |
| Pentaerythritoltetranitrate (E) | 43 |
| Royal Demolition Explosive (E) | 38 |
| High Melting Explosive (E) | 7 |

Source: (U.S. Department of the Navy, 2008a)
[1] Units are milligrams per liter at 20 degrees Celsius
[2]Table salt is not an explosive degradation product
Notes: D = explosive degradation product, E = explosive, O = oxidizer additive, TNT = trinitrotoluene

According to Walker et al. (2006), trinitrotoluene (TNT), royal demolition explosive, and high melting explosive experience rapid biological and photochemical degradation in marine systems. The authors noted that productivity in marine and estuarine systems is largely controlled by the limited availability of nitrogen. Because nitrogen is a key component of explosives, they are attractive as substrates for marine bacteria that metabolize other naturally occurring organic matter, such as polycyclic aromatic hydrocarbons. Juhasz and Naidu (2007) also noted that microbes use explosives as sources of carbon and energy.

Carr and Nipper (2003) indicated that conversion of TNT to carbon dioxide, methane, and nitrates in coastal sediments (a process referred to as "mineralization") occurred at rates that were typical for naturally occurring compounds such as phenanthrene, fluoranthene, toluene, and naphthalene. They noted that transformation of 2, 6-dinitrotoluene and picric acid by organisms in sediments is dependent on temperature and type of sediment (e.g., finer-grained). Pavlostathis and Jackson (2002) reported that the marine microalgae *Anabaena* spp. was highly efficient at the removal and metabolism of TNT in a continuous flow experiment. Nipper et al. (2002) noted that irreversible binding to sediments and biodegradation of 2, 6-dinitrotoluene, tetryl, and picric acid occurred in fine-grained sediments high in organic carbon resulting in lower concentrations of the contaminants. Cruz-Uribe et al. (2007) noted that three species of marine macroalgae metabolize TNT to 2-amino-4,6-dinitrotoluene and 4-amino-2, 6-dinitrotoluene, and speculate that “the ability of marine macroalgae to metabolize TNT is widespread, if not generic.” The studies cited above indicate that TNT and its constituent products can be removed from the environment by naturally occurring biological processes in sediments, reducing sediment toxicity from these chemical contaminants.

Singh et al. (2009) indicated that biodegradation of royal demolition explosive and high melting explosive occurs with oxygen (aerobic) and without oxygen (anoxic or anaerobic), but that they were more easily degraded under anaerobic conditions. Crocker et al. (2006) indicated that the mechanisms of high melting explosive and royal demolition explosive biodegradation are similar, but that high melting explosive degrades more slowly. Singh et al. (2009) noted that royal demolition explosive and high melting explosive are biodegraded under a variety of anaerobic conditions by specific microbial species and by mixtures of such species. Zhao et al. (2004a); Zhao et al. (2004b) found that biodegradation of royal demolition explosive and high melting explosive occurs in cold marine sediments.

According to Singh et al. (2009), typical end products of the degradation of royal demolition explosive include nitrite, nitrous oxide, nitrogen, ammonia, formaldehyde, formic acid, and carbon dioxide. Crocker et al. (2006) stated that many of the primary and secondary intermediate compounds from biodegradation of royal demolition explosive and high melting explosive are unstable in water and spontaneously decompose. Thus, these explosives are degraded by a combination of biotic and abiotic reactions. Formaldehyde is subsequently metabolized to formic acid, methanol, carbon dioxide, or methane by various microorganisms (Crocker et al., 2006).

A series of research efforts focused on World War II underwater munitions disposal sites in Hawaii (Briggs et al., 2016; Kelley et al., 2016; Koide et al., 2016) and an intensively used live fire range in the Mariana Islands (Smith & Marx, 2016) provide information in regard to the effects of undetonated materials and unexploded munitions on marine life.

On a localized scale, research at World War II munitions ocean disposal sites in Hawaii investigated nearby sediments, seawater, or marine life to determine if released constituents from the munitions (including explosive materials and metals) could be detected. Comparisons were made between disposal site samples and “clean” reference sites. Analysis of the samples showed no confirmed detection for explosive materials.

Investigations by Kelley et al. (2016) and Koide et al. (2016) found that intact munitions (i.e., ones that failed to detonate or non-explosive practice munitions) residing in or on soft sediments habitats provided hard substrate similar to other disposed objects or “artificial reefs” that attracted “hard substrate species,” which would not have otherwise colonized the area. Sampling these species revealed that there was no bioaccumulation of munitions-related chemicals in the species (Koide et al., 2016).

On a broader scale, the island of Farallon De Medinilla (in the Mariana Islands) has been used as a target area for both explosive and non-explosive munitions since 1971. Between 1997 and 2012, the Navy conducted 14 underwater scientific surveys around the island, providing a consistent, long-term investigation of a single site where munitions have been used regularly (Smith & Marx, 2016). Marine life assessed during these surveys included algae, corals, benthic invertebrates, sharks, rays, bony fishes, and sea turtles. The investigators found no evidence over the 16-year period that the condition of the physical or biological resources had been adversely affected to a significant degree by the training activities (Smith & Marx, 2016). Furthermore, they found that the health, abundance, and biomass of fishes, corals and other marine resources were comparable to or superior to those in similar habitats at other locations within the Mariana Archipelago.

These findings are consistent with other assessments such as that done for the Potomac River Test Range at Dahlgren, Virginia which was established in 1918 and is the nation's largest fully instrumented, over-the-water gun-firing range. Munitions tested at Dahlgren have included rounds from small caliber guns up to the Navy's largest (16 inch guns), bombs, rockets, mortars, grenades, mines, depth charges, and torpedoes (U.S. Department of the Navy, 2013a). Results from the assessment indicate that munitions expended at Dahlgren have not contributed significant concentrations of explosive materials or explosives byproducts to the Potomac River water and sediments given those contributions are orders of magnitude less than concentrations already present in the Potomac River from natural and manmade sources (U.S. Department of the Navy, 2013f).

In summary, multiple investigations since 2007 involving survey and sampling of World War II munition dump sites off Oahu Hawaii and other locations, have found the following: (1) chemicals and degradation products from underwater munitions "do not pose a risk to human health or to fauna living in direct contact with munitions"; (2) metals measured in sediment samples next to World War II munitions are lower than naturally occurring marine levels and "do not cause a significant effect on the environment"; and (3) sediment is not a significant sink of chemicals released by degradation of the explosive components in munitions (Edwards et al., 2016).

The concentration of explosive munitions and any associated explosives byproducts at any single location in the Hawaii-California Training and Testing (HCTT) Study Area would be a small fraction of the totals that have accumulated over decades at World War II era dump sites and military ranges. Based on findings from much more intensively used locations, effects on sediments from the use of explosive munitions during training and testing activities would be negligible by comparison. As a result, explosives by-products and unexploded munitions would have no meaningful effect on sediments.

Most explosive material is consumed in an explosion, so the vast majority of explosive material entering the marine environment would be encased in munitions that failed to detonate. Failure rates for all of the munitions used in the Proposed Action are not available; however, based on the data that are available, a 5 percent munitions failure rate was applied to estimate failure rates for all munitions used in the Proposed Action. Based on the available data, low-order detonation rates are assumed to be at least an order of magnitude less than failure rates for all munitions and are not considered in the analysis. Table F-2 provides information about the rates of failure and low-order detonations for explosives and other munitions (MacDonald & Mendez, 2005).

### Table F-2: Failure and Low-Order Detonation Rates of Military Munitions

| Munitions | Failure Rate (Percent) | Low-Order Detonation Rate (Percent) |
|---|---|---|
| Guns/artillery | 4.68 | 0.16 |
| Hand grenades | 1.78 | n/a |
| Explosive munitions | 3.37 | 0.09 |
| Rockets | 3.84 | n/a |
| Submunitions | 8.23 | n/a |

Note: n/a = not available

Most activities involving explosives and explosives byproducts would be conducted more than 3 nautical miles (NM) offshore in each range complex. Activities in these areas (3–200 NM) would be subject to federal sediment and water quality standards and guidelines.

Explosives are also used in nearshore areas (low tide line to 3 NM) specifically designated for mine countermeasure and mine neutralization activities. These activities would be subject to state sediment and water quality standards and guidelines.

For explosives byproducts, “local” refers to the water column in the vicinity of the underwater detonation. For unconsumed explosives, “local” refers to the area of potential effect from explosives in a zone of sediment about 6 feet (ft.) in diameter around the unconsumed explosive where it comes to rest on the seafloor.

### F.2.2 Chemicals Other Than Explosives

Under the Proposed Action, chemicals other than explosives are associated with the following military expended materials: (1) solid-fuel propellants in missiles and rockets; (2) Otto Fuel II torpedo propellant and combustion byproducts; (3) polychlorinated biphenyls (PCBs) in target vessels used during sinking exercises; (4) other chemicals associated with munitions; and (5) chemicals that simulate chemical warfare agents, referred to as “chemical simulants.”

Hazardous air pollutants from explosives and explosives byproducts are discussed in Section 3.1 (Air Quality). Explosives and explosives byproducts are discussed in Section 3.2.3.1 (Explosives and Explosives Byproducts). Fuels onboard manned aircraft and vessels are not reviewed, nor are fuel-loading activities, onboard operations, or maintenance activities reviewed, because normal operation and maintenance of Navy equipment is not part of the Proposed Action.

The largest chemical constituent of missiles is solid propellant. Solid propellant contains both the fuel and the oxidizer, a source of oxygen needed for combustion. An extended-range Standard Missile-2 typically contains 1,822 pounds (lb.) of solid propellant. Ammonium perchlorate is the oxidizing agent used in most modern solid-propellant formulas (Chaturvedi & Dave, 2015). It normally accounts for 50 to 85 percent of the propellant by weight. Ammonium dinitramide may also be used as an oxidizing agent. Aluminum powder as a fuel additive ranges from 5 to 22 percent by weight of solid propellant; it is added to increase missile range and payload capacity. The explosives high melting explosive (octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine) and royal demolition explosive (hexahydro-1,3,5-trinitro-1,3,5-triazine) may be added, although they usually comprise less than 30 percent of the propellant by weight. Many of the constituents used in propellants are also commonly used for commercial purposes but require additional processing to achieve certain properties necessary for rocket and missile propulsion. (Missile Technology Control Regime, 1996).

The U.S. Environmental Protection Agency (USEPA) issued a paper characterizing the munitions constituents accumulated at over 30 military sites around the United States and Canada where explosives and propellants have been used (U.S. Environmental Protection Agency, 2012). The sites assessed in the paper were all land-based ranges; however, the results are useful for analyzing similar activities conducted at sea. The paper noted that perchlorate was generally not detected at anti-tank ranges and that perchlorate is so soluble in water and mobile in soil that surface accumulation apparently does not occur. The paper includes a case study that estimates the amount of residual perchlorate deposited from a rocket fired at a test track. The rocket propellant contained 68 lb. of ammonium perchlorate. Samples were collected both behind the firing point and along the test track before and after the rocked was fired. No differences in perchlorate concentrations in soils were detected at any location before or after the firing, and all measurements recorded perchlorate concentrations of less than 1 microgram per kilogram. That case study concluded that 99.997 percent of perchlorate is consumed by the rocket motor (U.S. Environmental Protection Agency, 2012). Fitzpatrick et al. (2006) found similar results from an air-launched AIM-7 missile, a missile used by the Navy and similar to missiles used in the Proposed Action. These studies, and others cited in each paper, demonstrate that the motors used in rockets and missiles are highly efficient at burning propellant fuels, leaving only trace amounts often at undetectable levels in the environment.

Several torpedoes (e.g., MK-54) use Otto Fuel II as a liquid propellant. Otto Fuel II is composed of primarily three synthetic substances: Propylene glycol dinitrate and nitro-diphenylamine (76 percent), dibutyl sebacate (22 percent), and 2-nitrodiphenylamine as a stabilizer (2 percent). Propylene glycol dinitrate, which is a liquid, is the explosive component of Otto Fuel II. Dibutyl sebacate, also known as sebacic acid, is also a liquid. It is used commercially to make plastics, many of which are used for packaging food, and to enhance flavor in foods such as ice cream, candy, baked goods, and nonalcoholic drinks. The third component, 2-nitrodiphenylamine, is a solid substance used to control the combustion of the propylene glycol dinitrate (Espinoza & Wehrtmann, 2008). Combustion byproducts of Otto Fuel II include nitrous oxides, carbon monoxide, carbon dioxide, hydrogen, nitrogen, methane, ammonia, and hydrogen cyanide. During normal venting of excess pressure or upon failure of the torpedo's buoyancy bag, the following constituents are discharged: carbon dioxide, water, hydrogen, nitrogen, carbon monoxide, methane, ammonia, hydrochloric acid, hydrogen cyanide, formaldehyde, potassium chloride, ferrous oxide, potassium hydroxide, and potassium carbonate (Waters et al., 2013).

Target vessels are only used during sinking exercises, which occur infrequently. Polychlorinated biphenyls are a concern because they are present in certain solid materials (e.g., insulation, wires, felts, and rubber gaskets) on vessels used as targets for sinking exercises. These vessels are selected from a list of Navy-approved vessels that have been cleaned in accordance with USEPA guidelines (U.S. Environmental Protection Agency, 2014a). By rule, a sinking exercise must be conducted at least 50 NM offshore and in water at least 6,000 ft. deep (40 Code of Federal Regulations 229.2).

The USEPA estimates that as much as 100 lb. of PCBs remain onboard sunken target vessels. The USEPA considers the contaminant levels released during the sinking of a target to be within the standards of the Marine Protection, Research, and Sanctuaries Act (16 United States Code 1341, et seq.) (U.S. Environmental Protection Agency, 2014a). Under the 2014 agreement with the USEPA, the Navy will not likely use aircraft carriers or submarines as the targets for a sinking exercise (U.S. Environmental Protection Agency, 2014a). Based on these considerations, PCBs will not be considered further.

Table F-3 lists the chemical constituents produced in the combustion of propellants and fuels, as described above, and list constituents remaining after the detonations of non-munitions, such as spotting charges and tracers. Not all of the listed chemical constituents in propellant and Otto Fuel II would be used in combination; some are substitutes that would replace another chemical in the list, depending on the type of propellant used. For example, ammonium perchlorate is the preferred oxidizer in propellant, but ammonium dinitramide could act as the oxidizer in some propellants. These constituents are in addition to the explosives contained in munitions, which were discussed in Section 3.2.3.1 (Explosives and Explosives Byproducts).

### Table F-3: Constituents in Munitions Other Than Explosives

| Munitions Component | Constituent |
|---|---|
| Pyrotechnics<br>Tracers<br>Spotting Charges | Barium chromate<br>Potassium perchlorate<br>Chlorides<br>Phosphorus<br>Titanium compounds |
| Oxidizers | Lead (II) oxide |
| Propellant (rockets and missiles) | Ammonium perchlorate (50 to 85 percent by weight)<br>Ammonium dinitramide<br>Aluminum powder (5 to 21 percent by weight)<br>High melting explosive<br>Royal demolition explosive<br>Hydroxyl-terminated polybutadiene<br>Carboxyl-terminated polybutadiene<br>Polybutadiene-acrylic acid-acrylonitrile<br>Triphenyl bismuth<br>Nitrate esters<br>Nitrated plasticizers<br>Polybutadiene-acrylic acid polymer<br>Elastomeric polyesters<br>Polyethers<br>Nitrocellulose plasticized with nitroglycerine<br>2-nitrodiphenylamine<br>N-methyl-4-nitroaniline<br>Hydrazine |
| Otto Fuel II (torpedoes) | Propylene glycol dinitrate and Nitro-diphenylamine (76 percent by weight)<br>dibutyl sebacate (22 percent by weight)<br>2-nitrodiphenylamine (2 percent by weight)<br>Combustion products (nitrous oxides, carbon monoxide, carbon dioxide, hydrogen, nitrogen, methane, ammonia, hydrogen cyanide)<br>Venting or buoyancy bag failure (hydrochloric acid, hydrogen cyanide, formaldehyde, potassium chloride, ferrous oxide, potassium hydroxide, and potassium carbonate) |
| Chemical Simulants | Navy Chemical Agent Simulant 82<br>glacial acetic acid<br>triethyl phosphate<br>sulfur hexafluoride<br>1,1,1,2 tetrafluoroethane<br>1,1-difluoroethane |
| Delay Elements | Barium chromate<br>Potassium perchlorate |

| Munitions Component | Constituent |
|---|---|
| | Lead chromate |
| Fuses | Potassium perchlorate |
| Detonators | Fulminate of mercury<br>Potassium perchlorate |
| Primers | Lead azide |

The environmental fate of Otto Fuel II and its components is largely unknown. Neither the fuel mixture nor its three main components are particularly volatile or soluble in water; however, when mixed with water propylene glycol dinitrate forms a volatile mixture, making evaporation an important fate process (Espinoza & Wehrtmann, 2008). The compound 2-Nitrodiphenylamine may precipitate from water or be taken up by particulates. Dibutyl sebacate is rapidly biodegraded. Neither propylene glycol dinitrate nor 2-nitrodiphenylamine are readily biodegradable, but both of these chemicals break down when exposed to ultraviolet light (Powell et al., 1998).

Lead azide, titanium compounds, perchlorates, barium chromate, and fulminate of mercury are not natural constituents of seawater. Lead oxide is a rare, naturally occurring mineral. It is one of several lead compounds that form films on lead objects in the marine environment (Agency for Toxic Substances and Disease Registry, 2007). Metals are discussed in more detail in Section 3.2.3.2 (Metals).

Because chemical and biological warfare agents remain a security threat, the Department of Defense uses relatively harmless compounds (chemical simulants) as substitutes for chemical and biological warfare agents to test equipment intended to detect their presence. Chemical and biological agent detectors monitor for the presence of chemical and biological warfare agents and protect military personnel and civilians from the threat of exposure to these agents. The simulants trigger a response by sensors in the detection equipment without irritating or injuring personnel involved in testing detectors.

Navy Chemical Agent Simulant 82 (commonly referred to as NCAS-82), glacial acetic acid, triethyl phosphate, sulfur hexafluoride, 1,1,1,2 tetrafluoroethane (a refrigerant commonly known as R134), and 1,1-difluoroethane (a refrigerant commonly known as R-152a) are also referred to as gaseous simulants and can be released in smaller quantities in conjunction with glacial acetic acid or triethyl phosphate releases. The types of biological simulants that may be used include spore-forming bacteria, non-spore-forming bacteria, ovalbumin, bacteriophage MS2, and *Aspergillus niger*. The simulants are generally dispersed by hand at the detector or by aircraft as a fine mist or aerosol. The exposure of military personnel or the public to even small amounts of real warfare agents, such as nerve or blistering agents, or harmful biological organisms, such as anthrax, is potentially harmful and is illegal in most countries, including the United States. Furthermore, their use, including for the testing of detection equipment, is banned by international agreement.

Simulants must have one or more characteristic of a real chemical or biological agents—size, density, or aerosol behavior—to effectively mimic the agent. Simulants must also pose a minimal risk to human health and the environment to be used safely in outdoor tests. Simulants are selected using the following criteria: (1) safety to humans and the environment, and (2) the ability to trigger a response by sensors used in the detection equipment. Simulants must be relatively benign (e.g., low toxicity or effects potential) from a human health, safety, and environmental perspective. Exposure levels during testing activities should be well below concentrations associated with any adverse human health or environmental effects. The degradation products of simulants must also be harmless. Given these criteria for choosing simulants for use in testing activities, it is reasonable to conclude that simulants would have no effect on sediments and water quality in the Study Area. Simulants are not analyzed further in this section.

### F.2.3 Metals

Anthropogenic sources of metals include the processing of industrial ores (e.g., iron ore), production of chemicals, fertilizers used in agriculture, the marine industry (e.g., anti-fouling anti-corrosion paints), runoff from urban and suburban sprawl, dredge spoil disposal, exhaust from automotive transportation, atmospheric deposition, and industrial emissions (Haugland et al., 2006). Metals would be introduced into nearshore and offshore marine waters and sediments by the Proposed Action. Because of the physical and chemical reactions that occur with metals in marine systems, many metals will precipitate out of seawater and settle in solid form on the seafloor where they can concentrate in sediments. Thus, metal contaminants in sediments pose a greater environmental concern than metals in the water column.

Military expended materials such as steel bomb bodies or fins, missile casings, small arms projectiles, and naval gun projectiles may contain small percentages (less than 1 percent by weight) of lead, manganese, phosphorus, sulfur, copper, nickel, tungsten, chromium, molybdenum, vanadium, boron, selenium, columbium, or titanium. Small-caliber projectiles are composed of steel with small amounts of aluminum and copper and brass casings that are 70 percent copper and 30 percent zinc. Medium- and large-caliber projectiles are composed of steel, brass, copper, tungsten, and other metals. The 20 mm cannon shells used in close-in weapons systems are composed mostly of tungsten alloy. Some projectiles have lead cores (U.S. Department of the Navy, 2008b). Torpedo guidance wire is composed of copper and cadmium coated with plastic (U.S. Department of the Navy, 2008a). Sonobuoy components include batteries and battery electrodes, lead solder, copper wire, and lead used for ballast. Thermal batteries in sonobuoys are contained in an airtight, sealed and welded stainless steel case that is 0.03-0.1 inch (in.) thick and resistant to the battery electrolytes (U.S. Department of the Navy, 2008b). Rockets are usually composed of steel and steel alloys, although composite cases made of glass, carbon, or Kevlar fiber are also used (Missile Technology Control Regime, 1996). Anchors used to moor mine shapes or other seafloor devices are often recovered but in some cases may be left on the seafloor to facilitate recovery of the device (see Section 3.0, Introduction). Metal anchors and other types of anchors (e.g., concrete blocks) with metal components are composed primarily of steel.

Non-explosive practice munitions consist of ammunition and components that contain no explosive material, and may include (1) ammunition and components that have had all explosive material removed and replaced with non-explosive material, (2) empty ammunition or components, and (3) ammunition or components that were manufactured with non-explosive material in place of all explosive material. These practice munitions vary in size from 25 to 500 lb. and are designed to simulate the characteristics of explosive munitions for training and testing activities. Some non-explosive practice munitions may also contain unburned propellant (e.g., rockets), and some may contain spotting charges or signal cartridges for locating the point of effect (e.g., smoke charges for daylight spotting or flash charges for night spotting) (U.S. Department of the Navy, 2010d). Large, non-explosive bombs—also called “practice” or “bomb dummy units”—are composed mainly of iron and steel casings filled with sand, concrete, or vermiculite. These materials are similar to those used to construct artificial reefs. Non-explosive bombs are configured to have the same weight, size, center of gravity, and ballistics as explosive bombs (U.S. Department of the Navy, 2006b). Practice bombs do not contain the explosive materials.

Decommissioned vessels used as targets for sinking exercises are selected from a list of U.S. Navy-approved vessels that have been cleaned or remediated in accordance with USEPA guidelines. By rule, vessel-sinking exercises must be conducted at least 50 NM offshore and in water at least 6,000 ft. deep (40 Code of Federal Regulations part 229.2). The USEPA requires the contaminant levels released during the sinking of a target to be within the standards of the Marine Protection, Research, and Sanctuaries Act (16 United States Code 1341, et seq.).

In general, three things happen to materials that come to rest on the ocean floor: (1) they lodge in sediments where there is little or no oxygen below 4 in., (2) they remain on the ocean floor and begin to react with seawater, or (3) they remain on the ocean floor and become encrusted by marine organisms. As a result, rates of deterioration depend on the metal or metal alloy and the conditions in the immediate marine and benthic environment. If buried deep in ocean sediments, materials tend to decompose at much lower rates than when exposed to seawater (Ankley, 1996). With the exception of torpedo guidance wires and sonobuoy components, sediment burial appears to be the fate of most munitions used in marine warfare (Environmental Sciences Group, 2005b).

When metals are exposed to seawater, they begin to slowly corrode, a process that creates a layer of corroded material between the seawater and uncorroded metal. This layer of corrosion removes the metal from direct exposure to the corrosiveness of seawater, a process that further slows movement of the metals into the adjacent sediments and water column. This is particularly true of aluminum. Elevated levels of metals in sediments would be restricted to a small zone around the metal, and any release to the overlying water column would be diluted. In a similar fashion, as materials become covered by marine life, both the direct exposure of the material to seawater and the rate of corrosion decrease. Dispersal of these materials in the water column is controlled by physical mixing and diffusion, both of which tend to vary with time and location. The analysis of metals in marine systems begins with a review of studies involving metals used in military training and testing activities that may be introduced into the marine environment.

In one study, the water was sampled for lead, manganese, nickel, vanadium, and zinc at a shallow bombing range in Pamlico Sound (estuarine waters of North Carolina) immediately following a training event with non-explosive practice bombs. All water quality parameters tested, except nickel, were within the state limits. The nickel concentration was significantly higher than the state criterion, although the concentration did not differ significantly from the control site located outside the bombing range. The results suggest that bombing activities were not responsible for the elevated nickel concentrations (U.S. Department of the Navy, 2010d). A recent study conducted by the U.S. Marine Corps sampled sediments and water quality for 26 different constituents, including lead and magnesium, related to munitions at several U.S. Marine Corps water-based training ranges. These areas also were used for bombing practice. No munitions constituents were detected above screening values used at the U.S. Marine Corps water ranges (U.S. Department of the Navy, 2010d).

A study by Pait et al. (2010) of previous Navy training areas at Vieques, Puerto Rico, found generally low concentrations of metals in marine sediments. Areas in which live ammunition and loaded weapons were used ("live-fire areas") were included in the analysis. These results are relevant because the concentrations of expended munitions at Vieques are significantly greater than would be found anywhere in the HCTT Study Area. Table F-4 compares the sediment concentrations of several metals from those naval training areas with sediment screening levels established by the National Oceanic and Atmospheric Administration (Buchman, 2008).

As shown in Table F-4, average sediment concentrations of the metals evaluated, except for copper, were below both the threshold and probable effects levels (metrics similar to the effects range levels). The average copper concentration was above the threshold effect level, but below the probable effect level. For other elements: (1) the mean sediment concentration of arsenic at Vieques was 4.37 micrograms per gram (µg/g), and the highest concentration was 15.4 µg/g. Both values were below the sediment quality guidelines examined; and (2) the mean sediment concentration of manganese in sediment was 301 µg/g, and the highest concentration was 967 µg/g (Pait et al., 2010). The National Oceanic and Atmospheric Administration did not report threshold or probable effects levels for manganese.

**Table F-4: Concentrations of and Screening Levels for Selected Metals in Marine Sediments, Vieques, Puerto Rico**

| Metal | Sediment Concentration (µg/g) | | | Sediment Guidelines – National Oceanic and Atmospheric Administration (µg/g) | |
|---|---|---|---|---|---|
| | Minimum | Maximum | Average | Threshold Effects Level* | Probable Effects Level* |
| Cadmium | 0 | 1.92 | 0.15 | 0.68 | 4.21 |
| Chromium | 0 | 178 | 22.5 | 52.3 | 160 |
| Copper | 0 | 103 | 25.9 | 18.7 | 390 |
| Lead | 0 | 17.6 | 5.42 | 30.24 | 112 |
| Mercury | N/R | 0.112 | 0.019 | 130 | 700 |
| Nickel | N/R | 38.3 | 7.80 | 15.9 | 42.8 |
| Zinc | N/R | 130 | 34.4 | 124 | 271 |

Notes: N/R = not reported, µg/g = micrograms per gram

*Threshold Effects Level and Probable Effects Level are metrics similar to the effects range metrics (i.e., Effects Range Low and Effects Range Median) used to assess potential effects of contaminants on sediments. The Threshold Effects Levels is the average of the 50th percentile and the 15th percentile of a dataset and the Probable Effects Level is the average of the 50th percentile and the 85th percentile of a dataset.

The effects of lead and lithium were studied at the Canadian Forces Maritime Experimental and Test Ranges near Nanoose Bay, British Columbia, Canada (Environmental Sciences Group, 2005b). These materials are common to expendable mobile anti-submarine warfare training targets, acoustic device countermeasures, sonobuoys, and torpedoes. The study noted that lead is a naturally occurring metal in the environment and that typical concentrations of lead in seawater in the test range were between 0.01 and 0.06 parts per million (ppm), while concentration of lead in sediments was between 4 and 16 ppm. Cores of marine sediments in the test range show a steady increase in lead concentration from the bottom of the core to a depth of approximately 8 in. (20.3 centimeters [cm]). This depth corresponds to the late 1970s and early 1980s, and the lead contamination was attributed to atmospheric deposition of lead from gasoline additives. The sediment cores showed a general reduction in lead concentration to the present time, coincident with the phasing out of lead in gasoline by the mid-1980s. The study also noted that other training ranges have shown minimal effects of lead ballasts because they are usually buried deep in marine sediments where they are not biologically available. The study concluded that the lead ballasts would not adversely affect marine organisms because of the low probability of mobilization of lead.

A study by the Navy examined the effects of materials from activated seawater batteries in sonobuoys that freely dissolve in the water column (e.g., lead, silver, and copper ions), as well as nickel-plated steel housing, lead solder, copper wire, and lead shot used for sonobuoy ballast (U.S. Department of the Navy, 1993). The study concluded that constituents released by saltwater batteries as well as the decomposition of other sonobuoy components did not exceed state or federal standards, and that the reaction products are short-lived in seawater.

A series of research efforts focused on World War II underwater munitions disposal sites in Hawaii (Briggs et al., 2016; Kelley et al., 2016; Koide et al., 2016; University of Hawaii, 2010) and an intensively used live fire range in the Mariana Islands (Smith & Marx, 2016) provide information in regard to the effects of undetonated materials and unexploded munitions on marine life.

On a localized scale, research at World War II munitions ocean disposal sites in Hawaii investigated nearby sediments, seawater, or marine life to determine if metals could be detected. For metals, although there were localized elevated levels of arsenic and lead in several biota samples and in the sediment adjacent to the munitions, the origin of those metals could not be definitively linked to the munitions since comparison of sediment between the clean reference site and the disposal site showed relatively little difference. This was especially the case for a comparison with samples for ocean disposed dredge spoils sites (locations where material taken from the dredging of harbors on Oahu was disposed). At individual sampling sites adjacent to munitions, the concentrations of metals were not significantly higher as compared to the background at control sites and not significant in comparison to typical deep-sea marine sediments (Briggs et al., 2016). Observations and data collected also did not indicate any adverse effect to the localized ecology due to the presence of munitions degrading for over 75 years when compared to control sites. When specifically looking at marine organisms around the munitions (Kelley et al., 2016; Koide et al., 2016), the analysis indicated that in soft bottom habitats the expended items were providing hard substrate similar to other disposed objects or “artificial reefs” that attracted “hard substrate species” that would not have otherwise colonized the area and that there was no bioaccumulation of munitions-related chemicals for the species sampled (Koide et al., 2016).

On a broader scale, the island of Farallon de Medinilla (in the Mariana Islands) has been used as a target area since 1971. Between 1997 and 2012, there were 14 underwater scientific survey investigations around the island providing a long-term look at potential effects on the marine life from training and testing involving the use of munitions (Smith & Marx, 2016). Munitions use has included explosive rounds from gunfire, high explosive bombs by Navy aircraft and U.S. Air Force B-52s, in addition to the expenditure of inert rounds and non-explosive practice bombs. Marine life assessed during these surveys included algae, corals, benthic invertebrates, sharks, rays, bony fishes, and sea turtles. The investigators found no evidence over the 16-year period, that the condition of the biological resources had been adversely affected to a significant degree by the training activities (Smith & Marx, 2016). Furthermore, they found that the health, abundance, and biomass of fishes, corals, and other marine resources were comparable to or superior to those in similar habitats at other locations within the Mariana Archipelago.

These findings are consistent with other assessments such as those performed for the Potomac River Test Range at Dahlgren, Virginia which was established in 1918 and is the nation’s largest fully instrumented, over-the-water gun-firing range. Munitions tested at Dahlgren have included rounds from small-caliber guns up to the Navy’s largest (16 in. guns), along with bombs, rockets, mortars, grenades, mines, depth charges, and torpedoes (U.S. Department of the Navy, 2013f). Results from the assessment indicate that munitions expended at Dahlgren have not contributed significant concentrations of metals to the Potomac River and that the concentrations of metals in local sediments are orders of magnitude lower than in other areas of the Potomac River where metals are introduced from natural and other manmade sources. (U.S. Department of the Navy, 2013f).

The concentrations of metals from munitions, expended materials, and devices in any one location in the HCTT Study Area would be a small fraction of that from a World War II era munitions disposal site, or a target island used for 45 years, or a water range in a river used for almost 100 years. Based on findings from much more intensively used locations, the water quality effects from the use of munitions, expended materials, and other devices resulting from any of the proposed training and testing activities would be negligible by comparison.

### F.2.4 Other Materials

Under the Proposed Action, other materials include marine markers and flares, chaff, towed and stationary targets, and miscellaneous components of other expended objects (e.g., concrete blocks used as anchors) (see Appendix I, Military Expended Materials and Direct Strike Impact Analyses, for details). These materials and components are either made mainly of non-reactive or slowly reactive materials (e.g., glass, carbon fibers, and plastics) or break down or decompose into benign byproducts (e.g., rubber, steel, iron, and concrete). Most of these objects would settle to the seafloor where they would (1) be exposed to seawater, (2) become lodged in or covered by seafloor sediments, (3) become encrusted by oxidation products such as rust, (4) dissolve slowly, or (5) be covered by marine organisms such as coral. Plastics may float or descend to the bottom, depending upon their buoyancy. Marine markers and flares are largely consumed during use.

Towed and stationary targets include floating steel drums; towed aerial targets; the trimaran; and inflatable, floating targets. The trimaran is a three-hulled boat with a 4 ft. square sail that is towed as a moving target. Large, inflatable, plastic targets can be towed or left stationary. Towed aerial targets are either (1) rectangular pieces of nylon fabric 7.5 ft. by 40 ft. that reflect radar or lasers; or (2) aluminum cylinders with a fiberglass nose cone, aluminum corner reflectors (fins), and a short plastic tail section. This second target is about 10 ft. long and weighs about 75 lb. These four targets are recovered after use, and will not be considered further.

Marine markers are pyrotechnic devices that are dropped on the water's surface during training exercises to mark a position, to support search and rescue activities, or as a bomb target. The MK 58 marker is a tin tube that weighs about 12 lb. Markers release smoke at the water surface for 40 to 60 minutes. After the pyrotechnics are consumed, the marine marker fills with seawater and sinks. Iron and aluminum constitute 35 percent of the marker by weight. To produce the lengthy smoke effect, approximately 40 percent of the marker by weight is made up of pyrotechnic materials. The propellant, explosive, and pyrotechnic constituents of MK 58 include red phosphorus (2.19 lb.) and manganese (IV) dioxide (1.40 lb.). Other constituents include magnesium powder (0.29 lb.), zinc oxide (0.12 lb.), nitrocellulose (0.000017 lb.), nitroglycerin (0.000014 lb.), and potassium nitrate (0.2 lb.). The failure rate of marine markers is approximately 5 percent (U.S. Department of the Navy, 2010c, 2010d).

Flares are used to signal, to illuminate surface areas at night in search and attack operations, and to assist with search and rescue activities. They range in weight from 5 to 14 kg. The major constituents of flares include magnesium granules and sodium nitrate. Containers are constructed of aluminum, and the entire assembly is usually consumed during flight. Flares may also contain a primer such as TNT, propellant (ammonium perchlorate), and other explosives. These materials are present in small quantities (e.g., $1.0 \times 10^{-4}$ ounces [oz.] of ammonium perchlorate and $1.0 \times 10^{-7}$ oz. of explosives). Small amounts of metals are used to give flares and other pyrotechnic materials bright and distinctive colors. Combustion products from flares include magnesium oxide, sodium carbonate, carbon dioxide, and water. Illuminating flares and marine markers are usually entirely consumed during use; neither is intended to be recovered. Table F-5 summarizes the components of markers and flares (U.S. Department of the Air Force, 1997).

**Table F-5: Summary of Components of Marine Markers and Flares**

| Flare or Marker | Constituents | Composition (%) |
|---|---|---|
| LUU-2 Paraflare | Magnesium granules, sodium nitrate, aluminum, iron, trinitrotoluene (TNT), royal demolition explosive, ammonium perchlorate, potassium nitrate, lead, chromium, magnesium, manganese, nickel | Magnesium (54), sodium nitrate (26), aluminum (14), iron (5) |
| MK45 Paraflare | Aluminum, sodium nitrate, magnesium powder, nitrocellulose, TNT, copper, lead, zinc, chromium, manganese, potassium nitrate, pentaerythritol-tetranitrate, nickel, potassium perchlorate | Magnesium (45), sodium nitrate (30), aluminum (22) |
| MK58 Marine Marker | Aluminum, chromium, copper, lead, lead dioxide, manganese dioxide, manganese, nitroglycerin, red phosphorus, potassium nitrate, silver, zinc, zinc oxide | Iron (60), aluminum (35) |

Most of the pyrotechnic components of marine markers are consumed and byproducts are released into the air. Thereafter, the aluminum and steel canister sinks to the bottom. Combustion of red phosphorus produces phosphorus oxides, which have a low toxicity to aquatic organisms. The amount of flare residue is negligible. Phosphorus contained in the marker settles to the seafloor, where it reacts with the water to produce phosphoric acid until all phosphorus is consumed by the reaction. Phosphoric acid is a variable, but normal, component of seawater (U.S. Department of the Navy, 2006a). The aluminum and iron canisters are expected to be covered by sand and sediment over time, to become encrusted by chemical corrosion, or to be covered by marine plants and animals. Elemental aluminum in seawater tends to be converted by hydrolysis to aluminum hydroxide, which is relatively insoluble, adheres to particulates, and is transported to the bottom sediments (Monterey Bay Research Institute, 2010).

Red phosphorus, the primary pyrotechnic ingredient, constitutes 18 percent of the marine marker by weight. Toxicological studies of red phosphorus revealed an aquatic toxicity in the range of 10–100 milligrams per liter (10–100 ppm) for fish, *Daphnia* (a small aquatic crustacean), and algae (European Flame Retardants Association, 2002). Red phosphorus slowly degrades by chemical reactions to phosphine and phosphorus acids. Phosphine is very reactive and usually undergoes rapid oxidation. The final products, phosphates, are harmless (U.S. Department of the Navy, 2010c, 2010d). A study by the U.S. Department of the Air Force (1997) found that, in salt water, the degradation products of flares that do not function properly include magnesium and barium.

Chaff is an electronic countermeasure designed to confuse enemy radar by deflecting radar waves and thereby obscuring aircraft, ships, and other equipment from radar tracking sources. Chaff consists of small, thin glass fibers coated in aluminum that are light enough to remain in the air anywhere from

10 minutes to 10 hours (Farrell & Siciliano, 2007). Chaff is typically packaged in cylinders and contain a few million fibers. Chaff may be deployed from an aircraft or may be launched from a surface vessel.

The major components of the chaff glass fibers and the aluminum coating are provided in Table F-6 (Arfsten et al., 2002; Farrell & Siciliano, 2007; U.S. Department of the Air Force, 1997; U.S. Department of the Navy, 1999).

Factors influencing chaff dispersion include the altitude and location where it is released, prevailing winds, and meteorological conditions (Spargo, 2007; U.S. Department of the Navy, 1999). Doppler radar has tracked chaff plumes containing approximately 900 grams of chaff drifting 200 miles from the point of release, with the plume covering a volume of greater than 400 cubic miles (Arfsten et al., 2002). Based on the dispersion characteristics of chaff, large areas of open water would be exposed to chaff, but the chaff concentrations would be low.

**Table F-6: Major Components of Chaff**

| ***Component*** | ***Percent by Weight*** |
|---|---|
| ***Glass Fiber*** | |
| Silicon dioxide | 52–56 |
| Alumina | 12–16 |
| Calcium oxide, magnesium oxide | 16–25 |
| Boron oxide | 8–13 |
| Sodium oxide, potassium oxide | 1–4 |
| Iron oxide | ≤ 1 |
| ***Aluminum Coating*** | |
| Aluminum | 99.45 (min.) |
| Silicon and Iron | 0.55 (max.) |
| Copper | 0.05 |
| Manganese | 0.05 |
| Zinc | 0.05 |
| Vanadium | 0.05 |
| Titanium | 0.05 |
| Others | 0.05 |

Chaff is generally resistant to chemical weathering and likely remains in the environment for long periods. However, all the components of chaff’s aluminum coating are present in seawater in trace amounts, except magnesium, which is present at 0.1 percent (Nozaki, 1997). Aluminum is the most common metal in the Earth’s crust and also occurs naturally in trace amounts in the aquatic environment. Aluminum oxide and silicon dioxide are the two most common minerals in the earth’s crust, and ocean waters are constantly exposed to both minerals, so the addition of small amounts of chaff would not affect water quality or sediment composition (U.S. Department of the Navy, 1999).

The dissolved concentration of aluminum in seawater ranges from 1 to 10 micrograms per liter (1 to 10 parts per billion). For comparison, the concentration in rivers is 50 micrograms per liter (50 parts per billion). In the ocean, aluminum concentrations tend to be higher on the surface, lower at middle depths, and higher again at the bottom (Li et al., 2008). Aluminum is a very reactive element and is seldom found as a free metal in nature except under highly acidic (low pH) or alkaline (high pH) conditions. It is found combined with other elements, most commonly with oxygen, silicon, and fluorine. These chemical compounds are commonly found in soil, minerals, rocks, and clays (Agency for Toxic Substances and Disease Registry, 2008; U.S. Department of the Air Force, 1994). Elemental aluminum in seawater tends to be converted by hydrolysis to aluminum hydroxide, which is relatively insoluble, and is scavenged by particulates and transported to bottom sediments (Monterey Bay Research Institute, 2010).

Because of their light weight, chaff fibers tend to float on the water surface for a short period. The fibers are quickly dispersed by waves and currents. They may be accidentally or intentionally ingested by marine life, but the fibers are non-toxic. Chemicals leached from the chaff would be diluted by the surrounding seawater, reducing the potential for chemical concentrations to reach levels that can affect sediment quality or benthic habitats.

Schiff (1977) placed chaff samples in Chesapeake Bay water for 13 days. No increases in concentration of greater than 1 ppm of aluminum, cadmium, copper, iron, or zinc were detected. Accumulation and concentration of chaff constituents is not likely under natural conditions. A U.S. Air Force study of chaff analyzed nine elements under various pH conditions: silicon, aluminum, magnesium, boron, copper, manganese, zinc, vanadium, and titanium. Only four elements were detected above the 0.02 milligrams per liter detection limit (0.02 ppm): magnesium, aluminum, zinc, and boron (U.S. Department of the Air Force, 1994). Tests of marine organisms detected no effects of chaff exposure at levels above those expected in the Study Area (Farrell & Siciliano, 2007).

## F.3 Vegetation

### F.3.1 Acoustic Stressors

Acoustic stressors are not applicable to vegetation and are therefore not analyzed further in this section.

### F.3.2 Explosive Stressors

Single-celled algae may overlap with underwater and sea surface explosion locations. If single-celled algae are in the immediate vicinity of an explosion, only a small number of individuals are likely to be affected relative to their total population level. Additionally, the extremely fast growth rate and ubiquitous distribution of phytoplankton (Caceres et al., 2013; Levinton, 2013) suggest no meaningful effect on the resource. The low number of explosions relative to the amount of single-celled algae in the Study Area also decreases the potential for effects on these vegetation types. Based on these factors, the effect on these types of vegetation would not be detectable and they are not discussed further in this section.

Macroalgae and marine vascular plants that are attached to the seafloor may occur in locations where explosions are conducted and may be adversely affected for different reasons. Much of the attached macroalgae grows on live hard bottom that would be mostly protected in accordance with Navy mitigation measures. Visual observation mitigation occurs for explosive activities to observe for floating vegetation prior to commencing firing or an explosive detonation until the floating vegetation is clear from the mitigation zone. For mitigation, the term “floating vegetation” refers specifically to floating concentrations of detached kelp paddies and Sargassum. Many of these activities will not occur in seafloor resource mitigation areas, which would benefit vegetation that occurs there.

Attached macroalgae grow quickly and are resilient through high levels of wave action (Mach et al., 2007), which may aid in their ability to withstand underwater explosions that occur near them. Attached macroalgae typically need hard or artificial substrate in order to grow. The potential distribution of attached macroalgae can be inferred by the presence of hard or artificial substrate that occurs at depths of less than 200 m throughout the Study Area. See Section 3.2 (Sediments and Water Quality) for information regarding the distribution of hard substrate in the Study Area. If attached macroalgae are in the immediate vicinity of an explosion, only a small number of them are likely to be affected relative to their total population level. Only explosions occurring on or at shallow depth beneath the surface have the potential to affect floating macroalgae. Sea surface or underwater explosions could uproot or damage marine vascular plants if activities overlap with areas where they are rooted.

The potential for marine vascular plants (seagrass and eelgrass) to be affected by underwater and surface explosions is unlikely as seagrass and eelgrass may have very limited overlap with explosives training areas. Eelgrass are much less resilient to disturbance than marine algae; regrowth after uprooting can take up to 10 years (Dawes et al., 1997). Explosions may also temporarily increase the turbidity (sediment suspended in the water) of nearby waters, but the sediment would settle to pre-explosion conditions within a number of days. Sustained high levels of turbidity may reduce the amount of light that reaches vegetation. This scenario is not likely because seagrass and eelgrass do not overlap with explosives training areas.

## F.3.3 Energy Stressors

Energy stressors are not applicable to vegetation and are therefore not analyzed further in this section.

## F.3.4 Physical Disturbance and Strike Stressors

### F.3.4.1 Effects from Vessels and In-Water Devices

Several different types of vessels (ships, submarines, boats, amphibious vehicles) and in-water devices (e.g., towed devices, unmanned underwater vehicles) are used during training and testing activities throughout the Study Area, as described in Chapter 2 (Description of Proposed Action and Alternatives). Vessel and in water device movements occur intermittently, are variable in duration (ranging from a few hours to a few weeks) and are dispersed throughout the Study Area. Events involving large vessels are widely spread over offshore areas, while smaller vessels are more active in nearshore areas.

The potential effects from Navy vessels and in-water devices used during training and testing activities on vegetation are based on the vertical distribution of the vegetation. Vessels and in-water devices may affect vegetation by striking or disturbing vegetation on the sea surface or seafloor (Spalding et al., 2003). In the open ocean, marine algae on the sea surface such as kelp paddies have a patchy distribution. Marine algae could be temporarily disturbed if struck by moving vessels or by the propeller action of transiting vessels. These strikes could also injure the organisms that inhabit kelp paddies or other marine algal mat, such as sea turtles, seabirds, marine invertebrates, and fish. Marine algae are resilient to winds, waves, and severe weather that could sink the mat or break it into pieces. Effects on marine algae by strikes may collapse the pneumatocysts (air sacs) that keep the mats afloat. Evidence suggests that some floating marine algae will continue to float even when up to 80 percent of the pneumatocysts are removed (Zaitsev, 1971).

Vegetation on the seafloor, such as marine vascular plants and macroalgae, may be disturbed by amphibious combat vehicles, and manned and unmanned underwater vehicles. Seagrasses are resilient to the lower levels of wave action that occur in sheltered estuarine shorelines, but are susceptible to vessel propeller scarring (Sargent et al., 1995). Seagrasses could take up to 10 years to fully regrow and recover from propeller scars (Dawes et al., 1997). Seafloor macroalgae may be present in locations where these vessels occur, but the effects would be minimal because of their resilience, distribution, and biomass. Because seafloor macroalgae in coastal areas are adapted to natural disturbances, such as storms and wave action that can exceed 32.8 ft. (10 meters [m]) per second (Mach et al., 2007), macroalgae will quickly recover from vessel movements. Macroalgae that is floating in the area may be disturbed by amphibious combat vehicle activities, but the effect would not be detectable because of the small amount of macroalgae in areas where these activities occur and will not be considered further in this section.

Towed in-water devices include towed targets that are used during activities such as missile exercises and gun exercises. These devices are operated at low speeds either on the sea surface or below it. The analysis of in-water devices will focus on towed surface targets because of the potential for effects on marine algae.

Unmanned underwater vehicles and autonomous underwater vehicles are used in training and testing activities in the Study Area. They are typically propeller driven and operate within the water column or crawl along the seafloor. The propellers of these devices are typically encased, eliminating the potential for seagrass propeller scarring. Although algae on the seafloor could be disturbed by these devices, unmanned underwater vehicles are not expected to compromise the health or condition of algae for the same reasons given for vessel disturbance.

Estimates of relative vessel and in-water device use and location for each alternative are provided in Section 3.0.3.3.4.1 (Vessels and In-Water Devices). These estimates are based on the number of activities predicted for each alternative. While these estimates provide a prediction of use, actual Navy vessel use depends upon military training requirements, deployment schedules, annual budgets, and other unpredictable factors. Testing and training concentrations are most dependent upon locations of Navy shore installations and established testing and training areas.

Because of the quantity of vessel traffic in Hawaiian nearshore waters since the 1940s (especially in waters off Oahu and within Pearl Harbor), it is thought that the existing vegetation community has shifted to dominance of species which are adapted to disturbance (Coles et al., 1997). In San Diego Bay, there are anticipated to be movements of Navy small boats, divers, and swimmers over eelgrass; otherwise, eelgrass beds are avoided to the maximum practicable extent. Because of the dredging history of San Diego Bay near the Navy ship berths, it is anticipated that any nearby vegetation is accustomed to increased sedimentation and disturbance from these activities; therefore effects from vessel movements on vegetation are expected to be similar and minimal (U.S. Department of the Navy, 2013d).

In open ocean areas, vessel strikes of vegetation would be limited to floating marine algae. Vessel and in-water device movements may disperse or injure floating algal mats. Because algal distribution is patchy, mats may re-form, and events would be on a small spatial scale, Navy training activities involving vessel movement would not affect the general health of marine algae. Navy mitigation measures would ensure that vessels avoid large algal mats, such as detached kelp paddies or Sargassum, or other sensitive vegetation that other marine life depend on for food or habitat; these measures would safeguard this vegetation type from vessel strikes. This Standard Operating Procedure for vessels is "Watch personnel monitor their assigned sectors for any indication of danger to the ship and the personnel on board, such as a floating or partially submerged object or piece of debris, periscope, surfaced submarine, wisp of smoke, flash of light, or surface disturbance." In addition, as a standard collision avoidance procedure, prior to deploying a towed in-water device from a manned platform, the Navy searches the intended path of the device for any floating debris, objects, or animals (e.g., driftwood, concentrations of floating vegetation, marine mammals) that have the potential to obstruct or damage the device. This standard operating procedure benefits marine mammals, sea turtles, and vegetation through a reduction in the potential for physical disturbance and strike by a towed in-water device.

#### F.3.4.2 Effects from Military Expended Materials

Military expended materials can potentially affect marine vascular plants on the seafloor by disturbing, crushing, or shading, which may interfere with photosynthesis. In the event that a marine vascular plant is not able to photosynthesize, its ability to produce energy is compromised. However, the intersection of marine vascular plants and military expended materials is limited. Marine vascular plants generally grow in waters that are sheltered from wave action such as estuaries, lagoons, and bays (Phillips & Meñez, 1988). Locations for the majority of Navy training and testing activities where military materials are expended do not provide this type of habitat. The potential for detectable effects on marine vascular plants from expended materials would be based on their size or low density (e.g., small projectiles, small decelerators/parachutes, endcaps, and pistons) of the majority of the materials that could be used in or drift into these areas from offshore. Larger, denser materials, such as non-explosive practice munitions and sonobuoys would be used farther offshore and are likely to sink rapidly where they land. Falling materials could cause bottom sediments to be suspended. Resuspension of the sediment could affect water quality and decrease light exposure, but since it would be short-term (hours), stressors from military expended materials would not likely affect the general health of marine vascular plants.

The following are descriptions of the types of military expended materials that could affect marine algae and marine vascular plants. Marine algae could overlap with military expended materials anywhere in the Study Area; however, the Silver Strand Training Complex is the only location in the Study Area where these materials could overlap with marine vascular plants.

**Small-, Medium-, and Large-Caliber Projectiles**. Small-, medium-, and large-caliber non-explosive practice munitions, or fragments of high-explosive projectiles, expended during training and testing activities rapidly sink to the seafloor. The majority of these projectiles would be expended in the open ocean areas of the Hawaii Study Area and California Study Area. Because of the small sizes of the projectiles and their casings, damage to marine vegetation is unlikely. Large-caliber projectiles are primarily used in offshore areas at depths greater than 85 ft., while small- and medium-caliber projectiles may be expended in both offshore and coastal areas (at depths mostly less than 85 ft.) within special use airspace in the California Study Area Warning Area 291 (W-291) and at selected areas on San Clemente Island (SCI). Marine algae could occur where these materials are expended, but seagrasses generally do not because these activities do not normally occur in water that is shallow enough for seagrass to grow.

**Bombs, Missiles, and Rockets**. Bombs, missiles, and rockets, or their fragments (if high-explosive) are expended offshore (at depths mostly greater than 85 ft.) during training and testing activities, and rapidly sink to the seafloor. Marine algae could occur where these materials are expended. However, marine vascular plants generally would not occur where these materials are expended because these activities do not normally occur in water that is shallow enough for marine vascular plants to grow.

**Decelerators/Parachutes**. Decelerators/parachutes of varying sizes are used during training and testing activities. The types of activities that use decelerators/parachutes, the physical characteristics of these expended materials, where they are used, and the number of activities that would occur under each alternative are discussed in Section F.3.5 (Entanglement Stressors). Kelp, other marine algae, and marine vascular plants could occur where these materials are expended.

**Targets**. Many training and testing activities use targets. Targets that are hit by munitions could break into fragments. Target fragments vary in size and type, but most fragments are expected to sink. Pieces of targets that are designed to float are recovered when practical. Target fragments would be spread out over large areas. Marine algae could occur where these materials are expended.

**Countermeasures**. Defensive countermeasures (e.g., chaff, flares, and acoustic devices) are used to protect against incoming weapons (e.g., missiles). Chaff is made of aluminum-coated glass fibers, and flares are pyrotechnic devices. Chaff, chaff canisters, and flare end caps are expended materials. Chaff and flares are dispensed from aircraft or fired from ships. Floating marine algal mats could occur in any of the locations that these materials are expended.

#### F.3.4.3 Effects from Seafloor Devices

Most seafloor device use would occur in the California Study Area. Seafloor devices use sandy substrates, devoid of marine vegetation, to the greatest extent practicable. Marine plant species found within the relatively shallow waters of the Study Area, including the Hawaii Range Complex and off SCI, are adapted to natural disturbance and recover quickly from storms, as well as from wave and surge action. Bayside marine plant species, such as eelgrass, are found in areas where wave action is minimal. Installation of seafloor devices may affect vegetation in benthic habitats, but the effects would be temporary and would be followed by rapid (i.e., within a few weeks) recovery, particularly in oceanside boat lanes in nearshore waters off San Diego and in designated training areas adjoining SCI. Eelgrass beds show signs of recovery after a cessation of physical disturbance; the rate of recovery is a function of the severity of the disturbance (Neckles et al., 2005). The main factors that contribute to eelgrass recovery include improving water quality and cessation of major disturbance activities (e.g., dredging) (Chavez, 2009). The Navy has used credits from the Navy Region Southwest San Diego Bay Eelgrass Mitigation Bank (Bank) to offset unavoidable eelgrass and other habitat effects from infrastructure projects and testing and training activities in San Diego Bay (U.S. Department of the Navy, 2023).

New range modernization and sustainment activities include installation of undersea cables integrated with hydrophones and underwater telephones to sustain the capabilities of the Southern California Anti-Submarine Warfare Range. Deployment of fiber-optic cables along the seafloor would occur in three locations: south and west of SCI in the California Study Area, to the northeast of Oahu in the Hawaii Study Area, and to the west of Kauai in the Hawaii Study Area. In all locations the installations would occur completely within the water; no land interface would be involved. Cable-laying activities in the California Study Area could disturb marine vegetation when the cable crosses rocky substrate at depths between 65 and 196 ft. (20 and 60 m) for the Shallow Water Training Range Extension. However, it is anticipated that rocky substrate would be avoided to the greatest extent possible throughout the cable corridor to minimize these effects.

Installation and maintenance of underwater platforms, mine warfare training areas, and installation of other training areas involve seafloor disturbance where those activities would take place. Each installation would occur on soft, typically sandy bottom, avoiding rocky substrates.

#### F.3.4.4 Effects from Pile Driving

Pile driving would not affect vegetation on the sea surface, such as marine algal mats; therefore, floating vegetation will not be discussed further in this section. Pile driving would occur only in Port Hueneme Harbor near existing piers where the area is disturbed.

Pile driving could affect marine vascular plants and seafloor macroalgae by physically removing vegetation (e.g., uprooting); crushing vegetation; temporarily increasing the turbidity (sediment suspended in the water) of waters nearby; or shading, which may interfere with photosynthesis. If vegetation is not able to photosynthesize, its ability to produce energy is compromised. However, the intersection of marine macroalgae and marine vascular plants with pile driving is limited, and any suspended sediments would settle in a few days.

In bay areas, recovery of marine vascular plants such as eelgrass from direct disturbance by pile driving would occur over longer timeframes. Eelgrass beds show signs of recovery after a cessation of physical disturbance; the rate of recovery is a function of the severity of the disturbance (Neckles et al., 2005). The main factors that contribute to eelgrass recovery include improved water quality and cessation of major disturbance activities (e.g., dredging) (Chavez, 2009). Pile driving, in contrast to dredging, has a minor effect that is limited to the area of the actual pile and footprint of the mooring.

### F.3.5 Entanglement Stressors

Entanglement stressors are not applicable vegetation and are therefore not analyzed further in this section.

### F.3.6 Ingestion Stressors

Ingestion stressors are not applicable to vegetation and are therefore not analyzed further in this section.

### F.3.7 Secondary Stressors

This section analyzes potential effects on marine vegetation exposed to stressors indirectly through effects on habitat and prey availability.

#### F.3.7.1 Effects on Habitat

Section 3.2 (Sediments and Water Quality) and Section 3.5 (Habitats) consider the effects on marine sediments and water quality and abiotic habitats from explosives and explosion byproducts, metals, chemicals other than explosives, and other materials (marine markers, flares, chaff, targets, and miscellaneous components of other materials). One example from the sediment and water quality analysis of a local effect on water quality could be an increase in cyanobacteria associated with munitions deposits in marine sediments. Cyanobacteria may proliferate when iron is introduced to the marine environment, and this proliferation can affect adjacent habitats by releasing toxins and can create hypoxic conditions. Introducing iron into the marine environment from munitions or infrastructure is not known to cause toxic red tide events; rather, these harmful events are more associated with natural causes (e.g., upwelling) and the effects of other human activities (e.g., agricultural runoff and other coastal pollution) (Hayes et al., 2007). High-order explosions consume most of the explosive material, leaving only small or residual amounts of explosives and combustion products. Many combustion products are common seawater constituents. All combustion products are rapidly diluted by ocean currents and circulation (see Section 3.2.3.1, Explosives and Explosives

Byproducts). Explosives byproducts from high-order detonations present no indirect stressors to marine vegetation through sediment or water.

The analysis included in Section 3.2 (Sediments and Water Quality) determined that neither state nor federal standards or guidelines for sediments or water quality would be violated by the No Action Alternative, Alternative 1, or Alternative 2. Because standards for sediment and water quality would not be violated, population-level effects on marine vegetation are not likely to be detectable and are therefore inconsequential. Therefore, because these standards and guidelines are structured to protect human health and the environment, and the proposed activities do not violate them, no indirect effects are anticipated on vegetation from the training and testing activities proposed by the No Action Alternative, Alternative 1, or Alternative 2.

Other materials that are re-mobilized after their initial contact with the seafloor (e.g., by waves or currents) may continue to strike or abrade marine vegetation. Secondary physical strike and disturbances are relatively unlikely because most expended materials are denser than the surrounding sediments (e.g., metal) and are likely to remain in place as the surrounding sediment moves. Potential secondary physical strike and disturbance effects may cease when (1) the military expended material is too massive to be mobilized by typical oceanographic processes, (2) the military expended material becomes encrusted by natural processes and incorporated into the seafloor, or (3) the military expended material becomes permanently buried. Although individual organisms could be affected by secondary physical strikes, the viability of populations or species would not be affected.

#### F.3.7.2 Effects on Prey Availability

Prey availability as a stressor is not applicable to vegetation and will not be analyzed further in this section.

## F.4 Invertebrates

### F.4.1 Acoustic Stressors

#### F.4.1.1 Background

A summary of available information related to each type of effect is presented in the following sections. Some researchers discuss effects in terms of the acoustic near field and far field. The near field is an area near a sound source where considerable interference between sound waves emerging from different parts of the source is present. Amplitude may vary widely at different points within this acoustically complex zone, and sound pressure and particle velocity are generally out of phase. The far field is the distance beyond which sound pressure and particle velocity are in phase, all sound waves appear to originate from a single point, and pressure levels decrease predictably with distance. The boundary between the near and far field is frequency-dependent, with the near field extending farther at lower frequencies. It has been estimated that the near field for a sound of 500 Hertz (Hz) (intensity not specified) would extend about 3 m from the source (Myrberg, 2001).

##### F.4.1.1.1 Injury

Injury refers to the direct effects on the tissues or organs of an animal due to exposure to pressure waves or particle motion. Available information on injury to invertebrates resulting from acoustic sources pertains mostly to damage to the statocyst, an organ sensitive to water particle motion and responsible for balance and orientation in some invertebrates. A few studies have also investigated effects to appendages and other organs, and one study investigated zooplankton mortality in response to air gun firing.

Researchers have investigated the effects of noise on American lobsters exposed to air gun firings in an aquarium and in the field (Payne et al., 2007). Lobsters in the aquarium were placed about 3.5 m from the air guns and exposed to sound levels of about 200 dB (peak-to-peak). Caged lobsters in the field were located 2 m from the air guns and exposed to higher-intensity sound levels (about 230 dB peak-to-peak). No physical damage to appendages and no effects on balance or orientation (indicating no damage to statocysts) were observed in any lobsters. No visible evidence of damage to hepatopancreata (digestive glands) or ovaries were found. Caged snow crabs (*Chionoecetes opilio*) were exposed to repeated air gun firings in the field (Christian et al., 2003). Crabs exposed to a single air gun were placed at depths of 2–15 m, while crabs exposed to air gun arrays were placed at depths of 4–m. Air guns were fired during multiple sessions, with each session consisting of a firing every 10 seconds for 33 minutes. Peak received levels were up to 207 dB re 1 µPa and 187 decibels referenced to 1 squared micropascal (dB re 1 µPa$^2$) (single gun), and 237 dB re 1 µPa and 175 dB re 1 µPa$^2$ (array). Post-experimental examination showed no physical damage to statocysts, hepatopancreata, heart muscle or surrounding tissue, carapace, or appendages. As a comparison, air guns operated at full capacity during Navy activities would produce an SPL of approximately 206 dB re 1 µPa root mean squared (rms) and a sound exposure level (SEL) of 185–196 decibels referenced to 1 micropascal squared per second (dB re 1 µPa$^2$-s) at a distance 1 m from the air gun. Air guns are also operated at less than full capacity, resulting in reduced sound levels.

In three instances, seismic air gun use has been hypothesized as the cause of giant squid strandings. This was based on the proximity in time and space of the squid and operating seismic vessels and, in two of the events, to physical injuries considered consistent with exposure to impulsive acoustic waves (Guerra & Gonzales, 2006; Guerra et al., 2004; Leite et al., 2016). However, because the animals were not observed at the time of potential effect, the cause(s) of the injuries and strandings cannot be determined conclusively.

Zooplankton abundance and mortality was investigated in the context of exposure to air gun firings in an open ocean environment (McCauley et al., 2017). Net tows and sonar surveys were conducted after transects involving air gun firings were completed. The results indicated decreased zooplankton abundance and increased mortality as a result of exposure. The most abundant organisms (copepods and cladocerans [water fleas]) showed a 50 percent decrease in abundance at distances of about 500 to 700 m from the source. Received noise level at this distance was about 156 dB re 1 µPa$^2$-s SEL and 183 dB re 1 µPa peak-to-peak. There was no effect on the abundance of these specific taxa at distances of about 1 kilometer (km) from the source (153 dB re 1 µPa$^2$-s SEL and 178 dB re 1 µPa peak-to-peak). However, an overall decrease in zooplankton abundance was reported at distances to about 1.2 km from the source. The authors speculated that the effects could have been caused by damage to external sensory hairs on the organisms.

Physiological studies of wild captured cephalopods found progressive damage to statocysts in squid and octopus species after exposure to two hours of low-frequency (50–400 Hz) sweeps (100 percent duty cycle) at SPL of 157 to 175 dB re 1 µPa (André et al., 2011; Sole et al., 2013). It is noted that the animals were in the near field (distance was not specified in the report, but animals were likely within a few to several feet of the sound source based on the experiment description), where there is significant particle motion. In a similar experiment designed to control for possible confounding effects of experimental tank walls, common cuttlefish (*Sepia officinalis*) were exposed to two hours of low-frequency sweeps (100 to 400 Hz; 100 percent duty cycle with a 1-second sweep period) in an offshore environment (Sole et al., 2017). Sounds were produced by a transducer located near the surface, and caged experimental animals were placed at depths between 7 and 17 m. Received sound levels ranged from 139 to 142 dB re 1 µPa$^2$. Maximum particle motion of 0.7 meter per squared second was recorded at the cage nearest the transducer (7.1 m between source and cage). Progressive damage to sensory hair cells of the statocysts were found immediately after and 48 hours after sound exposure, with the severity of effects being proportional to distance from the transducer. The authors suggest that whole-body vibrations resulting from particle motion were transmitted to the statocysts, causing damage to the structures. Statocyst damage was also found in captive individuals of two jellyfish species (Mediterranean jellyfish [*Cotylorhiza tuberculata*] and barrel jellyfish [*Rhizostoma pulmo*]) under the same exposure parameters (50–400 Hz sweeps; two-hour exposure time; 100 percent duty cycle with a one-second sweep period; approximately 157 to 175 dB re 1 µPa received SPL) (Sole et al., 2016). In the context of overall invertebrate population numbers, most individuals exposed to acoustic stressors would be in the far field where particle motion would not occur and, therefore, the types of damage described above would not be expected. In addition, exposure duration would be substantially less than two hours.

This limited information suggests that the potential for statocyst damage may differ according to the type of sound (impulsive or continuous) or among invertebrate taxa (e.g., crustaceans and cephalopods). Therefore, a definitive conclusion regarding potential effects on invertebrates in general is unsupported. Although invertebrate occurrence varies based on location, depth, season, and time of day (for example, the rising of the deep scattering layer, which consists of numerous invertebrate taxa), individuals could be present in the vicinity of impulsive or non-impulsive sounds produced by Navy activities. Estimation of invertebrate abundance at any particular location would generally not be feasible, but there is a general pattern of higher abundances in relatively productive estuarine and nearshore waters compared to abundances in offshore portions of the Study Area. The number of individuals affected would be influenced by sound sensing capabilities. As discussed in Section 3.4.3.1 (Acoustic Stressors), invertebrate acoustic sensing is probably limited to the particle motion component of sound. Water particle motion is most detectable near a sound source and at lower frequencies, which likely limits the range at which invertebrates can detect sound.

#### F.4.1.1.2 Physiological Stress

A stress response consists of one or more physiological changes (e.g., production of certain hormones) that help an organism cope with a stressor. However, if the magnitude or duration of the stress response is too great or too prolonged, there can be negative consequences to the organism. Physiological stress is typically evaluated by measuring the levels of relevant biochemicals in the subject organisms.

The results of two investigations of physiological stress in adult invertebrates caused by impulsive noise varied by species. Some biochemical stress markers and changes in osmoregulation were observed in American lobsters exposed to air gun firings at distances of approximately 2–4 m from the source (Payne et al., 2007). Increased deposits of carbohydrates, suggesting a possible stress response, were noted in digestive gland cells four months after exposure. Conversely, repeated air gun exposures caused no changes in biochemical stress markers in snow crabs located from 2 to 170 m from the source (Christian et al., 2003).

Several investigations of physiological reactions of captive adult invertebrates exposed to boat noise playback and other continuous noise have been conducted. Continuous exposure to boat noise playback resulted in changes to some biochemical levels indicating stress in common prawns (*Palaemon serratus*) (30-minute exposure to sound levels of 100 to 140 dB re 1 µPa rms) and European spiny lobsters

(30-minute exposure to sound levels up to 125 dB re 1 µPa rms) (Celi et al., 2015; Filiciotto et al., 2016; Filiciotto et al., 2014). Increased oxygen consumption, potentially indicating stress, was found in shore crabs exposed to ship-noise playback of 148 to 155 dB re 1 µPa for 15 minutes (Wale et al., 2013b). Red swamp crayfish (*Procambarus clarkii*) exposed to 30-minute continuous acoustic sweeps (frequency range of 0.1 to 25 kilohertz (kHz), peak amplitude of 148 dB rms at 12 kHz) showed changes in some biochemical levels indicating stress (Celi et al., 2013). Captive sand shrimp (*Crangon crangon*) exposed to low-frequency noise (30 to 40 dB above ambient) continuously for three months demonstrated decreases in growth rate and reproductive rate (Lagardère, 1982). Mediterranean mussels (*Mytilus galloprovincialis*) exposed to 30-minute continuous acoustic sweeps (frequency range of 0.1 to 60 kHz, maximum SPL of 150 dB rms re 1 µPa), although exhibiting no behavioral changes at any tested frequency, showed statistically significant increases in some biochemical stress indicators (e.g., glucose and heat shock protein) in the low-frequency exposure category (0.1 to 5 kHz) (Vazzana et al., 2016). Changes in glucose levels were found in blue crabs (*Callinectes sapidus*) exposed to low-frequency sound (broadband noise with a significant component of 60 Hz at approximately 170 dB re 1 µPa SPL) and mid-frequency pulsed tones and chirps (1.7 to 4 kHz at approximately 180 dB re 1 µPa SPL) (Dossot et al., 2017).

In addition to experiments on adult invertebrates, some studies have investigated the effects of impulsive and non-impulsive noise (air guns, boat noise, turbine noise) on invertebrate eggs and larvae. Data on similar effects resulting from sonar are currently unavailable. Developmental delays and body malformations were reported in New Zealand scallop (*Pecten novaezelandiae*) larvae exposed to seismic air gun playbacks at frequencies of 20 Hz to 22 kHz with SPL of 160–164 dB re 1 µPa (Aguilar de Soto et al., 2013). Although uncertain, the authors suggested physiological stress as the cause of the effects. Larvae in the relatively small (2 m diameter) experimental tank were considered close enough to the acoustic source to experience particle motion, which would be unlikely at the same pressure levels in the far field. Playbacks occurred once every three seconds and the larvae were periodically examined over the course of 90 hours. Snow crab (*Chionoecetes opilio*) eggs located in 2 m water depth and exposed to repeated firings of a seismic air gun (peak received SPL was 201 dB re 1 µPa) had slightly increased mortality and apparent delayed development (Christian et al., 2003). However, Dungeness crab (*Metacarcinus magister*) zoeae were not affected by repeated exposures to an air gun array (maximum distance of about 62 ft. slant distance) (Pearson et al., 1994), and exposure of southern rock lobster (*Jasus edwardsii*) eggs to air gun SELs of up to 182 dB re 1 $\mu Pa^2$-s did not result in embryonic developmental effects (Day et al., 2016). An investigation of the effects of boat noise playback on the sea hare (*Stylocheilus striatus*) found reduced embryo development and increased larvae mortality, but no effect on the rate of embryo development (Nedelec et al., 2014). Specimens were exposed to boat-noise playback for 45 seconds every five minutes over a 12-hour period. Continuous playback of simulated underwater tidal and wind turbine sounds resulted in delayed metamorphosis in estuarine crab larvae (*Austrohelice crassa* and *Hemigrapsus crenulatus*) that were observed for up to about 200 hours (Pine et al., 2016).

Overall, the results of these studies indicate the potential for physiological effects in some, but not all, adult invertebrates exposed to air guns near the source (about 2–4 m) and to boat and other continuous noise for durations of 15–30 minutes or longer. Larvae and egg development effects were reported for impulsive (distance from source of about 2 m) and non-impulsive noise exposures of extended duration (intermittently or continuously for several to many hours) and for air gun playback and field exposure, although air gun noise had no effect in one study. In general, exposure to continuous noise such as vessel operation during Navy training or testing events would occur over a shorter duration and sound sources would be more distant than those associated with most of the studies. Adverse effects resulting from short exposure times have not been shown experimentally. A range to effects was not systematically investigated for air gun use. Experiments using playback of air gun and boat noise were conducted in relatively small tanks where particle motion, which decreases rapidly with distance, could have been significant. Marine invertebrate egg and larval abundances are high relative to the number of adults, and eggs and larvae are typically subject to high natural mortality rates. These factors decrease the likelihood of population-level effects resulting from effects on eggs and larvae from physiological stress associated with Navy training and testing events.

#### F.4.1.1.3 Masking

Masking occurs when one sound interferes with the detection or recognition of another sound. Masking can limit the distance over which an organism can communicate or detect biologically relevant sounds. Masking can also potentially lead to behavioral changes.

Little is known about how marine invertebrates use sound in their environment. Some studies show that crab, lobster, oyster, and coral larvae and post-larvae may use nearby reef sounds when in their settlement phase. Orientation and movement toward reef sounds was found in larvae located at 60–80 m from a sound source in open water, and in experimental tanks (distance from the sound source was about 150 cm in one laboratory study) (Radford et al., 2007; Stanley et al., 2010; Vermeij et al., 2010). The component of reef sound used is generally unknown, but an investigation found that low-frequency sounds (200–1,000 Hz) produced by fish at dawn and dusk on a coral reef were the most likely sounds to be detectable a short distance from the reef (Kaplan & Mooney, 2016). Similarly, lobed star coral larvae were found to have increased settlement on reef areas with elevated sound levels, particularly in the frequency range of 25–1,000 Hz (Lillis et al., 2016). Mountainous star coral (*Orbicella faveolata*) larvae in their settlement phase were found to orient toward playbacks of reef sounds in an experimental setup, where received sound levels were about 145–149 dB re 1 µPa and particle velocity was about $9 \times 10^{-8}$ meters per second (Vermeij et al., 2010). Marine invertebrates may also use sound to communicate and avoid predators (Popper et al., 2001). Crabs (*Panopeus* species) exposed to playback of predatory fish vocalizations reduced foraging activity, presumably to avoid predation risk (Hughes et al., 2014). The authors suggest that, due to lack of sensitivity to sound pressure, crabs are most likely to detect fish sounds when the fish are nearby. Anthropogenic sounds could mask important acoustic cues such as detection of settlement cues or predators, and potentially affect larval settlement patterns or survivability in highly modified acoustic environments (Simpson et al., 2011). Low-frequency sounds could interfere with perception of low-frequency rasps or rumbles among crustaceans, particularly when conspecific sounds are produced at the far end of the hearing radius. Navy activities occurring relatively far from shore would produce transient sounds potentially resulting in only intermittent, short-term masking, and would be unlikely to affect the same individuals within a short time. Training and testing activities would generally not occur at known reef sites within the probable reef detection range of larvae. Effects could be more likely in locations where anthropogenic noise occurs frequently within the perceptive range of invertebrates (e.g., pierside locations in estuaries). There are likely many other non-Navy noise sources present in such areas, and potential effects on invertebrates would be associated with all anthropogenic sources.

#### F.4.1.1.4 Behavioral Reactions

Behavioral reactions refer to alterations of natural behaviors due to exposure to sound. Most investigations involving invertebrate behavioral reactions have been conducted in relation to air gun use, pile driving, and vessel noise. Studies of air gun effects on marine invertebrates (crustaceans and cephalopods) have typically been conducted with equipment used for seismic exploration, and the limited results suggest responses may vary among taxa. Snow crabs placed 48 m below a seismic air gun array did not react behaviorally to repeated firings (peak received SPL was 201 dB re 1 µPa) (Christian et al., 2003). Studies of commercial catch of rock lobsters (*Panulirus cygnus*) and multiple shrimp species in the vicinity of seismic prospecting showed no long-term adverse effects to catch yields, implying no detectable long-term effects on abundance from intermittent anthropogenic sound exposure over long periods (Andriguetto-Filho et al., 2005; Parry & Gason, 2006). Conversely, squid have exhibited various behavioral reactions when exposed to impulsive noise such as air gun firing (McCauley et al., 2000). Some squid showed strong startle responses, including inking, when exposed to the first shot of broadband sound from a nearby seismic air gun (received SEL of 174 dB re 1 µPa rms) Strong startle response was not seen when sounds were gradually increased, but the squid exhibited alarm responses at levels above 156 dB re 1 µPa rms (McCauley et al., 2000). Southern reef squids (*Sepioteuthis australis*) exposed to air gun noise displayed alarm responses at levels above 147 dB re 1 $\mu Pa^2$-s (Fewtrell & McCauley, 2012).

Pile driving produces sound pressure that moves through the water column and into the substrate, which may therefore affect both pelagic and benthic invertebrates. Impact pile driving produces a repetitive impulsive sound, while vibratory pile extraction produces a nearly continuous sound at a lower source level. Although few investigations have been conducted regarding effects on invertebrates resulting from impact pile driving and extraction, the effects are likely similar to those resulting from other impulsive and vibrational (e.g., drilling) sources. When an underwater sound encounters the substrate, particle motion can be generated, resulting in vibration. Invertebrates may detect and respond to such vibrations. Playback of impact pile driving sound (137–152 dB re 1 µPa peak to peak) in the water column near chorusing snapping shrimp resulted in an increase in the snap number and amplitude (Spiga, 2016). When exposed to playback of broadband impulsive pile driving sound of 150 dB SEL, Japanese carpet shell clams (*Ruditapes philippinarum*) exhibited reduced activity and valve closing, while Norway lobsters (*Nephrops norvegicus*) repressed burying, bioirrigation, and locomotion activity (Solan et al., 2016). Brittlestars (*Amphiura filiformis*) included in the experiment exhibited no overall statistically detectable behavioral changes, although the authors note that a number of individuals exhibited changes in the amount of sediment reworking activity. Pacific oysters (*Magallana gigas*) exposed to three-minute pure tones responded behaviorally (shell closure) to low-frequency sounds, primarily in the range of 10–200 Hz (Charifi et al., 2017). The oysters were most sensitive to sounds of 10–80 Hz at 122 dB rms re 1 µPa, with particle acceleration of 0.02 meter per squared second. Invertebrates exposed to vibrations of 5–410 Hz (which is a proxy for the effects of vibratory pile removal) at various particle acceleration amplitudes in the substrate of a holding tank for eight-second intervals exhibited behavioral reactions ranging from valve closure (common mussel [*Mytilus edulis*]) to antennae sweeping, changes in locomotion, and exiting the shell (common hermit crab [*Pagurus bernhardus*]) (Roberts et al., 2015; Roberts et al., 2016a). Sensitivity was greatest at 10 Hz and at particle acceleration of 0.1 m per squared second. The authors analyzed data on substrate acceleration produced by pile driving in a river and found levels that would be detectable by the hermit crabs at 17 and 34 m from the source. Measurements were not available for other distances or in marine environments. Similarly, underwater construction-related detonations of about 14-pound (lb.) charge weight (presumably in fresh water) resulted in substrate vibrations 297 m from the source that would likely be detected by crabs. Follow-up experiments showed that particle acceleration detection sensitivity in mussels and hermit crabs ranged from 0.06 to 0.55 meters per squared second (Roberts et al., 2016b). Subsequent semi-field experiments consisted of operating a small pile driver for two-hour periods in an enclosed dock (90 m long by 18 m wide, water depth of 2–3 m, and sediment depth of 3 to 4 m). Vibration in the sediment propagated farther (up to 30 m) in shallower water than in deeper water (up to 15 m). The signal in the sediment was mostly below 100 Hz and primarily from 25 to 35 Hz. Experimental animals in the enclosed area exhibited behavioral (e.g., width of shell opening) and physiological (e.g., oxygen demand) responses as a result of exposure, although information such as distance from the pile driver and particle acceleration at specific locations was not provided.

Common prawns and European spiny lobsters exposed to 30 minutes of boat noise playback in frequencies of 200 Hz to 3 kHz (sound levels of approximately 100 to 140 dB SPL [prawns] and 75 to 125 dB SPL [lobsters]) showed behavioral responses including changes in movement velocity, and distance moved, as well as time spent inside a shelter (Filiciotto et al., 2016; Filiciotto et al., 2014). Common cuttlefish exposed to playback of underwater ferry engine noise for 3.5 minutes (maximum sound level of about 140 dB re 1 µPa SPL) changed color more frequently, swam more, and raised their tentacles more often than control specimens or individuals exposed to playback of wave sounds (Kunc et al., 2014). Shore crabs (*Carcinus maenas*) exposed to ship noise playback did not exhibit changes in the ability or time required to find food, but feeding was often suspended during the playback (Wale et al., 2013a). Japanese carpet shell clams and Norway lobsters exposed to playback of ship noise for seven days at received levels of 135–140 dB re 1 µPa exhibited reactions such as reduced activity, movement, and valve closing (Solan et al., 2016). Brittlestars (*A. filiformis*) included in the study showed no overall statistically detectable behavioral changes, although individual animals were affected. Antarctic krill (*Euphausia superba*) did not respond to a research vessel approaching at 2.7 knots (source level below 150 dB re 1 µPa) (Brierley et al., 2003). Decreased activity levels were found in blue crabs exposed to low-frequency broadband sound with a significant component of 60 Hz (approximately 170 dB re 1 µPa SPL) and mid-frequency pulsed tones and chirps (1.7 –4 kHz at approximately 180 dB re 1 µPa SPL) (Dossot et al., 2017). Exposure to low-frequency sounds resulted in more pronounced effects than exposure to mid-frequency sounds. American lobsters appeared to be less affected than crabs.

A limited number of studies have investigated behavioral reactions to non-impulsive noise other than that produced by vessels. Red swamp crayfish (*Procambarus clarkii*) exposed to 30-minute continuous acoustic sweeps (frequency range of 0.1–25 kHz, peak amplitude of 148 dB rms at 12 kHz) exhibited changes in social behaviors (Celi et al., 2013). Caribbean hermit crabs (*Coenobita clypeatus*) delayed reaction to an approaching visual threat when exposed to continuous noise (Chan et al., 2010a; Chan et al., 2010b). The delay potentially put them at increased risk of predation, although the studies did not address possible simultaneous distraction of predators. Razor clams (*Sinonovacula constricta*) exposed to white noise and sine waves of 500 and 1,000 Hz responded by digging at a sound level of about 100 dB re 1 µPa (presumably as a defense reaction) but did not respond to sound levels of 80 dB re 1 µPa (Peng et al., 2016). Mediterranean mussels exposed to 30-minute continuous acoustic sweeps (frequency range of 0.1 to 60 kHz, maximum SPL of 150 dB rms re 1 µPa) showed no statistically significant behavioral changes compared to control organisms (Vazzana et al., 2016).

The results of these studies indicate that at least some invertebrate taxa would respond behaviorally to various levels of sound and substrate vibration produced within their detection capability. Comprehensive investigations of the range to effects of different sound and vibration sources and levels are not available. However, sound source levels for Navy pile diving and air gun use are within the range of received levels that have caused behavioral effects in some species (Solan et al., 2016). The low-frequency component of vessel noise would likely be detected by some invertebrates, although the number of individuals affected would be limited to those near enough to a source to experience particle motion.

#### F.4.1.2 Effects from Sonar and Other Transducers

Many non-impulsive sounds associated with training and testing activities are produced by sonar. Other transducers include items such as acoustic projectors and countermeasure devices. Most marine invertebrates do not have the capability to sense sound pressure; however, some are sensitive to nearby low-frequency sounds, such as could be approximated by some low-frequency sonars. As described in Section 3.4.3.1 (Acoustic Stressors), invertebrate species detect sound through particle motion, which diminishes rapidly with distance from the sound source. Therefore, the distance at which they may detect a sound is probably limited. Most activities using sonar or other transducers would be conducted in deep-water, offshore portions of the Study Area and are not likely to affect most benthic invertebrate species (including ESA-listed abalone species), although invertebrates in the water column could be affected. However, portions of the range complexes overlap nearshore waters of the continental shelf, and it is possible that sonar and other transducers could be used and affect benthic invertebrates in these areas. Sonar is also used in shallow water during pierside testing and maintenance testing.

Invertebrate species generally have their greatest sensitivity to sound below 1 to 3 kHz (Kunc et al., 2016) and would therefore not be capable of detecting mid- or high-frequency sounds, including the majority of sonars, or distant sounds in the Study Area. Studies of the effects of continuous noise such as boat noise, acoustic sweeps, and tidal/wind turbine sound (information specific to sonar use was not available) on invertebrates have found statocyst damage, elevated levels of biochemicals indicative of stress, changes in larval development, masking, and behavioral reactions under experimental conditions (see Section 3.4.2.1, General Background). Noise exposure in the studies generally lasted from a few minutes to 30 minutes. The direct applicability of these results is uncertain because the duration of sound exposure in many of the studies is greater than that expected to occur during Navy activities, and factors such as environmental conditions (captive versus wild conditions) may affect individual responses (Celi et al., 2013). Individuals of species potentially susceptible to statocyst damage (e.g., some cephalopods) could be physically affected by nearby noise. Available research has shown statocyst damage to occur after relatively long-duration exposures (two hours), which would be unlikely to occur to individual invertebrates due to transiting sources and potential invertebrate movement. An exception is pierside sonar testing and maintenance testing, where invertebrates (particularly sessile or slow-moving taxa such as bivalve molluscs, hydroids, and marine worms) could be exposed to sound for longer time periods compared to at-sea activities. Some studies also indicate the potential for effects on invertebrate larval development resulting from exposure to non-impulsive noise (continuous or intermittent exposures over time periods of 12 to 200 hours) although, similar to stress effects, sonar has not been studied specifically. Masking could affect behaviors such as larvae settlement, communication, predator avoidance, and foraging in mollusc, crustacean, and coral species.

#### F.4.1.3 Effects from Air Guns

Air guns produce shock waves that are somewhat similar to those produced by explosives (see Section 3.4.3.2, Explosives Stressors) but of lower intensity and slower rise times. An impulsive sound is generated when pressurized air is released into the surrounding water. Some studies of air gun effects on marine invertebrates have involved the use of an array of multiple seismic air guns, although arrays are not used during Navy training and testing activities. The volume capacity of air guns used for Navy testing (60 cubic inches at full capacity) is generally within the volume range of single air guns used in seismic exploration (typically 20–800 cubic inches). However, seismic air guns are used in arrays with a total volume of several thousands of cubic inches, which is far more than would be associated with any Navy activities. Generated impulses would have short durations, typically a few hundred milliseconds. The root-mean-squared SPL and SEL at a distance of 1 m from the air gun would be approximately 200 to 210 dB re 1 µPa and 185 to 195 dB re 1 $\mu Pa^2$-s, respectively.

The results of studies of the effects of seismic air guns on marine invertebrates, described in detail in Section 3.4.3.1 (Acoustic Stressors), suggest possible differences between taxonomic groups and life stages. Physical injury has been reported in relatively few crustaceans (crabs, shrimp, and lobsters) exposed to seismic air guns at received levels comparable to the source level of Navy air guns operated at full capacity, but one study reported injury and mortality for zooplankton at exposures below Navy source levels. Evidence of physiological stress was not found in crabs exposed to sound levels up to 187 dB re 1 $\mu Pa^2$. However, stress response was reported for lobsters located about 3.5 m from the source, where particle motion was likely detectable. While behavioral reaction to air guns has not been documented for crustaceans, squid have exhibited startle and alarm responses at various sound levels. Squid have shown startle response at received levels of 156–174 dB re 1 µPa rms (distance from sound source is unclear but presumed to be 30 m based on experimental description), although the reactions were less intense when ramp-up procedures (beginning with lower-intensity sound and progressing to higher levels) were used. In one study, onset of alarm response occurred at 147 dB re 1 $\mu Pa^2$-s; distance from the source was not provided. Developmental effects to crab eggs and scallop larvae were found at received levels of 210 and 164 dB 1 µPa SPL (about 7 ft. from the source). Conversely, crab zoeae located 62 ft. from an air gun source showed no developmental effects. Air gun use could also result in substrate vibration, which could cause behavioral effects in nearby benthic invertebrates.

#### F.4.1.4 Effects from Pile Driving

Effects on invertebrates resulting from pile driving and removal are considered in the context of impulsive sound and substrate vibration. Impact pile driving produces a pressure wave that is transmitted to the water column and the sediment (Reinhall & Dahl, 2011). The pressure wave may cause vibration within the sediment. Most acoustic energy would be concentrated below 1,000 Hz, which is within the general sound sensing range of invertebrates. Available information indicates that invertebrates may respond to particle motion and substrate vibration produced by pile driving or removal. As discussed in Section 3.4.3.1 (Acoustic Stressors), recent investigations have found effects to crustacean and mollusc species resulting from pile driving noise playback and substrate vibration (Roberts et al., 2015; Roberts et al., 2016a; Solan et al., 2016; Spiga, 2016). Responses include changes in chorusing (snapping shrimp), shell closing (clams and mussels), and changes in activity level (clams, lobsters, and hermit crabs). However, no statistically detectable changes were observed in brittlestars, suggesting that effects may vary among taxa or species. While one study was conducted in a sheltered coastal area (Spiga, 2016), the others used small experimental tanks with maximum dimension of about 20 inches (in.). Therefore, many of the effects were observed very close to the sound sources. Navy scientists are in the early stages of observing the response of marine life to pile driving in their unconfined environment using an adaptive resolution imaging sonar that allows observations in low visibility estuarine waters. Samples acquired to date include the response (or lack thereof) of various fish and crabs to Navy pile driving in the Mid-Atlantic region (Chappell, 2018).

#### F.4.1.5 Effects from Vessel Noise

Physiological effects included biochemical changes indicative of stress in crustacean species, decreased growth and reproduction in shrimp, and changes in sea hare embryo development. It is also possible that vessel noise may contribute to masking of relevant environmental sounds, such as predator detection or reef sounds. Low-frequency reef sounds are used as a settlement cue by the larvae of some invertebrate species. Behavioral effects resulting from boat noise playback have been observed in various crustacean, cephalopod, and bivalve species and include shell closing and changes in feeding, coloration, swimming, and other movements. Exposure to other types of non-impulsive noise (and therefore potentially relevant to vessel noise effects), including continuous sweeps and underwater turbine noise playback, has resulted in statocyst damage (squid and octopus), physiological stress, effects to larval development, and behavioral reactions. Noise exposure in several of the studies using boat and other continuous noise sources occurred over a duration of 3.5–30 minutes to captive individuals unable to escape the stimulus. In other studies, noise playback ranged from hours to days (and up to three months in one investigation) of continuous or intermittent exposure. Given the duration of exposure, direct applicability of the results to Navy training and testing activities is uncertain for mobile species. However, it is possible that invertebrates in the Study Area that are exposed to vessel noise could exhibit similar reactions.

While commercial vessel traffic and associated noise is relatively steady over time, Navy traffic is episodic in the ocean. Activities involving vessel movements occur intermittently and are variable in duration, ranging from a few hours to a few weeks. Vessels engaged in training and testing may consist of a single vessel involved in unit-level activity for a few hours or multiple vessels involved in a major training exercise that could last a few days within a given area. In the West Coast Exclusive Economic Zone, Navy ships are estimated to contribute roughly 10 percent of the total large vessel broadband energy noise (Mintz, 2012).

#### F.4.1.6 Effects from Weapons Noise

Underwater sound produced by weapons firing, launch, and impact of non-explosive practice munitions would be greatest near the surface and would attenuate with depth. However, the potential for in-air weapons noise to affect invertebrates would be small. Much of the energy produced by muzzle blasts and flying projectiles is reflected off the water surface. As discussed in Section 3.0.3.3.1.6 (Weapon Noise), sound generally enters the water only in a cone beneath the blast or projectile trajectory (within 13 to 14 degrees of vertical for muzzle blast noise, and 65 degrees behind the projectile in the direction of fire for projectile shock waves). An SEL of 180 to 185 dB re 1 $\mu Pa^2$-s was measured at water depth of 5 ft. directly below the muzzle blast of the largest gun analyzed, at the firing position closest to the water. Different weapons and angles of fire would produce less sound in the water. Bow waves from supersonic projectiles produce a brief “crack” noise at the surface, but transmission of sound into the water is minimal. Launch noise fades rapidly as the missile or target moves downrange and the booster burns out. Hull vibration from large-caliber gunfire produces only a small level of underwater noise. For example, analysis of 5-in. gun firing found that energy transmitted into the water by hull vibration is only 6 percent of that produced by the muzzle blast. Compared to weapons firing, launches, and hull vibration, impulsive sound resulting from non-explosive practice munition strikes on the water surface could affect a somewhat larger area, though far less than an explosive blast. Underwater sound would generally be associated only with relatively large munitions affecting at high speed.

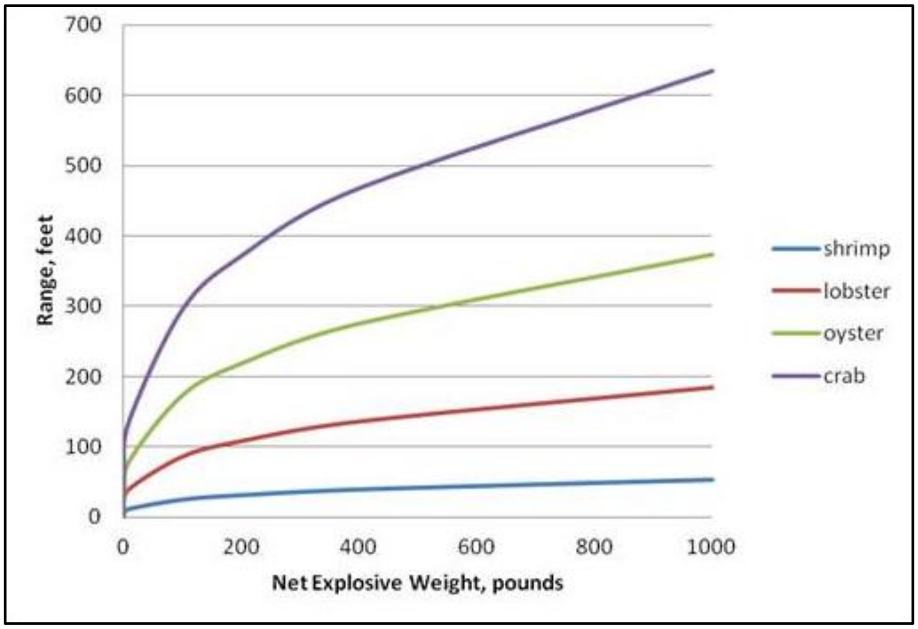
Based on the discussion above, invertebrates would likely only be affected by noise produced by muzzle blasts and impact of large non-explosive practice munitions. Effects would likely be limited to pelagic invertebrates, such as squid, jellyfish, and zooplankton, located near the surface. Injury and physiological stress has not been found in limited studies of invertebrates exposed to impulsive sound levels comparable to those produced beneath the muzzle blast of a 5-in. gun. Behavioral reactions have not been found in crustaceans, but have been observed for squid. While squid could display short-term startle response, behavioral reactions in response to sound is not known for jellyfish or zooplankton. Zooplankton may include gametes, eggs, and larval forms of various invertebrate species, including corals. Although prolonged exposure to repeated playback of nearby impulsive sound (air guns) has resulted in developmental effects to larvae and eggs of some invertebrate species, brief exposure to a single or limited number of muzzle blasts or munition impacts would be unlikely to affect development. Other factors would limit the number and types of invertebrates potentially affected. Most squid are active near the surface at night, when weapons firing and launch occur infrequently. Weapons firing and launch typically occurs greater than 12 NM from shore, which because of the water depths would substantially limit the sound level reaching the bottom. Therefore, effects on benthic invertebrates (e.g., bivalve molluscs, worms, and crabs) are unlikely.

## F.4.2 Explosive Stressors

### F.4.2.1 Background

Explosions may affect invertebrates at the water surface, in the water column, or on the bottom. The potential for effects is influenced by typical detonation scenarios and invertebrate distribution. The majority of explosions would occur in the air or at the surface, with relatively few at the bottom (Appendix A, Navy Activity Descriptions), which would decrease the potential for effects on benthic invertebrate species. Surface explosions typically occur during the day at offshore locations more than 12 NM from shore. There is a general pattern of lower invertebrate abundance in offshore portions of the Study Area compared to relatively productive estuarine and nearshore waters. Therefore, the typical offshore location of detonations would result in fewer invertebrates potentially exposed to detonation effects. In addition, invertebrate abundances in offshore surface waters tend to be lower during the day, when surface explosions typically occur, than at night.

In general, an explosion may result in direct trauma and mortality due to the associated rapid pressure changes. For example, gas-containing organs such as the swim bladder in many fish species and the lungs of marine mammals are subject to rapid contraction and overextension (potentially causing rupture) when exposed to explosive shock waves. Most marine invertebrates lack air cavities and are therefore comparatively less vulnerable to damaging effects of pressure waves. A report summarizing the results of all known historical experiments (from 1907 to the 1980s) involving invertebrates and detonations concluded that marine invertebrates are generally insensitive to pressure-related damage from underwater explosions (Keevin & Hempen, 1997). Limited studies of crustaceans have examined mortality rates at various distances from detonations in shallow water (Aplin, 1947; Chesapeake Biological Laboratory, 1948; Gaspin et al., 1976). Similar studies of molluscs have shown them to be more resistant than crustaceans to explosive effects (Chesapeake Biological Laboratory, 1948; Gaspin et al., 1976). Other invertebrates, such as sea anemones, polychaete worms, isopods, and amphipods, were observed to be undamaged in areas near detonations (Gaspin et al., 1976). Data from these experiments were used to develop curves that estimate the distance from an explosion beyond which at least 90 percent of certain adult benthic marine invertebrates would survive, depending on the weight of the explosive (Young, 1991) (Figure F-3). For example, 90 percent of crabs would survive a 200-lb. explosion if they are greater than about 350 ft. from the source, and shrimp, lobster, and oysters are less sensitive (i.e., greater survivability) to underwater explosions than crabs. Similar information on the effects of explosions to planktonic invertebrates and invertebrate larvae is not available.




Source: Young (1991)

**Figure F-3: Prediction of Distance to 90 Percent Survivability of Marine Invertebrates Exposed to an Underwater Explosion**

Charges detonated in shallow water or near the bottom, including explosive munitions disposal charges and some explosions associated with mine warfare, could kill and injure marine invertebrates on or near the bottom, depending on the species and the distance from the explosion. Taxonomic groups typically associated with the bottom, such as sponges, marine worms, crustaceans, echinoderms, corals, and molluscs, could be affected. Net explosive weight (NEW) for activities involving detonations on or near the bottom is relatively low. Most detonations occurring on or near the bottom would have a NEW of 60 lb. or less, although some explosives would be up to 1,000 lb. NEW. Based on the estimates shown on Figure 3.4-1, most benthic marine invertebrates beyond approximately 275 ft. from a 60 lb. blast would survive. The potential mortality zone for some taxa (e.g., shrimp, lobsters, worms, amphipods) would be substantially smaller. A blast near the bottom could disturb sessile invertebrates such as mussels and hard substrate suitable for their colonization. A blast in the vicinity of hard corals could cause direct effects on coral polyps or early life-stages of pre-settlement corals, or fragmentation and siltation of the corals. For example, in one study, moderate to substantial recovery from a single small blast directly on a reef was observed within five years, but reef areas damaged by multiple blasts showed no evidence of recovery during the six-year observation period (Fox & Caldwell, 2006). In another study, modeling results indicated that deep-water corals off Alaska damaged by trawling activities could require over 30 years to recover 80 percent of the original biomass (Rooper et al., 2011). The extent of trawling damage is potentially greater than that associated with detonations due to the small footprints of detonations compared to the larger surface area typically affected by trawling, as well as the avoidance of known shallow-water coral reefs and live hard bottom habitat during activities involving detonations. While the effects of trawling activities and underwater detonations are not directly comparable, the trawling model results illustrate the extended recovery time that may be required for deep-water coral regrowth following physical disturbance.

Effects on benthic invertebrates in deeper water would be infrequent because most offshore detonations occur in the air or at the surface. Benthic invertebrates in the abyssal zone (generally considered to be deeper than about 6,000 ft.) seaward of the coastal large marine ecosystems are sparsely distributed and tend to be concentrated around hydrothermal vents and cold seeps. These topographic features are typically associated with steep or high-relief areas of the continental shelf break (e.g., canyons, outcrops) or open ocean (e.g., seamounts).

Underwater surveys of a Navy bombing range in the Pacific Ocean (Farallon De Medinilla) were conducted annually from 1999 to 2012 (Smith & Marx, 2016). Although Farallon De Medinilla is a land range, bombs and other munitions occasionally strike the water. A limited number of observations of explosion-related effects were reported, and the results are summarized here to provide general information on the types of effects that may occur. However, the effects are not presumed to be broadly applicable to Navy training and testing activities. During the 2010 survey, it was determined that a blast of unknown size (and therefore of unknown applicability to proposed training and testing activities) along the waterline of a cliff ledge caused mortality to small oysters near the impact point. Corals occurring within 3 m of the affected substrate were apparently healthy. A blast crater on the bottom that was 5 m in diameter and 50 cm deep, presumably resulting from a surface detonation, was observed during one survey in water depth of 12 m. Although it may be presumed that corals or other invertebrates located within the crater footprint would have been damaged or displaced, evidence of such effects was not detected. The blast occurred in an area of sparse coral coverage and it is therefore unknown whether coral was present in the crater area prior to the blast.

The applicability of the mortality distance estimates shown on Figure 3.4-1 in Section 3.4 to invertebrates located in the water column is unknown. However, detonations that occur near the surface release a portion of the explosive energy into the air rather than the water, reducing effects on invertebrates in the water column. In addition to effects caused by a shock wave, organisms in an area of cavitation that forms near the surface above a large underwater detonation could be killed or injured. Cavitation is where the reflected shock wave creates a region of negative pressure followed by a collapse, or water hammer (see Appendix D, Acoustic and Explosive Concepts Supporting Information). The number of organisms affected by explosions at the surface or in the water column would depend on the size of the explosive, the distance of organisms from the explosion, and the specific geographic location within the Study Area. As discussed previously, many invertebrates that occur near the surface at night (e.g., squid and zooplankton) typically move down in the water column during the day, making them less vulnerable to explosions when most Navy activities involving detonations occur.

Marine invertebrates beyond the range of mortality or injurious effects may detect the impulsive sound produced by an explosion. At some distance, impulses lose their high pressure peak and take on characteristics of non-impulsive acoustic waves. Invertebrates that detect impulsive or non-impulsive sounds may experience stress or exhibit behavioral reactions in response to the sound. Repetitive impulses during multiple explosions, such as during a surface firing exercise, may be more likely to cause avoidance reactions. However, the distance to which invertebrates are likely to detect sounds is limited due to their sensitivity to water particle motion caused by nearby low-frequency sources. Sounds produced in water during training and testing activities, including activities that involve multiple impulses, occur over a limited duration. Any auditory masking, in which the sound of an impulse could prevent detection of other biologically relevant sounds, would be very brief.

### F.4.3 Energy Stressors

#### F.4.3.1 Effects from In-Water Electromagnetic Devices

Several different types of electromagnetic devices are used during training and testing activities. Information on the types of activities that use in-water electromagnetic devices is provided in Appendix B (Activity Stressor Matrices).

Little information is available regarding marine invertebrates' susceptibility to electromagnetic fields. Magnetic fields are not known to control spawning or larval settlement in any invertebrate species. Existing information suggests sensitivity to electric and magnetic fields in at least three marine invertebrate phyla: Mollusca, Arthropoda, and Echinodermata (Bureau of Ocean Energy Management, 2011; Lohmann & Lohmann, 2006; Lohmann et al., 1995). A possible magnetic sense has been suggested in jellyfish as well, although this has not been demonstrated experimentally (Fossette et al., 2015). Much of the available information on magnetic field sensitivity of marine invertebrates pertains to crustaceans. For example, a magnetic compass sense has been demonstrated in the spiny lobster (*Panulirus argus*) (Lohmann & Lohmann, 2006; Lohmann et al., 1995), and researchers suggest subtle behavioral response to magnetic fields of about 1 millitesla (1,000 microtesla) in the Dungeness crab and American lobster (Woodruff et al., 2013). A review of potential effects of undersea power cables on marine species provides a summary of numerous studies of the sensitivity of various invertebrate species to electric and magnetic fields (Bureau of Ocean Energy Management, 2011). Electric field sensitivity is reported in the summary for only two freshwater crayfish species, while magnetic field sensitivity is reported for multiple marine invertebrate species, including molluscs, crustaceans, and echinoderms. Sensitivity thresholds range from 300 to 30,000 microtesla, depending on the species. Most responses consisted of behavioral changes, although non-lethal physiological effects were noted in two sea urchin species in a 30,000 microtesla field (embryo development) and a marine mussel exposed to 300 to 700 microtesla field strength (cellular processes). Marine invertebrate community structure was not affected by placement of energized underwater power cables with field strengths of 73 to 100 microtesla (Love et al., 2016). Effects to eggs of the sea urchin *Paracentrotus lividus* and to brine shrimp (*Artemia* spp.) cysts have been reported at relatively high magnetic field strengths (750 to 25,000 microtesla) (Ravera et al., 2006; Shckorbatov et al., 2010). The magnetic field generated by the Organic Airborne and Surface Influence Sweep (a typical electromagnetic device used in Navy training and testing) is about 2,300 microtesla at the source. Field strength drops quickly with distance from the source, decreasing to 50 microtesla at 4 m, 5 microtesla at 24 m, and 0.2 microtesla at 200 m from the source. Therefore, temporary disruption of navigation and directional orientation is the primary effect considered in association with magnetic fields.

Studies of the effects of low-voltage direct electrical currents in proximity to marine invertebrates suggest a beneficial effect on at least some species at appropriate current strength. American oysters (*Crassostrea virginica*) and various stony and soft corals occurring on substrates exposed to low-voltage currents (between approximately 10 and 1,000 microamperes) showed increased growth rates and survival (Arifin et al., 2012; Goreau, 2014; Jompa et al., 2012; Shorr et al., 2012). It is theorized that the benefits may result from a combination of more efficient uptake of calcium and other structure-building minerals from the surrounding seawater, increased cellular energy production, and increased pH near the electrical currents. The beneficial effects were noted in a specific range of current strength; higher or lower currents resulted in either no observable effects or adverse effects. The moderate voltage and current associated with the Organic Airborne and Surface Influence Sweep are not expected to result in adverse effects to invertebrates. In addition, due to the short-term, transient nature of electromagnetic device use, there would be no beneficial effects associated with small induced electrical currents in structures colonized by invertebrates.

### F.4.4 Physical Disturbance and Strike Stressors

Most marine invertebrate populations extend across wide areas containing hundreds or thousands of discrete patches of suitable habitat. Sessile invertebrate populations may be connected by complex currents that carry adults and young from place to place. Effects on such widespread populations are difficult to quantitatively evaluate in terms of Navy training and testing activities that occur intermittently and in relatively small patches in the Study Area. Invertebrate habitats generally cover enormous areas (Section 3.2, Sediments and Water Quality) and, in this context, a physical strike or disturbance would affect individual organisms directly or indirectly, but not to the extent that viability of populations of common species would be affected. While the potential for overlap between Navy activities and invertebrates is reduced for those species living in rare habitats, if overlap does occur, any potential effects would be amplified for those invertebrate species or taxa with limited spatial extent. Examples of such organisms include abalones, stony corals, and sponges, which are mostly restricted to hard bottom habitat or artificial habitat. Shallow-water coral reefs, precious coral beds, live hard bottom, and other areas of hard substrate such as artificial reefs are protected to the extent they are included in current mitigation measures.

With few exceptions, activities involving vessels and in-water devices are not intended to contact the bottom due to potential damage to equipment and the resulting safety risks for vessel personnel. The potential for strike impact and disturbance of benthic or habitat-forming marine invertebrates would result from amphibious activities, bottom-crawling unmanned underwater vehicles, military expended materials, seafloor devices, and pile driving. For environmental and safety reasons, amphibious landings and other nearshore activities would avoid areas where corals are known to occur.

With the exception of habitat-forming benthic taxa (e.g., corals, sea pens, and sponges), most small invertebrate populations recover quickly from non-extractive disturbance. Many large invertebrates, such as crabs, shrimps, and clams, undergo massive disturbance during commercial and recreational harvests, storms, or beach restoration activities. Invertebrates that occur in the high-energy surf zone are typically resilient to dynamic processes of sediment erosion and accretion, although some community effects may occur due to rapid and relatively large-scale changes such as those associated with beach renourishment projects (U.S. Army Corps of Engineers, 2001).

Biogenic habitats such as shallow coral reefs, deep-water coral, and sponge communities may take decades to regrow following a strike or disturbance (Jennings & Kaiser, 1998; Precht et al., 2001). However, bottom-disturbing activities are not conducted on mapped coral reefs or live hard bottom. In soft bottom areas, recovery of benthic invertebrate populations after substantial human disturbance depends on factors such as size of the area disturbed, bottom topography, hydrodynamics of the affected area, seasonality of the disturbance, and the size and typical growth rate of affected species. Most studies of the effects of beach sand nourishment projects (which is a proxy for effects due to amphibious landings) have reported initial declines in benthic invertebrate populations due to burial and increased turbidity (which may affect filter-feeding capability), but subsequent recovery over time scales of weeks to years (Posey & Alphin, 2002; U.S. Army Corps of Engineers, 2001, 2012; Wilber et al., 2009). Recovery is typically greatest at nourishment sites when there is a close match in grain size between the existing and supplied sediment. However, species composition may be altered in the recolonized area, and overall invertebrate biomass may not recover for many years. Researchers found that trawling off the California coast resulted in no statistical difference in the abundance of sessile or mobile benthic invertebrates (Lindholm et al., 2013). However, repeated and intense bottom fishing disturbance can result in a shift from communities dominated by relatively high-biomass individuals towards dominance by high abundance of small-sized organisms (Kaiser et al., 2002). If activities are repeated at the same site, the benthic invertebrate community composition could be altered over time (years), especially for sessile invertebrates (e.g., coral). Some bottom-disturbing activities, such as mine countermeasures and neutralization training and testing, precision anchoring, and pile driving associated with the Port damage repair activity, may occur in the same locations or near the same locations yearly.

#### F.4.4.1 Effects from Vessels and In-Water Devices

### Vessels

The majority of the training and testing activities under all the alternatives involve vessels. For a discussion of the types of activities that use vessels and where they are used, refer to Appendix B (Activity Stressor Matrices). See Table 3.0-14 for a representative list of Navy vessel types, lengths, and speeds.

Vessels could affect adults and other life stages of marine invertebrates by directly striking organisms, or by disturbing the water column or sediments (Bishop, 2008). Species that occur at or near the surface (e.g., jellyfish, squid) would potentially be exposed to direct vessel strikes. Exposure to propeller-generated turbulence was found to result in mortality in a zooplankton species (the copepod *Acartia tonsa*) located near the surface (Bickel et al., 2011). However, many pelagic invertebrates such as squid and zooplankton move away from the surface during the day, reducing potential exposures during daytime vessel operations. Many vessel hulls have a hydrodynamic shape, and pelagic marine invertebrates are therefore generally disturbed, rather than struck, as the water flows around a vessel. Zooplankton are ubiquitous in the water column and typically experience high mortality rates.

In addition, vessel hull strikes and propeller cavitation and turbulence could displace, damage, injure, or kill invertebrate eggs and larvae in the upper portion of the water column throughout the Study Area. For example, turbulent water was found to decrease successful fertilization and resulted in abnormal development and low survival in eggs of the broadcast spawning purple sea urchin (*Strongylocentrotus purpuratus*) (Mead & Denny, 1995). In some areas of the Hawaii Study Area, vessels could transit through water containing coral gametes, eggs, embryonic stages, or planula larvae of broadcast spawning species. Eggs of cluster coral (*Acropora millepora*) were found to disintegrate into irregular groups or individual blastomeres when subjected to even very light shearing forces and turbulence (Heyward & Negri, 2012). Such dissociation can be beneficial through creation of more juveniles, but may also cause mortality. Early embryonic development of broadcast spawning coral species has reportedly been affected by handling of captive-reared embryos (Guest et al., 2010). Although the available information indicates that developmental stages of numerous invertebrate species could be physically affected, broadcast-spawning invertebrates produce very large numbers of eggs and planktonic larvae that typically experience high mortality rates under normal conditions (Nybakken, 1993). Any effects resulting from Navy vessel operation would be biologically insignificant by comparison.

Propeller wash (water displaced by propellers used for propulsion) of even the deepest draft vessels operated over the continental shelf is likely indistinguishable from the water motion associated with periodic storm events, and vessel operation in deeper waters beyond the shelf break would not affect the bottom. Therefore, the potential for vessels to disturb invertebrates on or near the bottom would occur mostly during nearshore and inshore training or testing activities, and along dredged navigation channels. Invertebrates on or near the bottom in such relatively shallow areas could be affected by sediment disturbance or direct strike during amphibious landings. Few sources of information are available on the effect of non-lethal chronic vessel disturbance to marine invertebrates. One study of seagrass-associated marine invertebrates, such as amphipods and polychaetes, found that chronic disturbance from vessel wakes resulted in the long-term displacement of some marine invertebrates from the affected shallow-water area (Bishop, 2008). However, invertebrates that typically occur in areas associated with nearshore or inshore activities, such as shorelines, are highly resilient to vessel disturbance. They are regularly disturbed by natural processes such as high-energy waves and longshore currents, and generally recover quickly. Potential exceptions include sessile or encrusting invertebrates that may occur along sheltered shorelines that are subject to a high frequency of boat propeller- or wake-induced erosion (Grizzle et al., 2002; Zabawa & Ostrom, 1980). Increased erosion of shoreline banks or suspension of bottom sediments may cause turbidity that affects filter-feeding invertebrates. The results of a small number of studies suggest that the wave energy resulting from boat wakes produced in relatively narrow water bodies may affect oyster occurrence, and studies of shallow freshwater areas found that waves generated from small boats caused about 10 percent of benthic invertebrates (e.g., amphipods) to become suspended in the water column where they presumably would be more vulnerable to predation (Bilkovic et al., 2017).

Non-amphibious vessels avoid contact with the bottom in order to prevent damage to the vessels and benthic habitat that supports encrusting organisms. The encrusting organisms (e.g., hard corals) living on hard substrate in the ocean are exposed to strong currents under natural conditions and would not likely be affected by propeller wash. Many activities occur in offshore areas and, therefore, would be unlikely to affect benthic invertebrates, although small-caliber gunnery exercises, blank firing, and smoke grenade use may occur in areas closer to shore. Many Navy vessel movements in nearshore waters are concentrated in established channels and ports or predictable transit corridors between the Hawaiian Islands or between San Diego Bay and SCI, and shallow-water vessels typically operate in defined boat lanes with sufficient depths to avoid propeller or hull strikes on the bottom.

The only source of shallow-water vessel movement in the Study Area with known direct effects on benthic invertebrates is amphibious landings, which are conducted in the Hawaii Study Area and California Study Area (Appendix A, Navy Activity Descriptions). Amphibious vessels would contact the bottom in the surf zone during amphibious assault and amphibious raid operations. Benthic invertebrates of the surf zone, such as crabs, clams, and polychaete worms, within the disturbed area could be displaced, injured, or killed during amphibious operations. Burrowing species such as ghost shrimp are present on many beaches, and individuals in relatively shallow burrows located just above harder sand layers could be injured or killed if amphibious vessels compress the sand above them. Passage of amphibious vessels could cause some elevated turbidity in the nearshore zone seaward of the surf zone. However, the sediment along landing beaches is constantly being reworked by nearshore wave energy and, to a lesser extent (although more frequently than disturbance caused by amphibious landings), storm events. Benthic invertebrates inhabiting these areas are adapted to a naturally disturbed environment and are expected to rapidly re-colonize similarly disturbed areas by immigration and larval recruitment. Studies indicate that benthic communities of high-energy sandy beaches recover relatively quickly (typically within two to seven months) following beach nourishment. Researchers found that the macrobenthic (visible organisms on the bottom) community required between 7 and 16 days to recover following excavation and removal of sand from a 200 square meter quadrant from the intertidal zone of a sandy beach (Schoeman et al., 2000). The number of invertebrates affected during amphibious landings would be small compared to the number that are affected during activities such as beach nourishment. The effects of amphibious vehicle operations on benthic communities would therefore likely be minor, short term, and local.

Other than organisms occurring at amphibious landing sites, invertebrates that occur on the bottom, including shallow-water corals, organisms associated with hard bottom, and deep-water corals, are not likely to be exposed to vessel strikes. Propeller movement has the potential to disrupt sediments that could affect shallow-water corals and hard bottom communities. However, shallow-water corals and abalone species do not occur along the shoreline adjacent to amphibious landing areas.

**In-Water Devices**

Some of the training and testing activities under both action alternatives involve the use of in-water devices, including remotely operated vehicles, unmanned surface vehicles, unmanned underwater vehicles, motorized autonomous targets, and towed devices. For a discussion of the types of activities that use in-water devices, see Appendix B (Activity Stressor Matrices). See Table 3.0-16 in Section 3.0 for the types, sizes, and speeds of representative Navy in-water devices used in the Study Area.

In-water devices can operate from the water's surface to the benthic zone. The devices could potentially affect marine invertebrates by directly striking organisms or by disturbing the water column. As discussed for vessel use, most invertebrates in the water column would be disturbed, rather than struck, as water flows around a device due to the hydrodynamic shape. In addition, in-water devices are smaller than most Navy vessels, decreasing the surface area in which invertebrates could be struck. The potential for direct strike is reduced for some types of devices because they are operated at relatively low speeds (e.g., unmanned underwater vehicles, which are typically operated at speeds of 1 to 15 knots). Unmanned surface vehicles are operated at the greatest speeds (up to 50 knots or more) and therefore have greater potential to strike invertebrates. However, relatively few invertebrates occur at the surface and consist mostly of squid, jellyfish, and zooplankton. Squid and many zooplankton species move away from the surface during the day (Nybakken, 1993), when unmanned surface vehicles are typically operated. In-water devices do not normally collide with invertebrates on the bottom because the devices are operated in relatively deep water and contact with the bottom is avoided. Devices operated very near the bottom could potentially disturb sediments and associated invertebrates through propeller wash. However, such disturbance would be infrequent and would affect a small area, and disturbed areas would be quickly reoccupied by benthic invertebrates.

As discussed for vessels, zooplankton and invertebrate eggs and larva could be displaced, damaged, injured, or killed by propeller wash or turbulence resulting from water flow around in-water devices. Effects due to turbulence would generally increase with increasing speed of the device. Many zooplankton species migrate away from the surface during the day, when Navy training and testing typically are conducted, decreasing the potential for effects in the upper portions of the water column. The number of individuals affected would be small in comparison to overall populations, and the affected species generally exhibit rapid growth and recovery rates.

#### F.4.4.2 Effects from Military Expended Materials

Military expended materials are deposited throughout the Study Area. However, the majority of military expended materials are deposited within established range complexes and testing ranges. These areas of higher military expended materials deposition are generally located away from the coastline on the continental shelf and slope and beyond (e.g., abyssal plain). Physical disturbance or strikes by military expended materials on marine invertebrates is possible at the water's surface, through the water column, and on the bottom. However, disturbance or strike effects on marine invertebrates by military expended materials falling through the water column are not very likely because military expended materials do not generally sink rapidly enough to cause strike injury. Exposed invertebrates would likely experience only temporary displacement as the object passes by. Therefore, the discussion of military expended materials disturbance and strikes will focus on items at the water's surface and on the bottom.

Potential effects on invertebrates generally consist of physical trauma, stress or behavioral responses, abrasion, and shading. Military expended materials may injure or kill invertebrates by directly striking individuals, causing breakage (particularly for species with exoskeletons or that build structures), crushing, or other physical trauma. Direct strike may result from the initial impact, or may occur after items fall through the water column and settle onto invertebrates or are moved along the bottom by water currents or gravity. Expended items may also bury or smother organisms although, depending on the size of the expended item relative to the animal, some mobile invertebrates may be able to move or dig out from underneath an item. In addition to physical strike, military expended materials may disturb individuals and cause them to change locations, behaviors, or activities. Disturbance could therefore result in effects such as briefly increased energy expenditure, decreased feeding, and increased susceptibility to predation. Expended items could also cause increased turbidity that could affect filter-feeding species, although such effects are likely to be localized and temporary. Expended items that come to rest on or near corals could cause abrasion or shading (in the case of corals that host symbiotic algae) that reduces photosynthesis in the algae, although these effects are unlikely based on the mitigation measures in place for shallow-water coral reefs where symbiotic algae are present. Abrasion refers to scraping or wearing down of a supporting structure or hard body part (e.g., coral skeleton, shell) through repeated impact on the same individual or structure. Abrasion would generally be associated with military expended materials such as flexible materials (e.g., wires or cords) that become fixed in a location for some time but that are moved repeatedly over sessile invertebrates by water currents.

Military expended materials that impact the water surface could directly strike zooplankton, the gametes, embryos, and larvae of various invertebrate species (including ESA-listed abalone species), and a small number of adult invertebrates (e.g., squid, jellyfish, swimming crabs). However, many zooplankton and squid are absent from the surface water column during the day when most training and testing activities occur. Inert military expended materials also have the potential to impact the water and produce a large impulse which could disturb nearby invertebrates. Potential effects on invertebrates resulting from impulsive sound and shock waves are discussed in Section 3.4.3.1 (Acoustic Stressors) and Section 3.4.3.2 (Explosive Stressors). In addition to direct strike of invertebrates and production of impulsive sound, surface water impacts could affect physical properties of the surrounding water (e.g., slight heating or increased dissolved gas concentrations due to turbulent mixing with the atmosphere), potentially affecting the suitability of the affected water mass as habitat for some invertebrate species. However, physical changes to the water column would be localized and temporary, persisting for only a few minutes.

Compared to surface waters and offshore areas, a greater number of macroinvertebrates typically occurs on the bottom and closer to shore. Benthic invertebrate taxa, including sponges, cnidarians, worms, bryozoans, molluscs, arthropods, and echinoderms, may occur in areas affected by military expended materials. However, some of the most sensitive benthic species (e.g., corals) are more likely to occur on hard bottom, reefs, and other hard substrates. Shallow-water coral reefs are protected by mitigation measures from most activities that generate military expended materials. Military expended materials that impact the bottom may affect invertebrates by strike (including injury or mortality), disturbance, burial, abrasion, or shading within the footprint of the item (the area of substrate physically covered by the item). Military expended materials may also cause physiological or behavioral reactions to individual invertebrates outside the footprint of the items. After items come to rest on the bottom, continued impacts are possible if the items are mobilized by currents or waves and damage benthic invertebrates as they move. Turbidity may also occur as water flows around deposited items. However, these impacts would generally cease when the military expended materials are incorporated into the seafloor by natural encrustation or burial processes, or become otherwise immobilized.

Sessile marine invertebrates and infauna (organisms attached to the bottom or living in the sediments) are generally more susceptible to military expended material disturbance and strike than benthic species with the ability to move relatively quickly over the bottom. Some susceptible species (e.g., hydroids, sponges, soft corals) have fragile structures and sensitive body parts that could be damaged or covered by military expended materials. Military expended materials could also break hard structures such as coral skeletons and mussel beds. Shallow- and deep-water corals that build complex or fragile structures could be particularly susceptible to breakage or abrasion. Such structures are resistant to physical forces typical of ambient conditions (e.g., water currents), but not as resilient to other types of physical disturbance involving greater force. Decelerators/parachutes would be unlikely to be carried by currents onto reef structures due to the typical offshore locations of use and the sink rate of the items. Expended items may provide new colonization sites for benthic invertebrates. Researchers found that military expended materials in a bombing range became covered by sedentary reef invertebrates over time (Smith & Marx, 2016). However, invertebrate species composition on artificial substrates may differ from that of the surrounding natural community.

Potential effects on shallow-water corals, invertebrates associated with hard bottom habitat, or deep-water corals present the greatest risk of long-term damage compared with other bottom communities because: (1) many corals and hard bottom invertebrates are sessile, fragile, and particularly vulnerable; (2) many of these organisms grow slowly and could require decades to recover; and (3) military expended materials are likely to remain exposed on hard bottom communities whereas shifting sediment patterns would tend to bury military expended materials in soft bottom communities. The probability of striking deep-water corals or invertebrates located on hard bottom habitat is low, given their low percent cover on suitable habitat (see Section 3.5.2.2, Bottom Habitats, for a discussion of hard bottom habitat).

A few investigations have been conducted to determine the presence and, in some cases, possible effects of military expended materials on the bottom. The results of multi-year underwater surveys at a military bombing range in the Mariana Archipelago (Pacific Ocean) provide an example of potential effects resulting from expended munitions. Water areas were not targeted at this range; bottom effects occurred only when the target land mass was missed or the munition bounced off the land into the water. The surveys found no overall long-term adverse effects on corals or other invertebrates due to expended items, despite several decades of use (Smith & Marx, 2016). Numerous intact bombs and fragments were observed on the bottom. Inert 500 lb. bombs were found to disturb a bottom area of 17 square meter each, although specific damage to invertebrates, if any, was not described. It may be presumed that invertebrates within this footprint could have been killed, injured, damaged, or displaced. Expended items, once settled in place, appeared to become encrusted with marine growth and pose no substantial long-term threat to invertebrates. The condition of corals indicated a healthy environment, with no apparent change in species composition, distribution, size, or stress indicators. However, the results of several other studies indicate that sessile invertebrate communities growing on artificial substrate such as the expended munitions are often different than those growing on natural substrate (Burt et al., 2009; Macreadie et al., 2011; Perkol-Finkel et al., 2006; Steimle & Zetlin, 2000). A remotely operated vehicle survey of deep portions of the Jacksonville Range Complex reported only two exposed items of military expended materials in about 37,800 m of survey line distance (U.S. Department of the Navy, 2010a, 2011). However, it is important to note that the survey was not designed to document military expended materials and these were only the items photographed using still frames. Another extensive remotely operated vehicle survey along the continental shelf break and canyons in the northeast and mid-Atlantic region found marine debris in 81 percent of individual dives, but the items did not include any visible military expended materials (Quattrini et al., 2015). Underwater surveys of bottom areas off the Gulf coast of Florida with a presumably high potential for military expended materials (based on reported obstructions by fishermen) found no items of military origin, suggesting that expended materials may be widely distributed or may become covered by sediments (U.S. Department of the Navy, 2013c). In a deep-sea trawl survey of the northern Gulf of Mexico, items of military origin were found (artillery shells and a missile), but were among the least-frequently encountered types of debris (Wei et al., 2012).

**Military Expended Materials - Munitions**

Military expended materials that are munitions and associated with training activities include small-, medium-, and large-caliber projectiles, bombs, missiles, rockets, and grenades. Fragments of exploded munitions are also included because they can result in effects on invertebrates that are similar to those associated with smaller intact munitions. Military expended materials associated with testing activities are the same except that there are no grenades. Navy training and testing activities in the Study Area include firing a variety of weapons and using a variety of non-explosive training and testing rounds, including small-, medium-, and large-caliber projectiles. Large-caliber projectiles are primarily used in the open ocean beyond 20 NM from shore. Direct strike from bombs, missiles, and rockets would result in types of effects similar to those of projectiles. However, they are larger than most projectiles and are likely to produce a greater number of fragments. Bombs, missiles, and rockets are designed to explode within about 3 ft. of the sea surface, where marine invertebrates larger than zooplankton are relatively infrequent.

**Military Expended Materials Other Than Munitions**

Military expended materials other than munitions associated with training and testing activities include a large number of items such as aerial countermeasures, targets (surface and aerial), mine shapes, ship hulk, decelerators/parachutes, acoustic countermeasures, sonobuoys, and other materials such as torpedo accessories, concrete slugs, marine markers, bathythermographs, endcaps, and pistons. Some expended materials used during training and testing activities, including some types of torpedoes and targets, non-explosive mine shapes, and bottom-placed instruments, are recovered.

Chaff, which consists of aluminum-coated glass fibers, may be transported great distances by the wind, beyond the areas where they are deployed, before contacting the sea surface. These materials contact the sea surface and bottom with very little kinetic energy, and their low buoyant weight makes them an inconsequential strike and abrasion risk. Therefore, chaff is not considered to be a potential strike and disturbance stressor.

During a sinking exercise, aircraft, ship, and submarine crews deliver munitions on a surface target, which is a clean, deactivated ship that is deliberately sunk using multiple weapon systems. Sinking exercises occur in specific open ocean areas, outside of the coastal range complexes. Habitat-forming invertebrates are likely absent where sinking exercises are planned because the activity occurs in depths greater than the range for shallow-water and many deep-water coral species (approximately 3,000 m) and away from typical locations for hydrothermal vent or cold seep communities (e.g., seamounts) (Cairns, 2007). It is unlikely that deep-sea hard corals would be affected by a sinking ship hulk or fragments of a hulk due to their lack of occurrence below depths of about 3,000 m (the depth of the aragonite saturation boundary; see Appendix C, Section 3.1.1, Habitat Use).

Decelerators/parachutes of varying sizes are used during training and testing activities and may be deployed from aircraft or vessels. Similar to other marine debris such as derelict fishing gear, decelerators/parachutes may kill or injure sessile benthic invertebrates due to covering/shading or abrasion. Activities that expend sonobuoy and air-launched torpedo decelerators/parachutes generally occur in relatively deep water away from the shore; however, there is some potential for near shore effects from testing events using sonobuoys to conduct mine detection in near shore environments. Decelerators/parachutes expended over deep offshore areas may affect deep-water invertebrates (particularly sessile species) by disturbance, strikes, burial, smothering, or abrasion. For example, a decelerator/parachute could cover a sponge or deep-water coral and impair feeding.

Proportional affect analysis determined that the total bottom area affected by all military expended materials in all training areas would be about 145 acres annually, ranging from less than 1 acre to about 120.5 acres in specific range complexes and substrate types. This represents much less than 1 percent of available bottom habitat in any range complex. In addition to expended items, recovered materials would temporarily disturb approximately 10 acres of bottom habitat in all training areas combined. The substrate types and associated invertebrate assemblages within the potentially disturbed areas are difficult to predict, as discussed in Appendix I (Military Expended Materials and Direct Strike Impact Analyses). Activities conducted throughout the Study Area have the potential to affect hard bottom communities as well as invertebrates within all other habitat types. Activities occurring at depths of less than about 3,000 m may affect deep-water corals. Consequences could include damage, injury, or mortality as a result of projectiles, munitions, or other items. Decelerators/parachutes, wires, and cables could also affect benthic communities if they are mobilized by water currents, although it is expected that most such materials would become buried, encrusted, or otherwise immobilized over time and would not continue to affect individual invertebrates or invertebrate assemblages. Effects would be most pronounced if all the materials expended within the applicable depth range were deposited on areas of hard substrate supporting long-lived, sessile organisms such as deep-water corals, because it may be assumed that many of the benthic invertebrates present in the impact area footprint would be killed, injured, displaced, or disturbed by the expended materials. In addition, some previously undisturbed bottom area would be affected by activities in subsequent years. Conversely, effects would be less if the materials were deposited on soft bottom areas containing invertebrate communities that recover relatively quickly from disturbance. Although hard substrate potentially supporting deep-water corals and other invertebrate communities is present in at least some areas in water depths less than 3,000 m, a scenario of all expended materials being deposited on such substrate is unrealistic. Deep-water stony corals are relatively rare in the Hawaiian Archipelago region, and most species are solitary. Hard and mixed bottom types, which support the occurrence of deep-water corals other than sea pens, are relatively rare off the U.S. west coast, accounting for about 10 percent of the substrate from the shelf to depths of 3,000 m (Clarke et al., 2015). These habitat types are often associated with seamounts, banks, and canyons (particularly banks in the Channel Islands region). Based on the results of limited investigation, a low percentage of available hard substrate may be inhabited by deep-water corals or other invertebrate species (Harter et al., 2009; U.S. Department of the Navy, 2010a). It is expected that most of the bottom type affected would be soft substrate (Appendix I, Military Expended Materials and Direct Strike Impact Analyses). Therefore, although it is possible for a portion of expended items to affect hard substrate and associated sensitive invertebrate communities, the number of exposed individuals would not likely affect the overall viability of populations or species. While the potential for overlap between Navy activities and invertebrates is reduced for those species living in rare habitats, if overlap does occur, any potential effects would be amplified for those invertebrate species or taxa with limited spatial extent. With the exception of abalones and some shallow-water corals, detailed distribution and habitat utilization information sufficient to support species-specific analysis is generally unavailable.

#### F.4.4.3 Effects from Seafloor Devices

For a discussion of the types of activities that use seafloor devices, where they are used, and how many activities would occur under each alternative, see Appendix B (Activity Stressor Matrices). Seafloor devices include items that are placed on, dropped on, or moved along the substrate for a specific purpose, and include mine shapes, anchor blocks, anchors, bottom-placed instruments, bottom-crawling unmanned underwater vehicles, and bottom placed targets that are recovered (not expended). Placement or deployment of seafloor devices would cause disturbance, injury, or mortality to marine invertebrates within the footprint of the device. These items could potentially break hard substrate and associated biogenic habitats (e.g., hard coral skeletons). Objects placed on the bottom may attract invertebrates, or provide temporary attachment points for invertebrates. Some invertebrates attached to the devices would be removed from the water when the devices are recovered. A shallow depression may remain for some time in the soft bottom sediment where an anchor was dropped, potentially altering the suitability of the affected substrate for benthic invertebrates temporarily (possibly months).

Seafloor devices may also disturb marine invertebrates outside the footprint of the device, and would cause temporary (possibly hours to days) local increases in turbidity and sedimentation near the bottom, along with some changes in scouring/deposition patterns in higher current areas with soft bottom. Sedimentation can smother sessile invertebrates, while turbidity may affect respiratory organs or impair the ability of filter-feeding invertebrates to obtain food (e.g., by clogging their feeding structures or diluting the amount of food in the surrounding volume of water). However, the brief episodes of minor turbidity associated with Navy seafloor devices would be localized and the effects do not change the substrate type. Compared to overall populations, relatively few individuals would be affected.

Precision anchoring, and the associated potential effects, is qualitatively different than other seafloor devices because the activity involves repeated disturbance to the same soft bottom areas. Precision anchoring may result in temporary and localized disturbances to water column and bottom habitats. For example, an anchor may shift due to changing currents or vessel movement and the mooring chain may drag across the bottom, causing abrasion and effects on benthic species (Davis et al., 2016). Anchor impacts on the bottom would likely crush a small number of benthic invertebrates. Bottom disturbance would result in localized sedimentation and turbidity, which could smother invertebrates or affect respiration or feeding. Turbidity would quickly dissipate (i.e., minutes to hours) following the exercise, and many soft bottom invertebrates are burrowing organisms that would be unaffected by shallow burial. Although precision anchoring occurs in soft-bottom areas, where invertebrate populations are generally resilient to disturbance, invertebrates in designated anchorage areas may be prevented from fully recovering due to long-term use, and benthic composition may be changed compared to historical conditions.

### F.4.5 Entanglement Stressors

This section analyzes the potential entanglement effects of the various types of expended materials used by the Navy during training and testing activities within the Study Area. Included are potential effects from wires and cables, and decelerators/parachutes. In this section, only potential effects of these items as entanglement stressors are discussed. Abrasion and covering/shading effects on sessile benthic invertebrates are discussed with physical effects in Section 3.4.3.3.3.2.1 (Effects from Military Expended Materials).

Marine invertebrates are likely less susceptible than vertebrates to entanglement, as illustrated by the fact that fishing nets which are designed to take pelagic marine invertebrates operate by enclosing or entrapping rather than entangling (Chuenpagdee et al., 2003). However, entanglement may be possible for some species and some expended items. A survey of marine debris entanglements found that marine invertebrates accounted for 16 percent of all animal entanglements (Ocean Conservancy, 2010). The same survey cites potential entanglement in military items only in the context of waste-handling aboard ships, and not for military expended materials. A summary of the effects of litter on various marine species identified potential effects on some invertebrate taxa, particularly mobile benthic species such as crabs and sea stars, that may become entangled in debris (e.g., nets) after attempting to move through the items (National Oceanic and Atmospheric Administration Marine Debris Program, 2014a). The potential for a marine invertebrate to become entangled in wires, cables, or decelerators/parachutes is considered remote. The materials generally do not have the characteristics required to entangle marine species. Wires and cables are essentially rigid lines. Sonobuoy components may include plastic mesh and a float unit. Although mesh items have increased potential for entangling marine animals in general, and invertebrates can become entangled in nets (Ocean Conservancy, 2010), invertebrates are not particularly susceptible to entanglement in these items. Decelerators/parachutes have large openings between the cords separating the decelerator/parachute fabric from the release mechanism. There is no plausible scenario in which decelerator/parachute cords would tighten around and hold a mobile invertebrate. Decelerators/parachutes sink slowly through the water column, although many have weights attached to their lines to speed their sinking. Invertebrates in the water column with limited mobility (e.g., jellyfish, zooplankton) could be trapped in decelerator/parachute fabric as it sinks. The potential effects of decelerators/parachutes covering sessile invertebrate species on the bottom is discussed in Section 3.4.3.4.2 (Decelerators/Parachutes).

#### F.4.5.1 Effects from Wires and Cables

Fiber-optic cables, torpedo guidance wires, sonobuoy wires, and expendable bathythermograph wires would be expended during training and testing activities. For a discussion of the types of activities that use wires and cables, see Appendix B (Activity Stressor Matrices).

A marine invertebrate could become temporarily entangled and escape unharmed, it could be held tightly enough that it could be injured during its struggle to escape, it could be preyed upon while entangled, or it could starve while entangled. The probability of these outcomes cannot be predicted because interactions between invertebrate species and entanglement hazards are not well known. However, it is unlikely that an invertebrate would become entangled in wires or cables. The items would be essentially linear after deployment, as they sink through the water column. Once the items reach the bottom, they could be moved into different shapes or loop around objects due to water currents, but the items are not expected to form tight coils, and the possibility of an invertebrate being ensnared is remote. Fiber-optic cables are relatively brittle and readily break if knotted, kinked, abraded against sharp objects, or looped beyond the items' bend radius of 3.4 millimeters. The wires and cables would eventually become buried in sediment or encrusted by marine growth, which would eliminate or further reduce the entanglement potential. The small number of items expended across the Study Area results in an extremely low rate of potential encounter for marine invertebrates.

### F.4.6 Ingestion Stressors

This section analyzes the potential ingestion effects of the various types of military expended materials used by the Navy during training and testing activities within the Study Area, which may be broadly categorized as munitions and materials other than munitions. Aspects of ingestion stressors that are applicable to marine organisms in general are presented in Section 1.5 (Conceptual Framework for Assessing Effects from Ingestion) of this appendix. The Navy expends the following types of materials that could become ingestion stressors during training and testing in the Study Area: non-explosive practice munitions (small- and medium-caliber), fragments from high-explosives, fragments from targets, chaff and flares, chaff and flare accessories (including end caps, compression pads or pistons, and o-rings), and small decelerators/parachutes. Very few invertebrates are large enough to ingest intact small- and medium-caliber munitions and casings; potential effect resulting from these items would be limited to a few taxa such as squid and octopus. Other military expended materials such as targets, large-caliber projectiles, intact training and testing bombs, guidance wires, sonobuoy tubes, and marine markers are too large for any marine invertebrate to consume and are eliminated from further discussion.

Expended materials could be ingested by marine invertebrates in all large marine ecosystems and open ocean areas. Ingestion could occur at the surface, in the water column, or at the bottom, depending on the size and buoyancy of the expended object and the feeding behavior of the animal. Floating material is more likely to be eaten by animals that may feed at or near the water surface (e.g., jellyfish, squid), while materials that sink to the bottom present a higher risk to both filter-feeding sessile (e.g., sponges) and bottom-feeding animals (e.g., crabs). Most military expended materials and fragments of military expended materials are too large to be ingested by marine invertebrates, and relatively large predatory or scavenging individuals are unlikely to consume an item that does not visually or chemically resemble food (Koehl et al., 2001; Polese et al., 2015). Many arthropods such as blue crab (*Callinectes sapidus*) and spiny lobster are known to discriminate between palatable and unpalatable food items inside the mouth, so in a strict sense, only items that are passed into the interior digestive tract should be considered to be ingested (Aggio et al., 2012). If expended material is ingested by marine invertebrates, the primary risk is blockage in the digestive tract. Most military expended materials are relatively inert in the marine environment, and are not likely to cause injury or mortality via chemical effects (see Section 3.4.3.7, Secondary Stressors, for more information on the chemical properties of these materials). However, pollutants (e.g., heavy metals and PCBs) may accumulate on the plastic components of some military expended materials. Plastic debris pieces collected at various locations in the North Pacific Ocean had polycyclic aromatic hydrocarbons and pesticides associated with them (Rios et al., 2007). Relatively large plastic pieces could be ingested by some species. However, filter- or deposit-feeding invertebrates have the greatest potential to ingest small plastic items, and any associated pollutants could harm the individual animal or subsequently be incorporated into the food chain.

The potential for marine invertebrates to encounter fragments of ingestible size increases as the military expended materials degrade into smaller fragments over months to decades. Intact munitions, fragments of munitions, and other items could degrade into metal and plastic pieces small enough to be consumed by indiscriminate feeders, such as some marine worms. Deposit-feeding, detritus-feeding, and filter-feeding invertebrates such as amphipods, polychaete worms, zooplankton, and mussels have been found to consume microscale plastic particles (microplastics) that result from the breakdown of larger plastic items (National Oceanic and Atmospheric Administration Marine Debris Program, 2014c; Wright et al., 2013). Ingestion by these types of organisms is the most likely pathway for degraded military expended materials to enter the marine food web. Transfer of microplastic particles to higher trophic levels was demonstrated in one experiment (Setala et al., 2014). Ingestion of microplastics may result in physical effects such as internal abrasion and gut blockage, toxicity due to leaching of chemicals, and exposure to attached pollutants. Potentially harmful bacteria may also grow on microplastic particles (Kirstein et al., 2016). In addition, consumption of microplastics may result in decreased consumption of natural foods such as algae (Cole et al., 2013). Microplastic ingestion by marine worms was shown in one study to result in lower energy reserves (Wright et al., 2013). Microplastic ingestion has been documented in numerous marine invertebrates (e.g., mussels, worms, mysid shrimp, bivalve molluscs, zooplankton, and scleractinian corals (Cole et al., 2013; Hall et al., 2015; Setala et al., 2016; Wright et al., 2013). In an experiment involving pelagic and benthic marine invertebrates with different feeding methods, all species exposed to microplastic particles ingested some of the items (Setala et al., 2016). Deposit-feeding worms and an amphipod species ingested the fewest particles, while bivalves and free-swimming crustaceans ingested higher amounts. Ingestion of plastic particles may result in negative physical and chemical effects to invertebrates, although invertebrates are generally able to discharge these particles from the body. Overall population-level effects across a broad range of species are currently uncertain (Kaposi et al., 2014; Wright et al., 2013).

The most abundant military expended material of ingestible size is chaff. The materials in chaff are generally nontoxic in the marine environment except in quantities substantially larger than those any marine invertebrate would likely encounter as a result of Navy training and testing activities. Chaff fibers are composed of an aluminum alloy coating on glass fibers of silicon dioxide (Section 3.0.3.3.6.3, Military Expended Materials). Chaff is similar in form to fine human hair, and is somewhat analogous to the spicules of sponges or the siliceous cases of diatoms (U.S. Department of the Navy, 1999). Many invertebrates ingest sponges, including the spicules, without suffering harm (U.S. Department of the Navy, 1999). Marine invertebrates may occasionally encounter chaff fibers in the marine environment and may incidentally ingest chaff when they ingest prey or water. Literature reviews and controlled experiments suggest that chaff poses little environmental risk to marine organisms at concentrations that could reasonably occur from military training and testing (Arfsten et al., 2002; U.S. Department of the Navy, 1999). Studies were conducted to determine the effects of chaff ingestion on various estuarine invertebrates occurring near a site of frequent chaff testing in Chesapeake Bay (Schiff, 1977). American oysters (various life stages), blue crabs (*Callinectes sapidus*), blue mussels (*Mytilus edulis*), and the polychaete worm *Nereis succinea* were force fed a chaff-and-food mixture daily for a few weeks at concentrations 10–100 times the predicted exposure level in the Bay. Although some mortality occurred in embryonic oyster larvae from 0 to 48 hours, the authors suggest confounding factors other than chaff (e.g., contaminated experimental water) as the cause. The authors reported no statistically significant mortality or effects on growth rate for any species. Because many invertebrates (e.g., crabs, shrimp) actively distinguish between food and non-food particles, the experimental design represents an unrealistic scenario with respect to the amount of chaff consumed. An investigation of sediments in portions of Chesapeake Bay exposed to aluminized chaff release for approximately 25 years found no significant increase in concentration compared to samples collected 3.7 km from the release area (Wilson et al., 2002).

As described in Section 3.4.2 (Affected Environment), many thousands of marine invertebrate species inhabit the Study Area. Most available literature regarding the effects of debris ingestion on marine invertebrates pertains to microplastics (Goldstein & Goodwin, 2013; National Oceanic and Atmospheric Administration Marine Debris Program, 2014c; Wright et al., 2013). Discussion of potential consumption of larger items is typically focused on fishes, reptiles, mammals, and birds. Consequently, it is not possible to speculate in detail on which invertebrates in which locations might ingest all types of military expended materials. Despite the potential effects, it is reasonable to conclude that relatively large military expended materials would not be intentionally consumed by actively foraging invertebrates unless they are attracted by other cues (e.g., visual cues such as flashing metal bits that squid might attack). Passively-feeding invertebrates (e.g., shellfish, jellyfish) may accidently ingest small particles by filtration or incidental adhesion to sticky mucus. The potential for effects on invertebrates from ingestion of military expended materials is also related to the locations of Navy training and testing activities relative to invertebrate population densities. Increased invertebrate densities are associated with the highest densities of microscopic plant food, which are typically located in nearshore waters in closer proximity to nutrient sources or in areas where upwelling tends to occur. Conversely, activities that generate military expended materials occur mostly seaward of nearshore water. Small deposit-feeding, detritus-feeding, and filter-feeding invertebrates would be most likely to ingest small items such as degraded plastic particles, although lobsters reportedly may also ingest microplastics (National Oceanic and Atmospheric Administration Marine Debris Program, 2014c). Though ingestion is possible in some circumstances, due to the overall size and composition of military expended materials, effects on populations would likely not be detectable.

#### F.4.6.1 Effects from Military Expended Materials – Munitions

Ingestion of intact military expended materials that are munitions is not likely for most types of expended items because they are too large to be ingested by most marine invertebrates. Though ingestion of intact munitions or large fragments is conceivable in some circumstances (e.g., a relatively large invertebrate such as an octopus or lobster ingesting a small-caliber projectile), such a scenario is unlikely due to the animal's ability to discriminate between food and non-food items. Indiscriminate deposit- and detritus-feeding invertebrates such as some marine worms could potentially ingest munitions fragments that have degraded to sediment size. Metal particles in the water column may be taken up by suspension feeders (e.g., copepods, mussels) (Chiarelli & Roccheri, 2014; Griscom & Fisher, 2004), although metal concentrations in the water are typically much lower than concentrations in sediments (Bazzi, 2014; Brix et al., 2012).

### F.4.7 Secondary Stressors

This section analyzes potential effects on marine invertebrates exposed to stressors indirectly through effects on their habitat (sediment or water quality) or prey. The assessment of potential water and sediment quality stressors refers to previous sections (Section 3.2, Sediments and Water Quality), and addresses specific activities in local environments that may affect invertebrate habitats. The terms "indirect" and "secondary" do not imply reduced severity of environmental consequences, but instead describe how the effect may occur in an organism or its ecosystem. Stressors from Navy training and testing activities that could pose indirect effects on marine invertebrates via habitat or prey include: (1) explosives and explosive byproducts, (2) chemicals other than explosives, and (3) metals.

Secondary or indirect stressors may affect benthic and pelagic invertebrates, gametes, eggs, and larvae by changes to sediment and water quality. Physical and biological features of ESA-listed black abalone critical habitat are defined in Appendix H, Section 3.2.1.1 (Status and Management). These features are rocky substrate, food resources, juvenile settlement habitat, suitable water quality, and suitable nearshore circulation patterns. Exemptions from critical habitat designation include areas offshore of San Nicolas Island and SCI. However, exemption does not preclude analysis of ESA-listed black abalones. Potential effects to rocky substrate would be associated with physical effects such as breakage or covering. Potential effects to water quality would be associated with introduction of metal, plastic, or chemical substances into the water column.

**Explosives and Explosive Byproducts**

Secondary effects on invertebrates resulting from explosions at the surface, in the water column, or on the bottom would be associated with changes to habitat structure and effects to prey species. Most explosions on the bottom would occur in soft bottom habitat and would displace some amount of sediment, potentially resulting in cratering. However, water movement would redistribute the affected sediment over time. A small amount of sediment would be suspended in the water column temporarily but would resettle to the bottom. There would be no overall reduction in the surface area or volume of sediment available to benthic species that occur on the bottom or within the substrate. Activities that inadvertently result in explosions on or near hard bottom habitat or reefs could break hard structures and reduce the amount of colonizing surface available to encrusting organisms (e.g., corals, sponges).

Explosions in the water column or on the bottom could affect invertebrate prey species. Some species of most invertebrate taxa prey upon other invertebrate species, with prey items ranging in size from zooplankton to relatively large shrimps and crabs. Therefore, in a strict sense, mortality to invertebrate species resulting from an explosion may represent a reduction in prey to other invertebrate species. A few invertebrates such as squid and some jellyfish prey upon fish, although jellyfish capture fish passively rather than through active pursuit. Therefore, fish mortality resulting from an explosion would reduce the number of potential prey items for invertebrates that consume fish. In addition to mortality, fish located near a detonation would likely be startled and leave the area, temporarily reducing prey availability until the affected area is repopulated.

Some invertebrates (e.g., worms, crustaceans, sea stars) are scavengers that would feed on any vertebrate or invertebrate animal that is killed or significantly impaired by an explosion. Therefore, scavenging invertebrates that are not killed or injured themselves could benefit from physical effects on other animals resulting from explosions in the water column or on the bottom.

High-order explosions consume most of the explosive material, leaving only small or residual amounts of explosives and combustion products. Most of the combustion products of trinitrotoluene (i.e., TNT), such as carbon dioxide and nitrogen, are common seawater constituents, although other products such as carbon monoxide are also produced (Becker, 1995). Other explosive compounds may produce different combustion products. All combustion products are rapidly diluted by ocean currents and circulation (see Section 3.2.3.1, Explosives and Explosives Byproducts). Therefore, explosives byproducts from high-order detonations would not degrade sediment or water quality or result in indirect stressors to marine invertebrates. Low-order detonations and unexploded munitions present an elevated potential for effects on marine invertebrates. Deposition of undetonated explosive materials into the marine environment can be reasonably estimated by the known failure and low-order detonation rates of high explosives (Section 3.2.3.1, Explosives and Explosives Byproducts). Explosive material not completely consumed during a detonation from munitions disposal and mine clearing training are collected after the activities are completed; therefore, potential effects are likely inconsequential and not detectable for these activities.

Exposure to relatively high concentrations of various explosive materials in sediments and in the water may result in lethal and sub-lethal effects to invertebrates. The type and magnitude of effects appear to be different among various invertebrate species and are also influenced by the type of explosive material and physical characteristics of the affected water and sediment. For example, lethal toxicity has been reported in some invertebrate species (e.g., the amphipod *Eohaustorius estuarius*) exposed to trinitrotoluene (i.e., TNT), while mortality has not been found in other species (e.g., the polychaete worm *Neanthes arenaceodentata*), even when exposed to very high concentrations (Rosen & Lotufo, 2005). Exposure to water-borne explosive materials has been found to affect reproduction or larval development in bivalve, sea urchin, and polychaete worm species (Lotufo et al., 2013). Invertebrates on the bottom may be exposed to explosive materials by ingesting contaminated sediment particles, in addition to being exposed to materials in the overlying water column or in voids in the sediment (for burrowing invertebrates). However, toxicity and other sub-lethal effects have often been associated with exposure to higher concentrations of explosive materials than the concentrations expected to occur in marine or estuarine waters of the Study Area due to training and testing activities.

Indirect effects of explosives and unexploded munitions on marine invertebrates via sediment are possible near the munitions. Rosen and Lotufo (2010) exposed mussels and deposit-feeding amphipods and polychaete worms to levels of TNT and royal demolition explosive potentially associated with a breached munition or low-order detonation. The authors found concentrations in the sediment above toxicity levels within about 1 in. of the materials, although no statistical increase in mortality was observed for any species. Concentrations causing toxicity were not found in the water column. Explosive material in the marine environment is readily degraded via several biotic and abiotic pathways, as discussed in Section 3.2.3.1 (Explosives and Explosives Byproducts). The results of studies of explosive material deposition at munitions disposal sites and active military water ranges suggest that explosives and explosives residues pose little risk to fauna living in direct contact with munitions, and that sediment is not a significant sink for these materials (Kelley et al., 2016; Koide et al., 2016; Smith & Marx, 2016). Munitions constituents and degradation products would likely be detectable only within a few feet of a degrading munition, and the spatial range of toxic sediment conditions could be less (inches). It has been suggested that the risk of toxicity to invertebrates in realistic exposure scenarios is negligible (Lotufo et al., 2013). Indirect effects of explosives and unexploded munitions on marine invertebrates via water are likely to be inconsequential. Most explosives and explosive degradation products have relatively low solubility in seawater. This means that dissolution occurs extremely slowly, and harmful concentrations of explosives and degradation products are not likely to occur in the water column. Also, the low concentration of materials delivered slowly into the water column is readily diluted by ocean currents and would be unlikely to concentrate in toxic levels. Filter feeders such as sponges or some marine worms would be exposed to chemical byproducts only in the immediate vicinity of degrading explosives (inches or less) due to the low solubility and dilution by water currents. While marine invertebrates may be adversely affected by the indirect effects of degrading explosives via water, this is unlikely in realistic scenarios.

Effects on marine invertebrates, including zooplankton, eggs, and larvae, are likely only within a very small radius of the munition (potentially inches). These effects may continue as the munition degrades over decades (Section 3.2.3.1, Explosives and Explosives Byproducts). Because most munitions are deployed as projectiles, multiple unexploded or low-order detonations would not likely accumulate on spatial scales as small as feet to inches; therefore, potential effects are likely to remain local and widely separated. Explosives, explosives byproducts, and unexploded munitions would therefore generally not be present in these habitats.

**Chemicals Other Than Explosives**

Several Navy training and testing activities introduce potentially harmful chemicals into the marine environment, primarily propellants and combustion products, other fuels, PCBs in target vessels, other chemicals associated with munitions, and simulants (Section 3.2.3.3, Chemicals Other than Explosives). Ammonium perchlorate (a rocket and missile propellant) is the most common chemical used. Perchlorate is known to occur naturally in nitrate salts, such as those from Chile, and it may be formed by atmospheric processes such as lightning and reactions between ozone and sodium chloride in the air (associated with evaporated seawater) (Dasgupta et al., 2005; Sijimol & Mohan, 2014; U.S. Environmental Protection Agency, 2014b). Perchlorate may effect metabolic processes in plants and animals. Effects have been found in earthworms and aquatic (freshwater) insects (Smith, 2002; Srinivasan & Viraraghavan, 2009), although effects specific to marine invertebrates are unknown. Other chemicals with potential for adverse effects to invertebrates include some propellant combustion products, such as hydrogen cyanide and ammonia.

Potential effects on sediments and seawater resulting from use of chemicals are discussed in Section 3.2.3.3 (Chemicals Other than Explosives). Rockets and missiles are highly efficient at consuming propellants (for example, over 99.9 percent of perchlorate is typically consumed) and, therefore, very little residual material would enter the water column. Additionally, perchlorate does not readily absorb into sediments, potentially reducing the risk to deposit- and detritus-feeding invertebrates. Torpedoes are expended in the water and, therefore, torpedo propellant (e.g., Otto Fuel II) combustion products would enter the marine environment. Overall, analysis concludes that effects on sediments and water quality would be minimal for several reasons. The size of the area affected is large and, therefore, chemicals would not be concentrated. Most propellant combustion byproducts are benign, and those of concern (e.g., hydrogen cyanide) would be quickly diluted. Most propellants are consumed during normal operations, and the failure rate of munitions using propellants and other combustible materials is low. Most byproducts of Otto Fuel II combustion occur naturally in seawater, and most torpedoes are recovered after use, limiting the potential for unconsumed fuel to enter the water. In addition, most constituents are readily degraded by biotic and abiotic processes. Concentrations of chemicals in sediment and water are not likely to cause injury or mortality to marine invertebrates, gametes, eggs, or larvae.

Target vessels are only used during sinking exercises, which occur infrequently. Polychlorinated biphenyls may be present in certain solid materials (e.g., insulation, wires, felts, and rubber gaskets) on target vessels. The vessels are selected from a list of Navy-approved vessels that have been cleaned in accordance with USEPA guidelines. Sinking exercises must be conducted at least 50 NM offshore and in water at least 6,000 ft. deep. USEPA estimates that as much as 100 lb. of PCBs remain onboard sunken target vessels. USEPA considers the contaminant levels released during the sinking of a target to be within the standards of the Marine Protection, Research, and Sanctuaries Act (16 United States Code 1341, et seq.). Under a 2014 agreement with USEPA, the Navy will not likely use aircraft carriers or submarines as the targets for a sinking exercise. As discussed in Section 3.2.3.3 (Chemicals Other than Explosives), based on these considerations, PCBs are not evaluated further as a secondary stressor to invertebrate habitats.

**Metals**

Certain metals and metal-containing compounds (e.g., cadmium, chromium, lead, mercury, zinc, copper, manganese, and many others) are harmful to marine invertebrates at various concentrations above background levels (Chan et al., 2012; Negri et al., 2002; Wang & Rainbow, 2008). For example, physiological effects in crabs, limpets, and mussels due to copper exposure were reported (Brown et al., 2004), although the effects were found at concentrations substantially higher than those likely to be encountered due to Navy expended materials. Metals are introduced into seawater and sediments as a result of training and testing activities involving vessel hulks, targets, munitions, and other military expended materials (see Section 3.2.3.2, Metals). Some effects due to metals result from the concentrating effects of bioaccumulation, which is not discussed in this section. Bioaccumulation issues are discussed in the *Ecosystem Technical Report for the Hawaii-Southern California Training and Testing (HSTT) Environmental Impact Statement* (U.S. Department of the Navy, 2013b). Secondary effects may occur when marine invertebrates are exposed by contact with the metal, contact with trace amounts in the sediment or water (e.g., from leached metals), and ingestion of contaminated sediments.

Because metals tend to precipitate out of seawater and often concentrate in sediments, potential adverse indirect effects are much more likely via sediment than water (Zhao et al., 2012). However, studies have found the concentrations of metals in the sediments of military ranges (e.g., Navy training areas such as Vieques, Puerto Rico) or munitions disposal sites, where deposition of metals is very high, to rarely be above biological effects levels (Section 3.2.3.2, Metals). For example, researchers sampled areas associated with Vieques in which live ammunition and weapons were used and found generally low concentrations of metals in the sediment (Kelley et al., 2016; Pait et al., 2010). Comparison with guidelines suggested by the National Oceanic and Atmospheric Administration's National Status and Trends Program showed that average metal concentrations were below threshold effects levels for all constituents except copper, and were below probable effects levels for all constituents. The concentration of munitions at Vieques is substantially greater than would occur in the HCTT Study Area. Evidence from a number of studies at military ranges and disposal sites indicates metal contamination is very localized (Briggs et al., 2016; Kelley et al., 2016; Koide et al., 2016). Effects on invertebrates, eggs, or larvae would likely be limited to exposure in the sediment within a few inches of the object. Refer to Section 3.2.3.2 (Metals) for more detailed study results of metal contamination in sediments at military ranges.

Concentrations of metals in seawater affected by Navy training and testing activities are unlikely to be high enough to cause injury or mortality to marine invertebrates. Benthic invertebrates occurring very near (within a few inches of) Navy-derived materials on the seafloor could be affected by associated metal concentrations, but this is expected to affect relatively few individuals.

## F.5 Habitats

### F.5.1 Acoustic Stressors

Acoustic Stressors are not applicable to habitats, due to the lack of hearing capabilities of abiotic habitats and are not analyzed further in this section.

### F.5.2 Explosive Stressors

In-water detonations are used during various mine warfare training activities, surface-to-surface gunnery exercises, air-to-surface gunnery, missile, and bombing exercises, as well as sinking exercises, in-water demolition, and other training activities. Likewise, air-to-surface gunnery, missile, and bombing tests, anti-submarine warfare tracking tests, mine warfare, detection, neutralization tests, and other testing activities also employ in-water explosives. The potential effects of in-water detonations on marine habitats are assessed according to size of charge (net explosive weight), charge radius, height above the bottom, substrate types in the area, and equations linking all these factors.

Most explosive detonations during training and testing involving the use of high-explosive munitions, including bombs, missiles, and projectile casings, would occur in the air or near the water's surface. Explosives associated with torpedoes, explosive sonobuoys, and explosive mines would occur in the water column; demolition charges could occur near the surface, in the water column, or the ocean bottom. Most surface and water column detonations would occur in waters greater than 3 NM from shore at water depths greater than 100 ft. and would not be expected to affect the bottom, although mine warfare and demolition detonations could occur in shallow water and typically in a few specific locations within the Study Area. This section only evaluates the effect of explosives placed on the bottom because the physical structure of the water column is not affected by explosions.

An explosive charge would produce percussive energy that would be absorbed and reflected by the bottom. Hard bottom would mostly reflect the energy (Berglind et al., 2009), whereas a crater would be formed in soft bottom (Gorodilov & Sukhotin, 1996). For a specific size of explosive charge, crater depths and widths would vary depending on depth of the charge and substrate type. There is a nonlinear relationship between crater size and depth of water, with relatively small crater sizes in the shallowest water, followed by a spike in size at some intermediate depth, and a decline to an average flat line (indicating similar crater size for all charge weights) at greater depth (Gorodilov & Sukhotin, 1996; O'Keeffe & Young, 1984). Radii of the craters reportedly vary little among unconsolidated substrate types (O'Keeffe & Young, 1984). On substrate types with nonadhesive particles (everything except clay), the effects should be temporary, whereas craters in clay may persist for years (O'Keeffe & Young, 1984). Soft substrate moves around with the tides and currents and depressions are only short-lived (days to weeks) unless they are maintained.

### F.5.3 Energy Stressors

Energy stressors are not applicable to habitats, since activities that include the use of energy-producing devices are typically conducted at or near the surface of the water and would not affect bottom habitats. Therefore, they are not analyzed further in this section.

### F.5.4 Physical Disturbance and Strike Stressors

#### F.5.4.1 Effects from Vessels and In-Water Devices

Vessels conducting training and testing activities in the Study Area include large ocean-going ships and submarines typically operating in waters deeper than 100 m but also occasionally transiting inshore waters from ports and through the operating areas. Training and testing activities also include smaller vessels operating in inshore waters, typically at higher speeds (greater than 10 knots). Vessels used for training and testing activities range in size from small boats (less than 40 ft.) to nuclear aircraft carriers (greater than 980 ft.) Table 3.0-14 lists representative types of vessels, including amphibious warfare vessels, used during training and testing activities. Towed mine warfare and unmanned devices are much smaller than other Navy vessels, but would also disturb the water column near the device. Some activities involve vessels towing in-water devices used in mine warfare activities. The towed devices attached to a vessel by cables are smaller than most vessels, and are not towed at high speeds. Some vessels, such as amphibious vehicles, would intentionally contact the seafloor in the surf zone.

Vessels, in-water devices, and towed in-water devices could either directly or indirectly affect any of the habitat types discussed in this section, including soft and intertidal shores, soft and hard bottoms, and artificial substrates. In addition, a vessel or device could disturb the water column enough to stir up bottom sediments, temporarily increasing the local turbidity. The shore and nearshore environment is typically very dynamic because of its constant exposure to wave action and cycles of erosion and deposition. Along high-energy shorelines like ocean beaches, these areas would be reworked by waves and tides shortly after the disturbance. Along low-energy shoreline in sheltered inshore waters, the force of vessel wakes can result in elevated erosion and resuspension of fine sediment (Zabawa & Ostrom, 1980). In deeper waters where the tide or wave action has little influence, sediments suspended into the water column would eventually settle. Sediment settlement rates are highly dependent on grain size. Disturbance of deeper bottom habitat by vessels or in-water devices is possible where the propeller wash interacts with the bottom. However, most vessel transiting in shallow, nearshore waters is confined to navigation channels where bottom disturbance only occurs with the largest vessels. An exception would be for training and testing activities that occur in shallow, nearshore environments. Turbidity caused by vessel operation in shallow water, propeller scarring, and vessel grounding could affect habitats in shallow-water areas. In addition, physical contact with hard bottom areas can cause structural damage to the substrate. However, direct effects to the substrate are typically avoided because they could slow or damage the vessel or in-water device. These disturbances would not alter the overall nature of the sediments to a degree that would impair their function as habitat. The following alternatives analysis specifies where these effects could occur in terms of number of events with vessel movement or in-water devices training/testing in different habitat areas.

#### F.5.4.2 Effects from Military Expended Materials

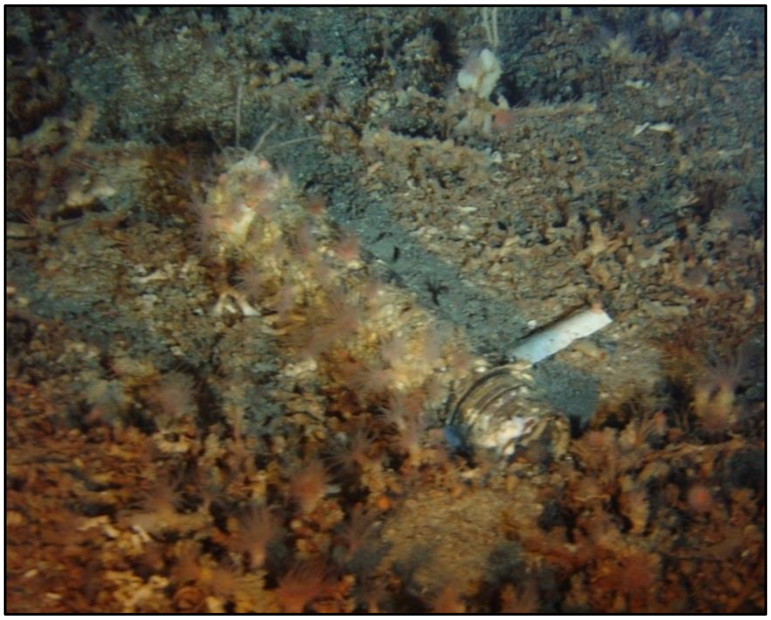
This section analyzes the potential for physical disturbance to marine substrates from the following categories of military expended materials: (1) non-explosive practice munitions, (2) fragments from high-explosive munitions, and (3) expended materials other than munitions, such as sonobuoys, expendable targets, and ship hulks. Note that expended materials do not include materials that are recovered or categorized as in-water or seafloor devices. Areas expected to have the greatest amount of expended materials are the Hawaii Study Area and California Study Area. Military expended materials have the potential to physically disturb marine substrates to the extent that they impair the substrate's ability to function as a habitat. These disturbances can result from several sources, including the effect of the expended material contacting the seafloor and moving around, the covering of the substrate by the expended material, or alteration of the substrate from one type to another.

The potential for military expended materials to physically impact marine substrates as they come into contact with the seafloor depends on several factors. These factors include, but are not limited to, the size, shape, type, density, and speed of the material through the water column; the amount of the material expended; the frequency of training or testing; water depth, water currents, or other disturbances; and the type of substrate. Most of the kinetic energy of the expended material, however, is dissipated within the first few feet of the object entering the water causing it to slow considerably by the time it reaches the substrate. Because the damage caused by a strike is proportional to the force of the strike, slower speeds result in lesser impacts. Due to the water depth at which most training and testing events take place, a direct strike on either hard bottom or artificial structures (e.g., artificial reefs and shipwrecks) is unlikely to occur with sufficient force to damage the substrate. In softer substrates (e.g., sand, mud, silt, clay, and composites), the effect of the expended material coming into contact with the seafloor, if large enough and striking with sufficient momentum, may result in a depression and a localized redistribution of sediments as they are temporarily suspended in the water column. There may also be redistribution of unconsolidated sediment in areas with sufficient flow to move the sediment, creating a pattern of scouring on one side of the material and deposition on the other.

During Navy training and testing, countermeasures such as flares and chaff are introduced into marine habitats. These types of military expended materials are not expected to affect marine habitats as strike stressors, given their smaller size and low velocity compared to projectiles, bombs, and missiles.

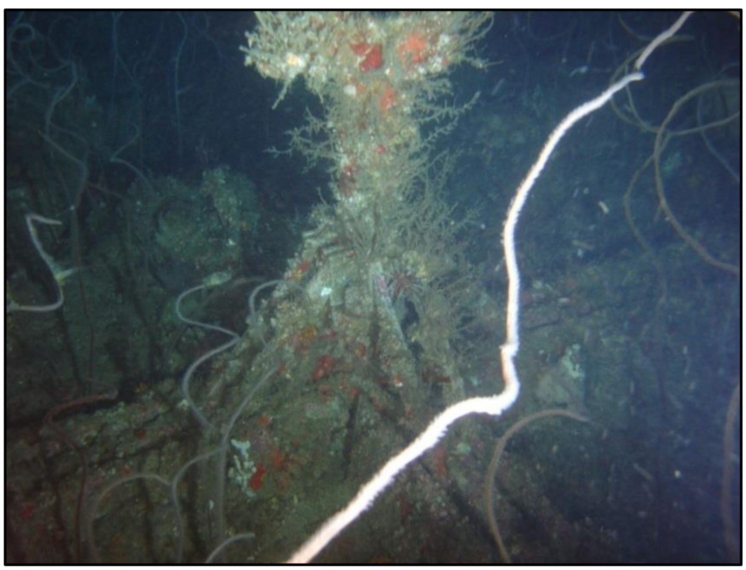
Another potential physical disturbance that military expended materials could have on marine substrates would be to cover them or to alter the type of substrate and, therefore, its function as habitat. The majority of military expended materials that settle on hard bottoms or artificial substrates, while covering the seafloor, may serve a similar habitat function as the substrate it is covering by providing a hard surface on which organisms can attach (Figure F-4 and Figure F-5). Similarity in attached organisms over the long term depends on similarity in structural features (Perkol-Finkel et al., 2006; Ross et al., 2016), fine surface texture, and mineral content (Davis, 2009). Natural hard bottom and artificial structures of a similar shape will eventually have similar communities of attached organisms if they have similar fine texture and mineral content. However, the smooth surface texture of intact military expended materials and lack of mineral content suggests a difference in species composition and associated functions. An exception would be expended materials, like the decelerators/parachutes utilized to deploy sonobuoys, lightweight torpedoes, expendable mobile anti-submarine warfare training targets, and other devices from aircraft, which would not provide a hard surface for colonization. In these cases, the hard bottom or artificial structure covered by the expended material would not be physically damaged, but would have an impaired ability to function as a habitat for colonizing or encrusting organisms. There is potential for these items to drift over shallow-water or deep-sea coral habitats.

Most military expended materials that settle on soft bottom habitats, while not damaging the actual substrate, would inhibit the substrate's ability to function as a soft-bottom habitat by covering it with a hard surface. This would effectively alter the substrate from a soft surface to a hard structure and, therefore, would alter the habitat to be more suitable for organisms more commonly found associated with hard bottom environments (U.S. Department of the Navy, 2010a, 2011). Expended materials that settle in the shallower, more dynamic environments of the continental shelf would likely be eventually covered over by sediments due to currents and other coastal processes or encrusted by organisms. Depending on the substrate properties and the hydrodynamic characteristics of the area, military expended materials may become buried rather quickly while in other areas they may persist on the surface of the seafloor for a more extended time. The offshore portion of the continental shelf experiences more sediment redistribution from oceanic currents (e.g., California Stream) than distant surface waves. The effect of oceanic currents on sediment redistribution diminishes seaward of the continental shelf break: sediment along the continental slope experiences very little reworking from surface currents and waves. In the deeper waters of the continental slope and beyond where currents do not play as large of a role, expended materials may remain exposed on the surface of the substrate with minimal change for extended periods (Figure F-6).



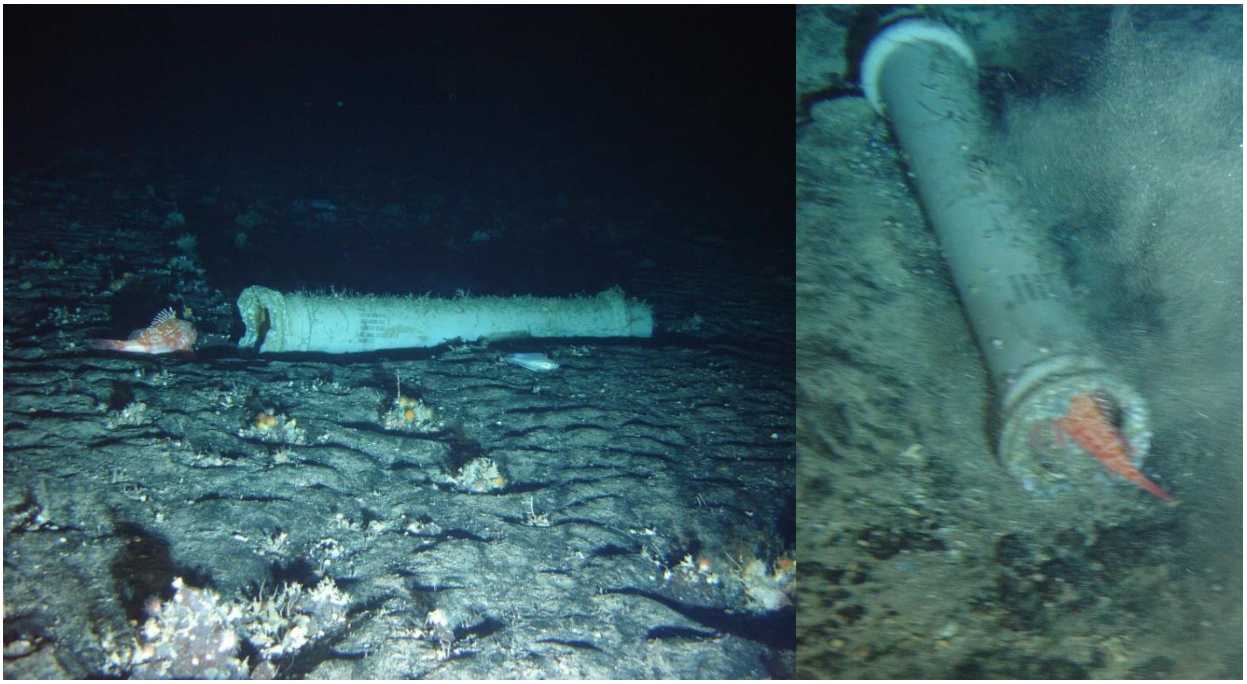
Note: Use of the smoke float as a colonizing substrate for a cluster of sea anemones (U.S. Department of the Navy, 2010a). Observed at approximately 350 meters in depth and 60 nautical miles east of Jacksonville, Florida.

**Figure F-4: A Marine Marker Observed in an Area Dominated by Coral Rubble on the Continental Slope**



Note: Encrusting organisms and benthic invertebrates readily colonize the artificial structure to a similar degree as the surrounding rock outcrop (U.S. Department of the Navy, 2010a). Observed on the ridge system that runs parallel to the shelf break at approximately 80 meters in depth and 55 nautical miles east of Jacksonville, Florida.

**Figure F-5: An Unidentified, Non-Military Structure on Hard Bottom**

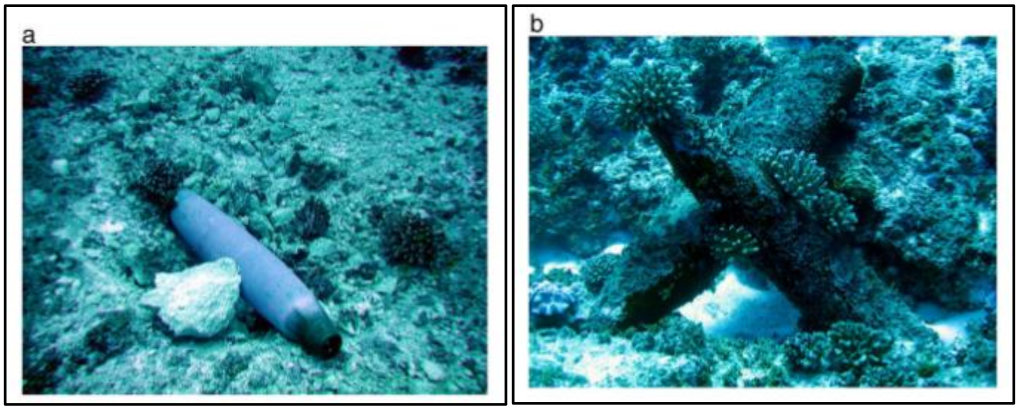


*Note: The casing was observed in a sandy area on the continental slope approximately 425 meters in depth and 70 nautical miles east of Jacksonville, Florida. The casing has not become covered by sediments or encrusting organisms due to the depth and the relatively calm, current-free environment.*

**Figure F-6: A 76-millimeter Cartridge Casing on Soft Bottom and a Blackbelly Rosefish (*Helicolenus dactylopterus*) Using the Casing for Protection When Disturbed**

Whereas the effects will accumulate somewhat through successive years of training and testing, some portion of the expended material will sink below the surface of shifting soft bottom habitat or become incorporated into natural hard bottom before crumbling into inorganic particulates. This will be the fate of military expended materials with a density greater than or equal to that of the underlying substrate (e.g., metal, cement, sand). Constituents of military expended materials that are less dense than the underlying substrate (e.g., fabric, plastic) will likely remain on the surface substrate after sinking. In this case, the effect on substrate as a habitat is likely temporary and minor due to the mobility of such materials (refer to living resources sections for more information on the entanglement and ingestion risk posed by plastic and fabric constituents of military expended materials).

The effect of dense expendable materials on bottom substrate is prolonged in the portions of the study area that are seaward of the continental shelf. Between initial settlement and burial or complete degradation, these relatively stable objects will likely function as small artificial habitats for encrusting algae, attached macroalgae/seaweed, sedentary invertebrates as well as small motile organisms (Figure F-7).




***a.*** *MK 82 inert bomb (168 centimeters long) that directly affected the sea floor at a depth of 12 meters in Z3E on 5 or 6 September 2007; photographed on 13 September 2007. Area of destruction/ disturbance was approximately 17 square meters.*
***b.*** *MK 82 bombs with Pocilloporid corals, algae, etc.*
Source: (Smith & Marx, 2016)

**Figure F-7: Military Expended Materials Functioning as Habitat**

To determine the potential level of disturbance that military expended materials have on soft, intermediate, and hard bottom substrates, an analysis to determine the impact footprint was conducted for each range complex for each alternative. Three main assumptions were made that resulted in the impact footprints calculated being considered overestimates. First, within each category of expended items (e.g., bombs, missiles, rockets, large-caliber projectiles, etc.), the size of the largest item that would be expended was used to represent the sizes of all items in the category. For example, the impact footprints of missiles used during training exercises range from 1.5 to 40 square feet. For the analyses, all missiles were assumed to be equivalent to the largest in size, or 40 square feet. Second, it was also assumed that the impact of the expended material on the seafloor was twice the size of its actual footprint. This assumption accounts for any displacement of sediments at the time of impact as well as any subsequent movement of the item on the seafloor due to currents or other forces. This should more accurately reflect the potential disturbance to soft bottom habitats, but would overestimate disturbance to hard bottom habitats since no displacement of the substrate would occur. Third, items with casings (e.g., small-, medium-, and large-caliber munitions; flares; sonobuoys; etc.) have their impact footprints doubled to account for both the item and its casing. Items and their casings were assumed to be the same size, even though, depending on the munitions, one of them is often smaller than the other.

Once the impact footprints were calculated, three analyses were performed for each range complex: (1) a conservative scenario in which potential effects to each habitat type (soft, intermediate, and hard bottom habitats) in that range complex if all expended materials settled in areas with that substrate type, (2) a proportional analysis in which potential effects to each habitat type expended materials settled proportionally across all habitat types in the area, (3) and a five-year scenario in which potential effect to the bottom habitats in that range complex over a five-year period if activities continued at anticipated levels and effects accumulated over that period. During the analyses, the same dimensions were used for high-explosive munitions as were used for non-explosive practice munitions. The total area of the seafloor covered by the expended materials should be similar regardless of whether the item is intact or fragmented, despite the fact that high-explosive munitions will explode in the air, at the surface, or in the water column and only fragments would make it to the substrate.

According to surveys conducted at Farallon De Medinilla (a U.S. Department of Defense bombing range in the Mariana Archipelago) between 1997 and 2012, there was no evidence that the condition of the living resources assessed had changed or been adversely affected to a significant degree by the training activities being conducted there. It should also be noted that the intended munition target was on the nearby land area, and water impacts were due to inaccuracy. The health, abundance, and biomass of fishes, corals, and other marine resources are comparable to or superior to those in similar habitats at other locations within the Mariana Archipelago (Smith & Marx, 2016). However, the study noted that decline in some important reef fish during their latest surveys was likely due to increasing attention from fishermen. Also, this is expected to be an extreme case based on the proximity to shallow-water coral reefs and the increased movement of military expended materials due to the shallow margins of the islands where wave impact is more severe. Effects to habitat from military expended materials in the Study Area would be expected to be less severe. See Appendix I (Military Expended Materials and Direct Strike Impact Analyses) for detailed analyses of the effects associated with military expended materials from Navy training and testing activities.

A 2023 literature review (Naval Facilities Engineering Systems Command Pacific, 2023) undertaken to consider effects of military training and testing on reef fish contaminant bioaccumulation, human, and ecological effects in the Mariana Islands resulted in several recommendations, including:

1. A recommendation to include additional references and information in future Environmental Impact Statement documents.
2. Future fate and transport studies should utilize a broader range of environmental conditions to facilitate a better understanding of explosive dynamics within the Mariana Islands Training and Testing Study Area.
3. Use additional empirical research to reduce uncertainty in future analyses.
4. Environmental Impact Statement/Overseas Environmental Impact Statement (EIS/OEIS) sections focusing on sediments, water quality, and aquatic organisms could be strengthened with additional literature and/or estimations of the concentrations of explosive compounds generated following planned activities.
5. Recommend further studies adopting a similar approach and a wider variety of exposure scenarios, such as different explosive compositions and detonation scenarios, would be useful to minimize the uncertainty associated with applying data from munitions dumpsites to military testing and training risk assessments.

#### F.5.4.3 Effects from Seafloor Devices

Mine shapes or other stationary targets and anchors are typically recovered within 7–30 days following the completion of the training or testing events. As a result of their temporary nature, recovered mine shapes do not permanently affect the substrate on which they are placed, but will temporarily impair the ability of the substrate to function as a habitat for as long as the mine shape and anchor is in place. The impairment is due to the temporary covering by artificial substrate along with changes in the bathymetry around the structures due to scouring and deposition patterns around objects on a soft bottom. Mine shapes, targets, or anchors that are not recovered would potentially have effects to abiotic habitat and, depending on the type of bottom substrate, could alter the ability to function as habitat but ultimately would likely become buried (on soft bottom) or become encrusted by similar types of organisms (on hard, intermediate, or artificial surfaces).

Potential effects of precision anchoring are qualitatively different from other seafloor devices because the activity involves repeated disturbance to the same area of seafloor. Precision anchoring training exercises involve releasing of anchors in designated locations. The intent of these training exercises is to practice anchoring the vessel within 300 ft. of the planned anchorage location. These training activities typically occur within predetermined shallow water anchorage locations near ports with seafloors consisting of soft bottom substrate. The level of effect to the soft sediments would depend on the size of the anchor used, which would vary according to vessel type. As most of these activities occur in areas along navigation channels subject to strong currents and shifting sediment, disturbed areas would quickly return to pre-disturbance conditions The Navy will implement mitigation that includes not conducting precision anchoring (except in designated anchorages) within the anchor swing circle of shallow-water coral reefs, precious coral beds, live hard bottom, artificial reefs, and shipwrecks to avoid potential effects from seafloor devices on habitats in mitigation areas throughout the Study Area. Mitigation for seafloor resources was not included in the quantitative assessment of habitat effects; however, it will help the Navy further avoid the potential for effects on habitats from precision anchoring activities.

Crawlers are fully autonomous, battery-powered amphibious vehicles used for functions such as reconnaissance missions in territorial waters. These devices are used to classify and map underwater mines in shallow water areas. The crawler is capable of traveling 2 ft. per second along the seafloor and can avoid obstacles. The crawlers are equipped with various sonar sensors and communication equipment that enable these devices to locate and classify underwater objects and mines while rejecting miscellaneous clutter that would not pose a threat.

Crawlers move over the surface of the seafloor and would not harm or alter any hard substrates encountered; therefore, hard bottom habitat would not be impaired. However, fragile abiotic or biogenic structures could be harmed by the crawlers moving over the substrate (refer to living resources sections for analysis). In soft substrates, crawlers may leave a trackline of depressed sediments approximately 2 ft. wide (the width of the device) in their wake. However, since these crawlers operate in shallow water, any disturbed sediments would be redistributed by wave and tidal action shortly (days to weeks) following the disturbance. Therefore, disturbance would not impair the ability of soft sediment to function as a habitat.

### F.5.5 Entanglement Stressors

Entanglement stressors are not applicable to habitats due to the lack of mobility capabilities of habitats and are not analyzed further in this section.

### F.5.6 Ingestion Stressors

Ingestion stressors are not applicable to habitats due to the lack of ingestion capabilities of habitats and are not analyzed further in this section.

### F.5.7 Secondary Stressors

Secondary stressors are not applicable to habitats as they are not susceptible to effects from secondary stressors and are not analyzed further in this section.

## F.6 Fishes

### F.6.1 Energy Stressors

#### F.6.1.1 Effects from In-Water Electromagnetic Devices

Several different in-water electromagnetic devices are used during training and testing activities. A discussion of the characteristics of energy introduced into the water through naval training and testing activities and the relative magnitude and location of these activities is presented in Section 3.0.3.3.3.1 (In-Water Electromagnetic Devices), while Table B-1 (Appendix B, Activity Stressor Matrices) lists the activities in each alternative that use the devices.

A comprehensive review of information regarding the sensitivity of marine organisms to electric and magnetic impulses is presented in Bureau of Ocean Energy Management (2011). The synthesis of available data and information contained in this report suggests that while many fish species (particularly elasmobranchs) are sensitive to electromagnetic fields (Hore, 2012), further investigation is necessary to understand the physiological response and magnitude of the potential effects. Most examinations of electromagnetic fields on marine fishes have focused on buried undersea cables associated with offshore wind farms in European waters (Boehlert & Gill, 2010; Gill, 2005; Ohman et al., 2007).

Many fish groups, including lampreys, elasmobranchs, eels, salmonids, and stargazers, have an acute sensitivity to electrical fields, known as electroreception (Bullock et al., 1983; Helfman et al., 2009). Fishes likely use the same sensory organs (e.g., lateral line system particularly around the head) for electroreception and also for detecting sounds. Some species of sharks, such as the scalloped hammerhead, have small pores near the nostrils, around the head, and on the underside of the snout, or rostrum called ampullae of Lorenzini to detect the electromagnetic signature of their prey. Electroreceptors are thought to aid in navigation, orientation, and migration of sharks and rays (Kalmijn, 2000). In elasmobranchs, behavioral and physiological response to electromagnetic stimulus varies by species and age, and appears to be related to foraging behavior (Rigg et al., 2009). Many elasmobranchs respond physiologically to electric fields of 10 nanovolts per centimeter (cm) and behaviorally at 5 nanovolts per cm (Collin & Whitehead, 2004), while Kajiura & Holland (2002) showed juvenile scalloped hammerhead sharks detected and behaviorally responded to electric fields of less than 1 nanovolt per cm.

There are two general types of electroreceptor organs in fishes (Helfman et al., 2009). Ampullary receptors, located in recesses in the skin, are connected to the surface by a canal filled with a conductive gel and are sensitive to electric fields of low frequency (<0.1–25 Hz). Tuberous receptors are located in depressions of the epidermis, are covered with loosely packed epithelial cells, and detect higher frequency electric fields (50 Hz to > 2 kHz). They are typically found in fishes that use electric organs to produce their own electric fields. The distribution of electroreceptors on the head of these fishes, especially around the mouth, suggests that these sensory organs may be used in foraging. Additionally, some researchers hypothesize that the electroreceptors aid in social communication (Collin & Whitehead, 2004).

While elasmobranchs and other fishes can sense the level of the earth's electromagnetic field, the potential effects on fishes resulting from changes in the strength or orientation of the background field are not well understood. When the electromagnetic field is enhanced or altered, sensitive fishes may experience an interruption or disturbance in normal sensory perception. Research on the electrosensitivity of sharks indicates that some species respond to electrical impulses with an apparent avoidance reaction (Helfman et al., 2009; Kalmijn, 2000). This avoidance response has been exploited as a shark deterrent, to repel sharks from areas of overlap with human activity (Marcotte & Lowe, 2008). A recent study on cat sharks (*Scyliorhinus canicula*) demonstrated that sharks may show habituation to electrical fields over short-term exposures (Kimber et al., 2014). Other studies suggest that sharks are attracted to electromagnetic sources when conditions in the water hinder their other senses, such as sight and hearing. This attraction to electromagnetic sources helps sharks to find prey when in these low sensory conditions (Fields, 2007).

The mechanism for direct sensing of magnetic fields is unknown; however, the presence of magnetite (a magnetic mineral) in the tissues of some fishes such as tunas and salmon, or other sensory systems such as the inner ear and the lateral line system, may be responsible for electromagnetic reception (Helfman et al., 2009). Magnetite of biogenic origins has been documented in the lateral line of the European eel (*Anguilla anguilla*) (Moore & Riley, 2009). Some species of salmon, tuna, and stargazers have likewise been shown to respond to magnetic fields and may also contain magnetite in their tissues (Helfman et al., 2009).

Experiments with electromagnetic pulses can provide indirect evidence of the range of sensitivity of fishes to similar stimuli. Two studies reported that exposure to electromagnetic pulses do not have any effect on fishes (Hartwell et al., 1991; Nemeth & Hocutt, 1990). The observed 48-hour mortality of small estuarine fishes (e.g., sheepshead minnow, mummichog, Atlantic menhaden, striped bass, Atlantic silverside, fourspine stickleback, and rainwater killifish) exposed to electromagnetic pulses of 100–200 kilovolts per m (10 nanoseconds per pulse) from distances greater than 50 m was not statistically different than the control group (Hartwell et al., 1991; Nemeth & Hocutt, 1990). During a study of Atlantic menhaden, there were no statistical differences in swimming speed and direction (toward or away from the electromagnetic pulse source) between a group of individuals exposed to electromagnetic pulses and the control group (Hartwell et al., 1991; Nemeth & Hocutt, 1990).

Electromagnetic sensitivity in some marine fishes (e.g., salmonids) is already well developed at early life stages (Ohman et al., 2007); however, most of the limited research that has occurred focuses on adults. A laboratory study on Atlantic salmon showed no behavioral changes for adults and post-smolts passing through an area with a 50 Hz magnetic field activated (Armstrong et al., 2015). Some species appear to be attracted to undersea cables, while others show avoidance (Ohman et al., 2007). Under controlled laboratory conditions, the scalloped hammerhead (*Sphyrna lewini*) and sandbar shark (*Carcharhinus plumbeus*) exhibited altered swimming and feeding behaviors in response to very weak electric fields (less than 1 nanovolt per cm) (Kajiura & Holland, 2002). In a test of sensitivity to fixed magnets, five Pacific sharks were shown to react to magnetic field strengths of 2,500–234,000 microtesla at distances ranging between 0.26 and 0.58 m and avoid the area (Rigg et al., 2009). A field trial in the Florida Keys demonstrated that southern stingrays (*Dasyatis americana*) and nurse sharks (*Ginglymostoma cirratum*) detected and avoided a fixed magnetic field producing a flux of 95,000 microtesla (O'Connell et al., 2010). A field study on white sharks (*Carcharodon carcharias*) in South Africa suggested behavioral changes in the sharks when approaching a towed prey item with an active electromagnetic field (Huveneers et al., 2013). No change was noticed in the sharks' behavior towards a static prey item. The maximum electromagnetic fields typically generated during Navy training and testing activities is approximately 2,300 microtesla.

Potential effects of electromagnetic activity on adult fishes may not be relevant to early life stages (eggs, larvae, juveniles) due to ontogenic (life stage-based) shifts in habitat utilization (Botsford et al., 2009; Sabates et al., 2007). Some skates and rays produce egg cases that occur on the bottom, while many neonate and adult sharks occur in the water column or near the water surface. Exposure of eggs and larvae (ichthyoplankton) to electromagnetic fields would be low since their distributions are extremely patchy. Early life history stages of ESA-listed steelhead occur in freshwater or estuarine habitats outside of the Study Area. For many sharks, skates, rays, and livebearers, the fecundity and natural mortality rates are much lower, and the exposure of the larger neonates and juveniles to electromagnetic energy would be similar across life stages for these species.

Based on current literature, only the fish groups identified above are capable of detecting electromagnetic fields (primarily elasmobranchs, salmonids, tuna, eels, and stargazers) and thus will be carried forward in this section. The remaining major fish groups will not be presented further. Aspects of electromagnetic stressors that are applicable to marine organisms in general are described in Section 1.2 (Conceptual Framework for Assessing Effects from Energy-Producing Activities) of this appendix.

### F.6.2 Physical Disturbance and Strike Stressors

How a physical strike affects a fish depends on the relative size of the object potentially striking the fish and the location of the fish in the water column. Before being struck by an object, salmon for example, would sense a pressure wave through the water (Hawkins & Johnstone, 1978) and have the ability to swim away from the oncoming object. The movement generated by a large object moving through the water would simply displace small fishes in open water, such as anchovies and sardines. Some fish might have time to detect the approaching object and swim away; others could be struck before they become aware of the object. An open-ocean fish that is displaced a small distance by movements from an object falling into the water nearby would likely continue on its original path as if nothing had happened. However, a bottom-dwelling fish near a sinking object would likely be disturbed, and may exhibit a general stress response, as described in Section 1 (Biological Resource Methods) of this appendix. As in all vertebrates, the function of the stress response in fishes is to rapidly alter blood chemistry levels or ratios to prepare the fish to flee or fight (Helfman et al., 2009). This generally adaptive physiological response can become a liability to the fish if the stressor persists and the fish is not able to return to its baseline physiological state. When stressors are chronic, the fish may experience reduced growth, health, or survival (Wedemeyer et al., 1990). If the object hits the fish, direct injury (in addition to stress) or death may result.

The potential responses to a physical strike are varied, but include behavioral changes such as avoidance, altered swimming speed and direction, physiological stress, and physical injury or mortality. Despite their ability to detect approaching vessels using a combination of sensory cues (e.g., sight, hearing, and lateral line), larger slow-moving fishes (e.g., whale sharks [*Rhincodon typus*], basking sharks [*Cetorhinus maximus*], manta rays [*Manta* spp.), and ocean sunfish) cannot avoid all collisions, with some collisions resulting in mortality (Braun et al., 2015; Couturier et al., 2012; Deakos et al., 2011; Foderaro, 2015; Germanov & Marshall, 2014; Graham et al., 2012; Miller & Klimovich, 2016; Ramirez-Macias et al., 2012; Rowat et al., 2007; Speed et al., 2008; Stevens, 2007). Many fishes respond by darting quickly away from the stimulus. Some other species may respond by freezing in place and adopting cryptic coloration, while still some other species may respond in an unpredictable manner. Regardless of the response, the individual must stop its current activity and divert its physiological and cognitive attention to responding to the stressor (Helfman et al., 2009). The energy costs of reacting to a stressor depend on the specific situation, but in all cases the caloric requirements of stress reactions reduce the amount of energy available to the fish for other functions, such as predator avoidance, reproduction, growth, and maintenance (Wedemeyer et al., 1990).

The ability of a fish to return to its previous activity following a physical strike (or near-miss resulting in a stress response) is a function of a variety of factors. Some fish species are more tolerant of stressors than others and become re-acclimated more easily. Within a species, the rate at which an individual recovers from a physical strike may be influenced by its age, sex, reproductive state, and general condition. A fish that has reacted to a sudden disturbance by swimming at burst speed would tire after only a few minutes; its blood hormone and sugar levels (cortisol and glucose) may not return to normal for up to, or longer than, 24 hours. During its recovery period, the fish would not be able to attain burst speeds and would be more vulnerable to predators (Wardle, 1986). If the individual were not able to regain a steady state following exposure to a physical stressor, it may suffer reduced immune function and even death (Wedemeyer et al., 1990).

Potential effects of physical disturbance or strike to adults may be different than for other life stages (e.g., eggs, larvae, juveniles) because these life stages do not necessarily occur together in the same location (Botsford et al., 2009; Sabates et al., 2007), and because they have different response capabilities. The numbers of eggs and larvae exposed to vessel movements would be low relative to total ichthyoplankton biomass (Able & Fahay, 1998); therefore, measurable effects on fish recruitment would not be expected. Also, the early life stages of most marine fishes (excluding sharks and other livebearers) already have extremely high natural mortality rates (10–85 percent per day) from predation on these life stages (Helfman et al., 2009), and therefore, most eggs and larvae are not expected to survive to the next life stage (Horst, 1977).

#### F.6.2.1 Effects from Vessels and In-Water Devices

Representative Navy vessel types, lengths, and speeds of vessels used in the Study Area is presented in Table 3.0-14. The number and location of activities for each Alternative is presented in Table 3.0-17, while Table B-1 in Appendix B (Activity Stressor Matrices) lists the activities in each alternative that use the devices.

Vessels do not normally collide with adult fishes, most of which can detect and avoid them. One study on Barents Sea capelin (*Mallotus villosus*) behavioral responses to vessels showed that most adults exhibit avoidance responses to engine noise, sonar, depth finders, and fish finders (Jorgensen et al., 2004), reducing the potential for vessel strikes. Misund (1997) found that fishes, such as polar cod (*Boreogadus saida*), haddock (*Melanogrammus aeglefinus*), jack mackerel (*Trachurus symmetricus*), sardine (*Sardina pilchardus*), herring, anchovy (*Engraulis ringens*), and capelin, that were ahead of a ship showed avoidance reactions and did so at ranges of 50–350 m. When the vessel passed over them, some fishes responded with sudden avoidance responses that included lateral avoidance or downward compression of the school. Conversely, Rostad, (2006) observed that some fishes are attracted to different types of vessels (e.g., research vessels, commercial vessels) of varying sizes, noise levels, and habitat locations. Fishes involved in that study included herring (*Clupea harengus*), sprat (*Sprattus sprattus*), and whitefish (*Merlangius merlangu*s) (Rostad et al., 2006). Fish behavior in the vicinity of a vessel is therefore quite variable, depending on the type of fish, its life history stage, behavior, time of day, and the sound propagation characteristics of the water (Schwartz, 1985). Early life stages of most fishes could be displaced by vessels and not struck in the same manner as adults of larger species. However, a vessel's propeller movement or propeller wash could entrain early life stages. The low-frequency sounds of large vessels or accelerating small vessels caused avoidance responses among herring (Chapman & Hawkins, 1973), but avoidance ended within 10 seconds after the vessel departed.

There are a few notable exceptions to this assessment of potential vessel strike effects on fish groups. Large slow-moving fishes such as whale sharks (Ramirez-Macias et al., 2012; Rowat et al., 2007; Speed et al., 2008; Stevens, 2007), basking sharks (Pacific Shark Research Center, 2017; The Shark Trust, 2017), and manta rays (Braun et al., 2015; Couturier et al., 2012; Deakos et al., 2011; Germanov & Marshall, 2014; Graham et al., 2012; Miller & Klimovich, 2016) may occur near the surface in open-ocean and coastal areas, thus making them more susceptible to ship strikes which may result in blunt trauma, lacerations, fin damage, or mortality. Stevens (2007) noted that increases in the numbers and sizes of shipping vessels in the modern cargo fleets make it difficult to gather strike-related mortality data for whale sharks because personnel on large ships are often unaware of collisions; therefore, the occurrence of vessel strikes is likely much higher than has been documented by the few studies that have been conducted. This holds true not just for whale sharks, but also for any of the aforementioned fish species.

Based on the typical physiological responses described in Section 3.5.3.4 (Physical Disturbance and Strike Stressors), vessel movements are not expected to compromise the general health or condition of individual fishes, except for large slow-moving fishes such as whale sharks, basking sharks, manta rays, and ocean sunfish (Foderaro, 2015; Rowat et al., 2007; Speed et al., 2008; Stevens, 2007).

In-water devices do not normally collide with adult fishes, as most can detect and avoid them. Fish responses to in-water devices would be similar to those discussed above for vessels. Fishes would likely show varying behavioral avoidance responses to in-water devices. Early life stages of most fishes could be displaced by in-water devices and not struck in the same manner as adults of larger species. Because in-water devices are continuously moving, most fishes are expected to move away from them or to follow behind them.

#### F.6.2.2 Effects from Military Expended Materials

While disturbance or strike from any of these objects as they sink through the water column is possible, it is not very likely for most expended materials because the objects generally sink through the water slowly and can be avoided by most, if not all fishes. Therefore, with the exception of sinking exercises, the discussion of military expended materials strikes focuses on strikes at the surface or in the upper water column from fragments (of high-explosives) and projectiles because those items have a greater potential for a fish strike as they hit the water, before slowing down as they move through the water column.

##### F.6.2.2.1 Ship Hulk

During a sinking exercise, aircraft, ship, and submarine crews fire or drop munitions on a seaborne target, usually a clean deactivated ship (Section 3.2, Sediments and Water Quality), which is deliberately sunk using multiple weapon systems. A description of sinking exercises is presented in Appendix A (Navy Activity Descriptions). Sinking exercises occur in specific open ocean areas, outside of the coastal range complexes, in waters exceeding 3,000 m (9,842.5 ft.) in depth. Direct munitions strikes from the various weapons used in these exercises are a source of potential effect. However, these effects are discussed for each of those weapons categories in this section and are not repeated in the respective sections. Therefore, the analysis of sinking exercises as a strike potential for benthic fishes is discussed in terms of the ship hulk landing on the seafloor.

##### F.6.2.2.2 Small-. Medium-, and Large-Caliber Projectiles

Various types of projectiles could cause a temporary (seconds), localized effect when they strike the surface of the water. Current Navy training and testing in the Study Area, such as gunnery exercises and testing events, include firing a variety of weapons and using a variety of non-explosive training and testing rounds, including 5-inch naval gun shells, and small-, medium-, and large-caliber projectiles. The larger-caliber projectiles are primarily used in the open ocean beyond 20 NM. Direct munitions strikes from firing weapons are potential stressors to fishes. There is a remote possibility that an individual fish at or near the surface may be struck directly if it is at the point of impact at the time of non-explosive munitions delivery. Expended rounds may strike the water surface with sufficient force to cause injury or mortality. However, limited fish species swim right at, or near, the surface of the water (e.g., with the exception of pelagic sharks, herring, salmonids, flyingfishes, jacks, tuna, mackerels, billfishes, ocean sunfishes, and other similar species).

Various projectiles would fall on soft or hard bottom habitats, where they could either become buried immediately in the sediments, or sit on the bottom for an extended time period. Most munitions would sink through the water column and come to rest on the seafloor, stirring up sediment and possibly inducing an alarm response, displacing, or injuring nearby fishes in extremely rare cases. Particular effects on a given fish species would depend on the size and speed of the munitions, the water depth, the number of rounds delivered, the frequency of training and testing, and the sensitivity of the fish (U.S. Department of the Navy, 2013c).

#### F.6.2.2.3 Bombs, Missiles, and Rockets

Direct munitions strikes from bombs, missiles, and rockets are potential stressors to fishes. Some individual fish at or near the surface may be struck directly if they are at the point of impact at the time of non-explosive munitions delivery. However, most missiles hit their target or are disabled before hitting the water. Thus, most of these missiles hit the water as fragments, which quickly dissipates their kinetic energy within a short distance of the surface. A limited number of fishes swim right at, or near, the surface of the water, as described for small-, medium-, and large-caliber projectiles.

Even though statistical modeling conducted for the Study Area (discussed in Appendix I, Military Expended Materials and Direct Strike Impact Analyses) indicates that the probability of military expended materials striking marine mammals or sea turtles is extremely low, modeling could not be conducted to estimate the probability of military expended material strikes on an individual fish. This is primarily due to the lack of fish density data available at the scale of a range complex or testing range.

In lieu of strike probability modeling, the number, size, and area of potential impact (or “footprints”) of each type of military expended material is presented in Appendix I (Military Expended Materials and Direct Strike Impact Analyses). The application of this type of footprint analysis to fish follows the notion that a fish occupying the impact area could be susceptible to potential effects, either at the water surface (e.g., pelagic sharks, salmonids, flyingfishes, jacks, tunas, mackerels, billfishes, and ocean sunfishes) or as military expended material falls through the water column and settles to the bottom (e.g., flounders, skates, and other benthic fishes listed in Table 3.6-2). Furthermore, most of the projectiles fired during training and testing activities are fired at targets, and most projectiles hit those targets, so only a very small portion of those would hit the water with their maximum velocity and force. Of that small portion, a small number of fish at or near the surface (pelagic fishes) or near the bottom (benthic fishes) may be directly affected if they are in the target area and near the expended item that hits the water surface (or bottom).

Propelled fragments are produced by an exploding bomb. Close to the explosion, fishes could potentially sustain injury or death from propelled fragments (Stuhmiller et al., 1991). However, studies of underwater bomb blasts have shown that fragments are large and decelerate rapidly (O'Keeffe & Young, 1984; Swisdak & Montanaro, 1992), posing little risk to marine organisms.

Fish disturbance or strike could result from bomb fragments (after explosion) falling through the water column in very small areas compared to the vast expanse of the testing ranges, range complexes, or the remainder of the Study Area. The expected reaction of fishes exposed to military expended materials would be to immediately leave the area where bombing is occurring, thereby reducing the probability of a fish strike after the initial expended materials hit the water surface. When a disturbance of this type concludes, the area would be repopulated and the fish stock would rebound, with inconsequential effects on the resource (Lundquist et al., 2010).

#### F.6.2.3 Effects from Seafloor Devices

The number and location of activities including seafloor devices is presented in Section 3.0.3.3.4.3 (Seafloor Devices). Additional information on stressors by testing and training activity is provided in Appendix B (Activity Stressor Matrices). Seafloor devices include items that are placed on, dropped on, or moved along the seafloor such as mine shapes, anchor blocks, anchors, bottom-placed instruments, bottom-crawling unmanned underwater vehicles, and bottom-placed targets that are not expended. As discussed in the military expended materials strike section, objects falling through the water column would slow in velocity as they sink toward the bottom and could be avoided by most, if not all fish.

Aircraft deployed mine shapes, anchor blocks, anchors, and bottom-placed instruments, and targets all have the potential to strike fish upon deployment as they sink through the water column and settle on the seafloor. Unmanned underwater vehicles (e.g., bottom crawl vehicles) also have the potential to strike a fish. Some fishes are attracted to virtually any tethered object in the water column for food or refuge (Dempster & Taquet, 2004) and could be attracted to a non-explosive mine assembly. However, while a fish might be attracted to the object, its sensory abilities allow it to avoid colliding with fixed tethered objects in the water column (Bleckmann & Zelick, 2009), so the likelihood of a fish striking one of these objects is implausible. Therefore, strike hazards associated with collision into other seafloor devices such as deployed mine shapes or anchored devices are highly unlikely to pose any strike hazard to fishes and are not discussed further.

### F.6.3 Entanglement Stressors

This section evaluates potential entanglement effects of various types of expended materials used by the Navy during training and testing activities within the Study Area. The likelihood of fishes being affected by an entanglement stressor is a function of the physical properties, location, and buoyancy of the object and the behavior and physical features of the fish, as described in Section 1.4 (Conceptual Framework for Assessing Effects from Entanglement) of this appendix. Two types of military expended materials are considered here: (1) wires and cables and (2) decelerators/parachutes.

Most entanglement observations involve abandoned or discarded nets, lines, and other materials that form loops or incorporate rings (Derraik, 2002; Keller et al., 2010; Laist, 1987; Macfadyen et al., 2009). A 25-year dataset assembled by the Ocean Conservancy reported that fishing line, rope, and fishing nets accounted for approximately 68 percent of fish entanglements, with the remainder due to encounters with various items such as bottles, cans, and plastic bags (Ocean Conservancy, 2010). No occurrences involving military expended materials were documented.

Fish entanglement occurs most frequently at or just below the surface or in the water column where objects are suspended. A smaller number involve objects on the seafloor, particularly abandoned fishing gear designed to catch bottom fishes or invertebrates (Ocean Conservancy, 2010). More fish species are entangled in coastal waters and the continental shelf than elsewhere in the marine environment because of higher concentrations of human activity (e.g., fishing, sources of entangling debris), higher fish abundances, and greater species diversity (Helfman et al., 2009; Macfadyen et al., 2009). The consequences of entanglement range from temporary and inconsequential to major physiological stress or mortality.

Some fishes are more susceptible to entanglement in derelict fishing gear and other marine debris, compared to other fish groups. Physical features, such as rigid or protruding snouts of some elasmobranchs (e.g., the wide heads of hammerhead sharks), increase the risk of entanglement compared to fishes with smoother, more streamlined bodies (e.g., lamprey and eels). Most fishes, except for jawless fishes and eels that are too smooth and slippery to become entangled, are susceptible to entanglement gear specifically designed for that purpose (e.g., gillnets).

The overall effects of entanglement are highly variable, ranging from temporary disorientation to mortality due to predation or physical injury. The evaluation of a species' entanglement potential should consider the size, location, and buoyancy of an object as well as the size, physical characteristics, and behavior of the fish species.

The following sections seek to identify entanglement potential due to military expended material. Where appropriate, specific geographic areas (open ocean areas, range complexes, and bays and inland waters) of potential effect are identified.

#### F.6.3.1 Effects from Wires and Cables

Fiber optic cables, guidance wires, and sonobuoys (which contain a wire) are used during training and testing activities. The number and location of items expended under each alternative is presented in Section 3.0.3.3.5.1 (Wires, Cables, and Nets), with additional details provided in Appendix B (Activity Stressor Matrices).

Fish groups identified in Table 3.6-2 that could be susceptible to entanglement in expended cables and wires are those such as sawfishes, with elongated snouts lined with tooth-like structures that easily snag on other similar marine debris, such as derelict fishing gear (Macfadyen et al., 2009). Some elasmobranchs (including hammerhead sharks and manta rays) and billfishes occurring within the offshore and continental shelf portions of the range complexes and testing ranges (where the potential for entanglement would occur) could be susceptible to entanglement in cables and wires. Species occurring outside the specified areas within these range complexes would not be exposed to fiber optic cables, guidance wires, or sonobuoy wire.

Once a guidance wire is released, it is likely to sink immediately and remain on the seafloor. In some cases, the wire may snag on a hard structure near the bottom and remain partially or completely suspended. The types of fish that encounter any given wire would depend, in part, on its geographic location and vertical location in the water column. In any situation, the most likely mechanism for entanglement would involve fish swimming through loops in the wire that tighten around it; however, loops are unlikely to form in a guidance wire or sonobuoy wire because of its size and rigidity (Environmental Sciences Group, 2005b).

Because of their physical characteristics, guidance wires and fiber optic cables pose a potential, though unlikely, entanglement risk to susceptible fishes. Analysis of potential entanglement for fishes is based on abandoned monofilament, nylon, and polypropylene lines used in commercial nets. Such derelict fishing gear is abundant in the ocean (Macfadyen et al., 2009) and pose a greater hazard to fishes than wires expended by the Navy. Fishing gear materials often have breaking strengths that can be up to orders of magnitude greater than that of guidance wire and fiber optic cables (Environmental Sciences Group, 2005b), and are far more prone to tangling, as discussed in Section 3.0.3.3.5.1 (Wires, Cables, and Nets). Fiber optic cables do not easily form loops, are brittle, and break easily if bent, so they pose a negligible entanglement risk. Additionally, the encounter rate and probability of effect from guidance wires and fiber optic cables are low, as few are expended (see Chapter 2, Description of Proposed Action and Alternatives, for further information).

Sonobuoys consist of a surface antenna and float unit and a subsurface hydrophone assembly unit. The two units are attached through a thin gauge dual conductor and hard draw copper strand wire, which is then wrapped by a hollow rubber tubing or bungee in a spiral configuration. The tensile breaking strength of the wire is a maximum of 40 lb. (Swope & McDonald, 2013). The length of the cable is housed in a plastic canister dispenser, which remains attached upon deployment. The length of wire that extends out is no more than 1,500 ft. and is dependent on the water depth and type of sonobuoy. Attached to the wire is a kite-drogue and damper disk stabilizing system made of non-woven nylon fabric. The nylon fabric is very thin and can be broken by hand. The cable runs through the stabilizing system and leads to the hydrophone components. The hydrophone components may be covered by thin plastic netting depending on type of sonobuoy but pose no entanglement risk. Each sonobuoy has a saltwater-activated polyurethane float that inflates when the sonobuoy is submerged and keeps the sonobuoy components floating vertically in the water column below it. Sonobuoys remain suspended in the water column for no more than 30 hours, after which they sink to the seafloor.

The sonobuoy itself is not considered an entanglement hazard for upon deployment (Environmental Sciences Group, 2005b), but its components may pose an entanglement hazard once released into the ocean. Aerial-launched sonobuoys are deployed with a decelerator/parachute. Sonobuoys contain cords, electronic components, and plastic mesh that may entangle fish (Environmental Sciences Group, 2005b). Open-ocean filter feeding species, such as basking sharks, whale sharks, scalloped hammerhead sharks, oceanic whitetip sharks, and manta rays could become entangled in these items, whereas smaller species could become entangled in the plastic mesh in the same manner as a small gillnet. Since most sonobuoys are expended in offshore areas, many coastal fishes would not encounter or have any opportunity to become entangled in materials associated with sonobuoys.

#### F.6.3.2 Effects from Decelerators/Parachutes

Decelerators/Parachutes of varying sizes are used during training and testing activities. Section 3.0.3.3.5.2 (Decelerators/Parachutes) describes the use and platforms where decelerators/parachutes would be released into the marine environment. Table 3.0-25 presents the size categories for decelerators/parachutes expended during training and testing activities that could present an entanglement risk to fishes. The types of activities that use decelerators/parachutes, physical characteristics and size of decelerators/parachutes, locations where decelerators/parachutes are used, and the number of decelerator/parachute activities proposed under each alternative are presented in Appendix B (Activity Stressor Matrices). Fishes face many potential entanglement scenarios in abandoned monofilament, nylon, polypropylene line, and other derelict fishing gear in the nearshore and offshore marine habitats of the Study Area (Macfadyen et al., 2009; Ocean Conservancy, 2010). Abandoned fishing gear is dangerous to fishes because it is abundant, essentially invisible, strong, and easily tangled. In contrast, decelerators/parachutes are rare, highly visible, and not designed to capture fishes. The weak entangling features reduce the risk to ESA-protected fishes.

Once a decelerator/parachute has been released to the water, it poses a potential entanglement risk to fishes. The Naval Ocean Systems Center identified the potential effects of torpedo air launch accessories, including decelerators/parachutes, on fish (U.S. Department of the Navy, 2001c). Unlike other materials in which fish become entangled (such as gill nets and nylon fishing line), the decelerator/parachute is relatively large and visible, reducing the chance that visually oriented fish would accidentally become entangled in it. No cases of fish entanglement have been reported for decelerators/parachutes (Ocean Conservancy, 2010; U.S. Department of the Navy, 2001a). Entanglement in a newly expended decelerator/parachute and its attachment lines while it is in the water column is unlikely because fish generally react to sound and motion at the surface with a behavioral reaction by swimming away from the source (see Section 3.6.3.5.2, Decelerators/Parachutes) and would detect the oncoming decelerator/parachute in time to avoid contact. While the decelerator/parachute is sinking, fish would have ample opportunity to swim away from the large moving object. Even if the decelerator/parachute landed directly on a fish, it would likely be able to swim away faster than the decelerator/parachute would sink because the resistance of the water would slow the decelerator/parachute's downward motion.

Once the decelerator/parachute is on the bottom, however, it is feasible that a fish could become entangled in the decelerator/parachute or its attachment lines while diving and feeding, especially in deeper waters where it is dark. If the decelerator/parachute is dropped in an area of strong bottom currents, it could billow open and pose a short-term entanglement threat to large fish feeding on the bottom. Benthic fishes with elongated spines could become caught on the decelerator/parachute or lines. Most sharks and other smooth-bodied fishes are not expected to become entangled because their soft, streamlined bodies can more easily slip through potential snares. A fish with spines or protrusions (e.g., some sharks [including hammerheads], manta rays, and billfishes,) on its body that swam into the decelerator/parachute or a loop in the lines, and then struggled, could become bound tightly enough to prevent escape. Although this scenario is possible based on the structure of the materials and the shape and behavior of fishes, it is not considered a likely event.

### F.6.4 Ingestion Stressors

Aspects of ingestion stressors that are applicable to marine organisms in general are presented in Section 1.5 (Conceptual Framework for Assessing Effects from Ingestion) of this appendix. Ingestion of expended materials by fishes could occur in coastal and open ocean areas, and can occur at or just below the surface, in the water column, or at the seafloor depending on the size and buoyancy of the expended object and the feeding behavior of the fish. Floating material is more likely to be eaten by fishes that feed at or near the water surface (e.g., ocean sunfish, basking sharks, manta rays, or flyingfishes), while materials that sink to the seafloor present a higher risk to bottom-feeding fishes (e.g., rockfishes, hammerhead sharks, skates, and flatfishes).

It is reasonable to assume that any item of a size that can be swallowed by a fish could be eaten at some time; this analysis focuses on ingestion of materials in two locations: (1) at the surface or water column and (2) at the seafloor. The potential for fish, including the ESA-listed fish species, to encounter and ingest expended materials is evaluated with respect to their feeding group and geographic range, which influence the probability that they would eat military expended materials.

The Navy expends the following types of materials during training and testing in the Study Area that could become ingestion stressors: non-explosive practice munitions (small- and medium-caliber), fragments from high-explosives, fragments from targets, chaff, flare casings (including plastic end caps and pistons), and small decelerators/parachutes. The location and number of activities that expend these items are detailed in Section 3.0.3.3.6 (Ingestion Stressors) and in Appendix B (Activity Stressor Matrices). Metal items eaten by fish are generally small (such as fishhooks, bottle caps, and metal springs), suggesting that small- and medium-caliber projectiles, pistons, or end caps (from chaff canisters or flares) are more likely to be ingested. Both physical and toxicological effects could occur as a result of consuming metal or plastic materials (Dantas et al., 2012; Davison & Asch, 2011; Possatto et al., 2011). Ingestion of plastics has been shown to increase hazardous chemicals in fish leading to liver toxicity of fishes (Rochman et al., 2013). Items of concern are those of ingestible size that either drift at or just below the surface (or in the water column) for a time or sink immediately to the seafloor. The likelihood that expended items would cause a potential effect on a given fish species depends on the size and feeding habits of the fish and the rate at which the fish encounters the item and the composition of the item. In this analysis only small- and medium-caliber munitions (or small fragments from larger munitions), chaff, small decelerators/parachutes, and end caps and pistons from flares and chaff cartridges are considered to be of ingestible size for a fish. For many small fish species (e.g., anchovy, sardines, etc.), even these items (with the exception of chaff) are often too large to be ingested, even though small pieces could sometimes be nibbled off by small fishes. Therefore, the discussion in this section focuses on those fish species large enough to potentially ingest these materials.

The analysis of ingestion effects on fishes is structured around the following feeding strategies:

#### F.6.4.1.1 Feeding at or Just Below the Surface or Within the Water Column

- **Open-Ocean Predators.** Large, migratory, open-ocean fishes, such as tunas, mahi mahi, sharks, and billfishes, feed on fast-swimming prey in the water column of the Study Area. These fishes range widely in search of unevenly distributed food patches. Smaller military expended materials could be mistaken for prey items and ingested purposefully or incidentally as the fish is swimming. A few of these predatory fishes (e.g., tiger sharks) are known to ingest any type of marine debris that they can swallow, even automobile tires. Some marine fishes, such as tunas, eat plastic fragments, strings, nylon lines, ropes, or even small light bulbs (Choy & Drazen, 2013; Rochman et al., 2015).
- **Open-Ocean Planktivores.** Plankton-eating fishes in the open-ocean portion of the Study Area include anchovies, sardines, flyingfishes, ocean sunfish, manta rays, whale sharks, and basking sharks. These fishes feed by either filtering plankton from the water column or by selectively ingesting larger zooplankton. These planktivores could encounter and incidentally feed on smaller types of military expended materials (e.g., chaff, end caps, pistons) at the surface or in the water column. Giant manta rays are the only ESA-listed species in the Study Area that is an open ocean planktivore, while some species in this group of fishes (e.g., anchovies) constitute a major prey base for many important predators, including tunas, sharks, marine mammals, and seabirds. While not a consumer of plankton, the ocean sunfish eats jellyfish and may consume a decelerator/parachute by accident at or just below the surface in the open ocean. Larger filter feeders such as whale sharks, basking sharks, and manta rays could also inadvertently ingest a decelerator/parachute.

Military expended materials that could potentially affect these types of fish at or just below the surface or in the water column include those items that float or are suspended in the water column for some period of time (e.g., decelerators/parachutes and end caps and pistons from chaff cartridges or flares).

#### F.6.4.1.2 Fishes Feeding at the Seafloor

- **Bottom Dwelling Predators.** Large predatory fishes near the seafloor are represented by rockfishes, groupers, and jacks, which are typical seafloor predators in the Study Area. These species feed opportunistically on or near the bottom, taking fish and invertebrates from the water column and from the bottom (e.g., crabs, octopus). Bottom-dwelling fishes in the nearshore coasts may feed by seeking prey and by scavenging on dead fishes and invertebrates (e.g., skates, rays, flatfishes, ratfishes).

- **Bottom Dwelling Foragers and Scavengers**. Bottom dwelling fishes may feed by seeking prey and by scavenging on dead fishes and invertebrates. Flatfishes, rays, and some sharks in the Study Area feed along the bottom on small fish and invertebrate prey, which could increase the likelihood of incidental ingestion of marine debris.

Military expended materials that could be ingested by fishes at the seafloor include items that sink (e.g., small-caliber projectiles and casings, fragments from high-explosive munitions).

Potential effects of ingestion on some adult fishes are different than for other life stages (eggs, larvae, and juveniles) because early life stages for some species are too small to ingest any military expended materials except for chaff, which has been shown to have limited effects on fishes in the concentration levels that it is released at (Arfsten et al., 2002; U.S. Department of the Air Force, 1997; U.S. Department of the Navy, 1999). Therefore, with the exception of later stage larvae and juveniles that could ingest microplastics, no ingestion potential effects on early life stages are expected.

Within the context of fish location in the water column and feeding strategies, the analysis is divided into (1) munitions (small- and medium-caliber projectiles, and small fragments from larger munitions); and (2) military expended material other than munitions (chaff, chaff end caps, pistons, decelerators/parachutes, flares, and target fragments).

#### F.6.4.2 Effects from Military Expended Materials – Munitions

Different types of explosive and non-explosive practice munitions are expended at sea during training and testing activities. This section analyzes the potential for fishes to ingest non-explosive practice munitions and fragments from high explosive munitions.

Types of non-explosive practice munitions generally include projectiles, missiles, and bombs. Of these, only small- or medium-caliber projectiles would be small enough for a large fish to ingest. Small- and medium-caliber projectiles include all sizes up to and including 2.25 inches in diameter. These solid metal materials would quickly move through the water column and settle to the seafloor. Ingestion of non-explosive practice munitions in the water column is possible when shiny fragments of the munitions sink quickly and could be ingested by fast, mobile predators that chase moving prey (e.g., tunas, jacks, billfishes, swordfishes, dolphinfishes, mackerel, wahoo, and barracudas). In addition, these fragments may also be accidentally ingested by fishes that forage on the bottom such as flatfishes, skates, and rays. Types of high explosive munitions that can result in fragments include demolition charges, projectiles, missiles, and bombs. Fragments would result from fractures in the munitions casing and would vary in size depending on the size of the net explosive weight and munitions type; however, typical sizes of fragments are unknown. These solid metal materials would quickly move through the water column and settle to the seafloor. Similar to non-explosive practice munitions described above, ingestion of high-explosive munition fragments by fast-moving mobile predators such tunas, jacks, billfishes, swordfishes, dolphinfishes, mackerel, wahoo, and barracudas in the water column is possible. In the unlikely event that explosive material, high-melting-point explosive (known as HMX), or royal demolition explosive (known as RDX) is exposed on the ocean floor, it would break down in a few hours (U.S. Department of the Navy, 2001b, 2001c). High-melting-point explosive or royal demolition explosive would not accumulate in the tissues of fish (Lotufo et al., 2010; Price et al., 1998). Fragments are primarily encountered by species that forage on the bottom.

It is possible that expended small-caliber projectiles on the seafloor could be colonized by seafloor organisms and mistaken for prey or that expended small-caliber projectiles could be accidentally or intentionally eaten during foraging. Over time, the metal may corrode or become covered by sediment in some habitats, reducing the likelihood of a fish encountering the small caliber, non-explosive practice munitions.

The potential effects of ingesting foreign objects on a given fish depend on the species and size of the fish. Fish that normally eat spiny, hard-bodied invertebrates may have tougher mouths and digestive systems than fish that normally feed on softer prey. Materials that are similar to the normal diet of a fish would be more likely to be ingested and more easily handled once ingested—for example, by fishes that feed on invertebrates with sharp appendages. These items could include fragments from high-explosives that a fish could encounter on the seafloor. Relatively small or smooth objects, such as small caliber projectiles or their casings, might pass through the digestive tract without causing harm. A small sharp-edged item could cause a fish immediate physical distress by tearing or cutting the mouth, throat, or stomach. If the object is rigid and large (relative to the fish's mouth and throat), it may block the throat or obstruct the flow of waste through the digestive system. An object may be enclosed by a cyst in the gut lining (Danner et al., 2009; Hoss & Settle, 1990). Ingestion of large foreign objects could lead to disruption of a fish's normal feeding behavior, which could be sublethal or lethal.

#### F.6.4.3 Effects from Military Expended Materials – Other than Munitions

Fishes feed throughout the water column and could mistake many types of marine debris for prey items. Ingesting nonfood items is common among a variety of marine fishes, particularly those that feed on the seafloor (Boerger et al., 2010; Hoss & Settle, 1990; Jackson et al., 2000). Many fishes are also known to accidentally ingest plastic materials, and the extent to which an individual fish might discriminate between a plastic item perceived as prey and an indistinct or less appealing shape is not clear. Once eaten, any type of plastic could cause digestive problems for the fish (Danner et al., 2009). Fishes have been reported to ingest a variety of materials or debris, such as plastic pellets, bags, rope, and line (Hoss & Settle, 1990; Jackson et al., 2000). As discussed above in Section 3.5.3.6 (Ingestion Stressors), some fish species such as the ocean sunfish eat jellyfish and may consume a decelerator/parachute at or just below the surface in the open ocean by accident. Larger filter feeders such as whale sharks, basking sharks, and manta rays could also inadvertently ingest a small or medium decelerator/parachute.

Chaff is used throughout the Study Area. It is composed of an aluminum alloy coating on glass fibers of silicon dioxide and is released or dispensed in cartridges or projectiles that contain millions of fibers. Based on the small size of chaff fibers, fish would likely not confuse the fibers with prey items or purposefully feed on them. However, some fishes could occasionally ingest low concentrations of chaff incidentally while feeding on prey items on the surface, in the water column, or the seafloor. Chaff fiber ingestion is not expected to affect fishes based on the low concentration that could reasonably be ingested and the small size of the chaff fibers. Therefore, exposure to chaff would cause no injury, mortality, or tissue damage to fishes. Potential effects of chaff ingestion by fish are not discussed further. Effects of ingestion of the end caps or pistons associated with chaff cartridges are analyzed together with effects of flares below.

Chaff end caps and pistons sink in saltwater (U.S. Department of the Navy, 1999). Fishes feeding on the where chaff canisters and flares are expended (e.g., range complexes would be more likely to encounter and ingest these items than in other locations. Ingested end caps or pistons could disrupt a fish's feeding behavior or digestive processes. If the item is particularly large relative to the fish ingesting it, the item could become permanently encapsulated by the stomach lining, and potentially lead to starvation and death (Danner et al., 2009; Hoss & Settle, 1990). The highest density of chaff and flare end caps/pistons would be expended in the Southern California Range Complex. Based on the low environmental concentration (Section 3.2, Sediments and Water Quality), it is unlikely that a larger number of fishes would ingest an end cap or piston, much less a harmful quantity. Furthermore, a fish might expel the item before swallowing it. The number of fish potentially affected by ingestion of end caps or pistons would be low based on the low environmental concentration and population-level effects are not expected to occur.

As described above, surface-feeding fishes have little opportunity to ingest end caps or pistons before they sink. However, some of these items could become entangled in dense algal mats near the surface. Predatory open-ocean fishes, such as tunas, dolphinfishes, and billfishes, are attracted to the many small prey species associated with algal mats. While foraging near the floating mats, predatory fishes may incidentally ingest end caps and pistons. The density of these items in any given location would vary based on release points and dispersion by wind and water currents. The number of end-caps and pistons that would remain at or just below the surface in algal mats and potentially available to fish is unknown. Unlike other plastic types of marine debris, end caps and pistons are heavier than water and not expected to float unless they are enmeshed in algal or other floating debris.

Most materials associated with airborne mine neutralization system activities are recovered, but pieces of fiber optic cable may be expended (U.S. Department of the Navy, 2001c). For a discussion of the physical characteristics of these expended materials, where they are used, and the number of activities in each alternative, please see Section 3.0.3.3.5.1 (Wires, Cables, and Nets). Only small amounts of fiber optic cable would be deposited onto the seafloor each year, and the small amount of fiber optic cable expended during training and testing would sink to the seafloor. Highly migratory pelagic predators (e.g., tunas, billfishes, pelagic sharks) would be unlikely to encounter the small, dispersed lengths of fiber optic cable unless they were in the immediate area when the cable was expended. The low number of fiber optic cables expended in the Study Area during this activity makes it unlikely that fishes would encounter any fiber optic cables. Potential effects of fiber optic cable ingestion by fishes are not discussed further.

## F.6.5 Secondary Stressors

### F.6.5.1 Explosions

Secondary effects on fishes resulting from explosions at the surface, in the water column, or on the bottom would be associated with changes to habitat structure and effects to prey species. Most explosions on the bottom would occur in soft bottom habitat and would displace some amount of sediment, potentially resulting in cratering. However, water movement would redistribute the affected sediment over time. A small amount of sediment would be suspended in the water column temporarily (turbidity), but would resettle to the bottom. Activities that inadvertently result in explosions on or near hard bottom habitat or reefs could break hard structures and reduce the amount of colonizing surface available to encrusting organisms (e.g., corals, sponges). Given the large spatial area of the range complexes compared to the small percentage covered by hard bottom habitat, it is unlikely that most of the small, medium, and large projectiles expended in the Study Area would fall onto this habitat type. Furthermore, these activities are distributed within discrete locations within the Study Area, and the overall footprint of these areas is quite small with respect to the spatial extent of biogenic habitat within the Study Area.

Sinking exercises could also provide secondary effects on deep sea populations. These activities occur in open-ocean areas, outside of the coastal range complexes, with potential direct disturbance or strike effects on deep-sea fishes, as covered in Section 3.6.3.4 (Physical Disturbance and Strike Stressors). Secondary effects on these fishes could occur after the ship hulks sink to the seafloor. Over time, the ship hulk would be colonized by marine organisms that attach to hard surfaces. For fishes that feed on these types of organisms, or whose abundances are limited by available hard structural habitat, the ships that are sunk during sinking exercises could provide an incidental beneficial effect on the fish community (Love & York, 2005; Macreadie et al., 2011).

The alternatives could result in localized and temporary changes to the benthic community during activities that affect fish habitat. Fish habitat could become degraded during activities that would strike the seafloor or introduce military expended materials, bombs, projectiles, missiles, rockets or fragments to the seafloor. During or following activities that affect benthic habitats, fish species may experience loss of available benthic prey at locations in the Study Area where these items might be expended. Additionally, plankton and zooplankton that are eaten by fishes may also be negatively affected by these same expended materials. The spatial area of habitat affected by the Proposed Action would be relatively small compared to the available habitat in the Study Area. However, there would still be vast expanses of habitat adjacent to the areas of habitat effect that would remain undisturbed by the Proposed Action. The majority of the physical and biological features required by steelhead are applicable to freshwater and estuaries (i.e., spawning sites, rearing sites, and migration corridors), and are outside the Study Area. Therefore, there would be no effects associated with secondary stressors.

#### F.6.5.2 Explosion By-Products

Deposition of undetonated explosive materials into the marine environment can be reasonably well estimated by the known failure and low-order detonation rates of high explosives. Undetonated explosives associated with mine neutralization activities are collected after the activity is complete; therefore, potential effects are assumed to be inconsequential for these training and testing activities, but other activities could result in unexploded munitions and unconsumed explosives on the seafloor. Fishes may be exposed by contact with the explosive, contact with contaminants in the sediment or water, and ingestion of contaminated sediments.

High-order explosions consume most of the explosive material, creating typical combustion products. In the case of royal demolition explosive, 98 percent of the products are common seawater constituents and the remainder is rapidly diluted below threshold effect level. Explosion byproducts associated with high order detonations present no indirect stressors to fishes through sediment or water. However, low order detonations and unexploded munitions present elevated likelihood of effects on fishes.

Indirect effects of explosives and unexploded munitions to fishes via sediment is possible in the immediate vicinity of the munitions. Degradation of explosives proceeds via several pathways discussed in Section 3.2 (Sediments and Water Quality). Degradation products of royal demolition explosive are not toxic to marine organisms at realistic exposure levels (Rosen & Lotufo, 2010). Trinitrotoluene (TNT) and its degradation products affect developmental processes in fishes and are acutely toxic to adults at concentrations similar to real-world exposures (Halpern et al., 2008; Rosen & Lotufo, 2010). Relatively low solubility of most explosives and their degradation products means that concentrations of these contaminants in the marine environment are relatively low and readily diluted. Furthermore, while explosives and their degradation products were detectable in marine sediment approximately 0.15–0.3 m away from degrading munitions, the concentrations of these compounds were not statistically distinguishable from background beyond 1–2 m from the degrading munitions (Section 3.2, Sediments and Water Quality). Taken together, it is likely that various life stages of fishes could be affected by the indirect effects of degrading explosives within a very small radius of the explosive (0.3–2 m).

If a high-explosive munitions does not explode, it would sink to the bottom. In the unlikely event that explosive material, high-melting-point explosive (known as HMX), or royal demolition explosive (known as RDX) is exposed on the ocean floor, it would break down in a few hours (U.S. Department of the Navy, 2001c). High-melting-point explosive or royal demolition explosive would not accumulate in the tissues of fishes (Lotufo et al., 2010; Price et al., 1998). Fish may take up TNT from the water when it is present at high concentrations but not from sediments (Lotufo et al., 2010). The rapid dispersal and dilution of TNT expected in the marine water column reduces the likelihood of a fish encountering high concentrations of TNT to near zero.

A series of research efforts focused on World War II underwater munitions disposal sites in Hawaii (Briggs et al., 2016; Edwards et al., 2016; Kelley et al., 2016; Koide et al., 2016; University of Hawaii, 2010) and an intensively used live fire range in the Mariana Islands (Smith & Marx, 2016) provide information in regard to the effects of undetonated materials and unexploded munitions on marine life. A summary of this literature which investigated water and sediment quality effects, on a localized scale, from munitions ocean disposal sites and ocean disposed dredge spoils sites is presented in the Sediments and Water Quality section and specifically in Section 3.2.3.1 (Explosives and Explosives Byproducts) and Section 3.2.3.2 (Metals). Findings from these studies indicate that there were no adverse effects on the local ecology from the presence of degrading munitions and there was no bioaccumulation of munitions-related chemicals in local marine species. Therefore, water quality effects from the use of munitions, expended material, or devices would be negligible, would have no long-term effect on water quality, and therefore would not constitute a secondary indirect stressor for fishes.

#### F.6.5.3 Metals

Certain metals and metal-containing compounds at concentrations above background levels (e.g., cadmium, chromium, lead, mercury, zinc, copper, manganese, and many others) can be toxic to fishes (Wang & Rainbow, 2008). Metals are introduced into seawater and sediments as a result of training and testing activities involving vessel hulks, targets, munitions, batteries, and other military expended materials (Section 3.2, Sediments and Water Quality). Some metals bioaccumulate, and physiological effects begin to occur only after several trophic transfers concentrate the toxic metals (U.S. Department of the Navy, 2012a). Indirect effects of metals on fish via sediment and water involve concentrations several orders of magnitude lower than concentrations achieved via bioaccumulation. Fishes may be exposed by contact with the metal, contact with contaminants in the sediment or water, and ingestion of contaminated sediments. Concentrations of metals in seawater are orders of magnitude lower than concentrations in marine sediments. It is extremely unlikely that fishes would be indirectly affected by toxic metals via the water.

#### F.6.5.4 Chemicals

Several Navy training and testing activities introduce potentially harmful chemicals into the marine environment, principally flares and propellants for rockets, missiles, and torpedoes. Polychlorinated biphenyls are discussed in Section 3.2 (Sediments and Water Quality), but there is no additional risk to fish because the Proposed Action does not introduce this chemical into the Study Area and the use of polychlorinated biphenyls has been nearly zero since 1979. Properly functioning flares, missiles, rockets, and torpedoes combust most of their propellants, leaving benign or readily diluted soluble combustion byproducts (e.g., hydrogen cyanide). Operational failures allow propellants and their degradation products to be released into the marine environment.

The greatest risk to fishes from flares, missiles, and rocket propellants is perchlorate, which is highly soluble in water, persistent, and affects metabolic processes in many plants and animals. Fishes may be exposed by contact with contaminated water or ingestion of re-suspended contaminated sediments. Since perchlorate is highly soluble, it does not readily adsorb to sediments. Therefore, missile and rocket fuels pose no risk of indirect effect on fishes via sediment. In contrast, the principal toxic components of torpedo fuel, propylene glycol dinitrate and nitrodiphenylamine, adsorbs to sediments, have relatively low toxicity, and are readily degraded by biological processes (Section 3.2, Sediments and Water Quality). It is conceivable that various life stages of fishes could be indirectly affected by propellants via sediment in the immediate vicinity of the object (e.g., within a few inches), but these potential effects would diminish rapidly as the propellant degrades.

#### F.6.5.5 Other Materials

Some military expended materials (e.g., decelerators/parachutes) could become remobilized after their initial contact with the seafloor (e.g., by waves or currents) and could pose an entanglement or ingestion hazard for fishes. For example, in some bottom types without strong currents, hard-packed sediments, and low biological productivity, items such as projectiles might remain intact for some time before becoming degraded or broken down by natural processes. These potential effects may cease only (1) when the military expended materials are too massive to be mobilized by typical oceanographic processes, (2) if the military expended materials become encrusted by natural processes and incorporated into the seafloor, or (3) when the military expended materials become permanently buried. In this scenario, a decelerator/parachute could initially sink to the seafloor, but then be transported laterally through the water column or along the seafloor, increasing the opportunity for entanglement. In the unlikely event that a fish would become entangled, injury or mortality could result. In contrast to large decelerators/parachutes, other devices with decelerators such as sonobuoys are typically used in deep open ocean areas. These areas are much lower in fish numbers and diversity, so entanglement hazards are greatly reduced for commercially and recreationally targeted species (ex. tuna, swordfish, etc.), as well as mesopelagic prey of other species. The entanglement stressor would eventually cease to pose an entanglement risk as it becomes encrusted or buried.

## F.7 Marine Mammals

### F.7.1 Energy Stressors

#### F.7.1.1 Effects from In-Water Electromagnetic Devices

There has been renewed interest in this topic of inquiry given the potential for electromagnetic fields generated by undersea power cables to possibly affect geo-navigation in migrating marine mammals (Gill et al., 2014; Kremers et al., 2016b; Kremers et al., 2014; Zellar et al., 2017). Horton et al. (2017) have indicated that future experiments involving empirical observation of free-ranging animals are still required for there to be sufficient evidence demonstrating causal relations between marine mammal movement decisions and environmental cues such as Earth's magnetic field.

Most of the early research investigated the possible correlations of where live-stranding locations occurred to determine if there was an associated local variation in Earth's magnetic field (Kirschvink, 1990; Klinowska, 1985; Walker et al., 1992). Species included long-finned and short-finned pilot whales, striped dolphin, Atlantic spotted dolphin, Atlantic white-sided dolphin, fin whale, common dolphin, harbor porpoise, sperm whale, and pygmy sperm whale, which had live stranding locations that correlated with areas where the earth's magnetic field was locally weaker than surrounding areas (Kirschvink, 1990). These statistical associations for locally weaker areas represented a total intensity variation of less than 0.05 microtesla in the magnetic field (Kirschvink et al., 1986). While this correlation seemed to have also been demonstrated for bottlenose dolphins in the Atlantic (Kirschvink et al., 1986), there was no correlation found in the Pacific (Kirschvink, 1990). Subsequent research regarding fin whale sightings over the continental shelf off the northeastern United States was consistent with the findings involving stranded fin whales (Kirschvink, 1990), supporting the hypothesis that fin whales possess a magnetic sense and that they use it to migrate (Walker et al., 1992). Bureau of Ocean Energy Management (2011) reviewed available information on electromagnetic and magnetic field sensitivity of marine organisms (including marine mammals) for effect assessment of offshore wind farms for the U.S. Department of the Interior and concluded there is no evidence to suggest any magnetic sensitivity for sea lions, fur seals, or sea otters (Bureau of Ocean Energy Management, 2011). However, the researchers concluded there was behavioral, anatomical, and theoretical evidence indicating that cetaceans sense magnetic fields.

Anatomical evidence suggests the presence of magnetic material in the brain (Pacific common dolphin, Dall's porpoise, bottlenose dolphin, Cuvier's beaked whale, and the humpback whale) and in the tongue and lower jawbones (harbor porpoise) (Bauer et al., 1985; Kirschvink, 1990). Zoeger et al. (1981) found what appeared to be nerve fibers associated with the magnetic material in a Pacific common dolphin and proposed that it may be used as a magnetic field receptor. Electrosensitivity was found in the Guiana dolphin (Czech-Damal et al., 2011). Kuzhetsov (1999) conducted experiments exposing bottlenose dolphins to permanent magnetic field intensities of 32, 108, and 168 microteslas and showed both behavioral and physiological reactions during 79 percent, 63 percent, and 53 percent of the trials, respectively (as summarized in Bureau of Ocean Energy Management (2011)). Behavioral reactions included sharp exhalations, acoustic activity, and movement, while physiological reactions included a change in heart rate. Kremers et al. (2014) conducted another experiment to observe the spontaneous reactions of captive bottlenose dolphins from a magnetized device compared to a demagnetized device. Results from this experiment confirmed that dolphins are capable of perceiving magnetic fields from a distance of more than 1.5 m from the 1.2 tesla magnetic strength device; creating a magnetic field with a strength of approximately 0.051 to 0.240 tesla between 2 to 5 cm from the source (Kremers et al., 2014). The dolphins approached the magnetized device with shorter latency compared to the demagnetized device that was identical in form and density and otherwise undistinguishable through echolocation (Kremers et al., 2014). The findings also suggest that dolphins may be able to discriminate between two items based on their magnetic properties (Kremers et al., 2016a). It is still unclear whether magnetic fields are attractive or repulsive to dolphins (Kremers et al., 2016a; Kremers et al., 2014) and further studies on the magnetic perception threshold on dolphin behavior need to be conducted (Kremers et al., 2016a).

Although it is not fully understood, based on the available evidence described above, it is probable that marine mammals use Earth's magnetic field for orientation or migration (Walker et al., 1992). If a marine mammal was in proximity of an in-water electromagnetic field source associated with Navy training and testing, emitting a field strong enough to be detected, and that animal is sensitive to the exposure, it is conceivable that this electromagnetic field could have an effect on a marine mammal, primarily affecting that animal's navigation.

#### F.7.1.2 Effects from High-Energy Lasers

As discussed in Section 3.0.3.3.3.3 (Lasers) in the 2018 Hawaii-Southern California Training and Testing (HSTT) EIS/OEIS, high-energy laser weapons are designed to disable surface targets, rendering them immobile. The primary effect from high-energy lasers would be from the laser beam striking a marine mammal at or near the water's surface, which could result in injury or death.

Marine mammals could be exposed to a laser only if the beam missed the target. Should the laser strike the sea surface, individual marine mammals at or near the surface could be exposed. The potential for exposure to a high-energy laser beam decreases as the water depth increases. Most marine mammals are unlikely to be exposed to laser activities due to mitigation measures discussed in Chapter 5 (Mitigation).

## F.7.2 Physical Disturbance and Strike Stressors

### F.7.2.1 Effects from Vessels and In-Water Devices

Surface vessels can be a source of acute and chronic disturbance for cetaceans (Au & Green, 2000; Bejder et al., 2006; Hewitt, 1985; Lusseau et al., 2009; Magalhães et al., 2002; Nowacek et al., 2007; Nowacek et al., 2004b; Richter et al., 2006; Richter et al., 2003; Schoeman et al., 2020; Watkins, 1986; Würsig & Richardson, 2009). Studies have established that cetaceans engage in avoidance behavior when surface vessels move toward them. Overall, strike avoidance success is dependent on a marine mammal’s ability to identify and locate the vessel from its radiated sound and the animal’s ability to maneuver away from the vessel in time.

Various research findings report that mysticetes have variable responses to vessels dependent on the context (Nowacek et al., 2004a; Richardson et al., 1995; Watkins, 1986). Similarly, odontocetes have also demonstrated responses to vessels. One study showed that harbor porpoises in a net-pen displayed behavioral responses (increasing swim speed or repeated alternating surfacing and diving behaviors [i.e., porpoising]) to the high-frequency components of vessel noise at long ranges (more than 1,000 m) in shallow waters (Dyndo et al., 2015). These distances correspond to where radiated noise would be more likely to elicit the response, rather than physical presence of the vessel (Dyndo et al., 2015; Palka & Hammond, 2001). Conversely, another study demonstrated that physical vessel presence, and not just noise, was associated with a short-term reduction in foraging activity in bottlenose dolphins (Pirotta et al., 2015). It is noteworthy that the dolphins associated with this report were exposed primarily to commercial and leisure boat traffic, not related to military vessel activities. Even repeated exposures from increasing vessel traffic in the same area resulting in increased responses to the disturbance may not be biologically significant. Mathematic modeling has predicted that bottlenose dolphin population dynamics would remain unchanged from a sixfold increase in vessel traffic (70 to 470 vessels per year) as dolphins are able to compensate for increased disturbance levels with little to no effects on health and vital rates (New et al., 2013). Aside from the potential for an increased risk of strike addressed below, physical disturbance from vessel use is not expected to result in more than a short-term behavioral response.

Hauled-out pinnipeds are also disturbed when approached at close distance, although the research indicates this is somewhat context-dependent. For example, one study showed that harbor seals were disturbed by tourism-related vessels, small boats, and kayaks that stopped or lingered by haulout sites, but that the seals “do not pay attention to” passing vessels at closer distances (Johnson & Acevedo-Gutiérrez, 2007). Pinnipeds in the water generally appear less responsive (Richardson et al., 1995) than those at haulout sites. Walrus and polar bears have also appeared to be attracted to vessels at times (Harwood et al., 2005) and manatees have displayed vulnerabilities to vessel impacts (Nowacek et al., 2004b).

In some circumstances, marine mammals respond to vessels with the same behavioral repertoire and tactics they employ when they encounter predators. It is not clear what environmental cue or cues marine animals might respond to; they may include the sounds of water being displaced by the ships, the sounds of the ships' engines, or a combination of environmental cues surface vessels produce while they transit. For example, in one study, North Atlantic right whales showed little overall reaction to the playback of sounds of approaching vessels, but they did respond to a novel sound by swimming strongly to the surface, which may increase their risk of strike (Nowacek et al., 2004a).

Vessel strikes from commercial, recreational, and Navy vessels are known to have resulted in serious injury and occasional fatalities to cetaceans (Abramson et al., 2011; Berman-Kowalewski et al., 2010; Calambokidis, 2012; Douglas et al., 2008; Laggner, 2009; Lammers et al., 2003; Van der Hoop et al., 2013; Van der Hoop et al., 2012). Reviews of the literature on ship strikes mainly involve strikes between commercial vessels and whales (Jensen & Silber, 2004; Laist et al., 2001). Juvenile whales of some species may be particular vulnerable to vessel strikes due to their particular habitat use and surface foraging behavior in nearshore waters, where smaller vessel number are higher (Stepanuk et al., 2021).

Vessel speed, size, and mass are all important factors in determining potential impacts of a vessel strike to marine mammals (Conn & Silber, 2013; Gende et al., 2011; Silber et al., 2010; Vanderlaan & Taggart, 2007; Wiley et al., 2016). For large vessels, speed and angle of approach can influence the severity of a strike. Based on modeling conducted by Silber et al. (2010), researchers found that whales at the surface experienced impacts that increased in magnitude with the ship's increasing speed. Another study found that there was a 3.4-fold decrease in close encounters between their research vessel and humpback whales when they traveled at speeds of 12.5 knots or less as opposed to greater than 12.5 knots (Currie et al., 2017).

#### F.7.2.1.1 Mysticetes

Vessel strikes have been documented for almost all of the mysticete species (Van der Hoop et al., 2012). This includes blue whales (Berman-Kowalewski et al., 2010; Calambokidis, 2012; Van Waerebeek et al., 2007), fin whales (Douglas et al., 2008; Van Waerebeek et al., 2007), North Atlantic right whales (Firestone, 2009; Fonnesbeck et al., 2008; Vanderlaan et al., 2009; Wiley et al., 2016) sei whales (Felix & Van Waerebeek, 2005; Van Waerebeek et al., 2007), Bryde's whales (Felix & Van Waerebeek, 2005; Van Waerebeek et al., 2007), minke whales (Van Waerebeek et al., 2007), humpback whales (Douglas et al., 2008; Lammers et al., 2003; Van Waerebeek et al., 2007), and bowhead whales (Halliday, 2020). Generally, mysticetes are larger than odontocetes and are not able to maneuver as well as odontocetes to avoid vessels. In addition, mysticetes do not typically aggregate in large groups and are therefore difficult to visually detect from the water surface.

Research suggests that the increasing noise in the ocean has made it difficult for whales to detect approaching vessels, which has indirectly raised the risk of vessel strike (Elvin & Taggart, 2008). For example, North Atlantic right whales are documented to show little overall reaction to the playback of sounds of approaching vessels, suggesting that some whales perform only a last-second flight response (Nowacek et al., 2004a). Some individuals may become habituated to low-frequency sounds from shipping and fail to respond to an approaching vessel (National Marine Fisheries Service, 2008b). Because surface activity includes feeding, breeding, and resting, whales may be engaged in this activity and not notice an approaching vessel. Acoustic shadows may also form ahead of a moving vessel, where radiated ship noise levels approach or fall below ambient noise and therefore would be hard to detect if an animal is directly ahead of the ship (Gerstein et al., 2005).

On the other hand, the lack of an acoustic cue of vessel presence can be detrimental as well. One study documented multiple cases where humpback whales struck anchored or drifting vessels; in one case a humpback whale punched a 1.5 m hole through the hull of an anchored 22 m wooden sailboat, and another instance a humpback whale rammed a powered down 10 m fiberglass sailboat (Neilson et al., 2012). These results suggest that either the whales did not detect the vessel, or they intentionally struck it. In this study, vessel strikes to multiple cetacean species were included in the investigation; however, humpback whales were the only species that displayed this type of interaction with an unpowered vessel.

Another study found that 79 percent of reported strikes between sailing vessels and cetaceans occurred when the vessels were under sail, suggesting it may be difficult for whales to detect the faint sound of sailing vessels (Ritter, 2012). However, in some instances, avoidance behavior has been observed even after exposure to noise. A blue whale was observed in a near strike with a ship while the whale was tagged with a tag that collected depth information (Szesciorka et al., 2019). A 263 m container ship approached the whale while traveling at 11.3 knots and came within 93 m of the whale while the whale was at a depth of 67.5 m ascending from a foraging dive. The whale slowed its ascent and switched to a descent dive, surfacing three minutes later. This incident took place in Southern California, and prior to the near strike with the ship, the blue whale had been exposed to simulated mid-frequency (3 to 4 kHz) active sonar (Southall et al., 2019), which ended 62 min prior to the observation presented here.

Vessel strikes are a primary threat to North Atlantic right whale survival (Firestone, 2009; Fonnesbeck et al., 2008; Knowlton & Brown, 2007; Nowacek et al., 2004a; Vanderlaan et al., 2009). Studies of North Atlantic right whales tagged in April 2009 on the Stellwagen Bank feeding grounds found that they spent most of their time at a depth of 6.5 feet (ft.), which makes them less visible at the water's surface (Bocconcelli, 2009; Parks & Wiley, 2009). Between 2017 and 2023, 12 North Atlantic right whales were confirmed to have been killed by vessel strikes, and two more are considered to have serious injuries as the result of vessel strike (Koubrak et al., 2021; Kowarski et al., 2020; National Marine Fisheries Service, 2023).

Mysticetes that occur within the Study Area have varying patterns of occurrence and distribution, which overlap with areas where vessel use associated with Navy military readiness activities would occur. For example, humpback whales that utilize the waters of the Chesapeake Bay near Naval Station Norfolk were found to spend considerable time (82 percent) engaged in foraging behavior at or near the mouth of the bay in close proximity to or directly in the shipping channel (Aschettino et al., 2020). Most of these animals were found to be juveniles, so there may be higher risk in younger animals who also have less experience maneuvering around vessels (Aschettino et al., 2020). Age-specific differences in habitat use compared to vessel density has been found in other areas within the Study Area as well (Stepanuk et al., 2021).

Risk of vessel strikes may increase depending on behavior. Increases in both nighttime foraging of some species and ship traffic overall contributes to increased risk of strike in some areas (Caruso et al., 2021). North Atlantic right whale mother-calf pairs spend 45–80 percent of their time surface resting or near-surface feeding during the first nine months of the calf's life (Cusano et al., 2019).

#### F.7.2.1.2 Odontocetes

Odontocetes that occur within the Study Area have varying patterns of occurrence and distribution, which overlap with areas where vessel use associated with Navy military readiness activities would occur. Available literature suggests based on their smaller body size, maneuverability, larger group sizes, and hearing capabilities, odontocetes are not as likely to be struck by a vessel as mysticetes. When generally compared to mysticetes, odontocetes are more capable of physically avoiding a vessel strike, and, since some species occur in large groups, they are more easily seen when they are close to the water surface.

In general, odontocetes move quickly and seem to be less vulnerable to vessel strikes than other cetaceans; however, most small whale and dolphin species have at least occasionally suffered from vessel strikes, including killer whale (Van Waerebeek et al., 2007; Visser & Fertl, 2000), short-finned and long-finned pilot whales (Aguilar et al., 2000; Van Waerebeek et al., 2007), bottlenose dolphin (Bloom & Jager, 1994; Van Waerebeek et al., 2007; Wells & Scott, 1997), white-beaked dolphin (Van Waerebeek et al., 2007), short-beaked common dolphin (Van Waerebeek et al., 2007), spinner dolphin (Camargo & Bellini, 2007; Van Waerebeek et al., 2007), striped dolphin (Van Waerebeek et al., 2007), Atlantic spotted dolphin (Van Waerebeek et al., 2007), and pygmy sperm whales (*Kogia breviceps*) (Van Waerebeek et al., 2007). Beaked whales documented in vessel strikes include Arnoux's beaked whale (Van Waerebeek et al., 2007), Cuvier's beaked whale (Aguilar et al., 2000; Van Waerebeek et al., 2007), and several species of *Mesoplodon* (Van Waerebeek et al., 2007).

However, evidence suggests that beaked whales may be able to hear the low-frequency sounds of large vessels and thus potentially avoid strike (Ketten, 1998). Sperm whales may be exceptionally vulnerable to vessel strikes as they spend extended periods of time "rafting" at the surface to restore oxygen levels within their tissues after deep dives (Jaquet & Whitehead, 1996; Watkins et al., 1999). Based on hearing capabilities and dive behavior, sperm whales may not be capable of successfully completing an escape maneuver, such as a dive, in the time available after perceiving a fast-moving vessel. This supports the suggestion that vessel speed is a critical parameter for sperm whale strike risks (Gannier & Marty, 2015). Data on vessel strikes of smaller cetaceans are generally scarce likely due, at least in part, to a reporting bias rather than strikes being less frequent (Schoeman et al., 2020).

#### F.7.2.1.3 Pinnipeds

Ship strikes were not reported as a global threat to pinniped populations by Kovacs et al. (2012). Pinnipeds in general appear to suffer fewer impacts from vessel strikes than do cetaceans or sirenians. This may be due, at least in part, to the time they spend on land resting and breeding, and their high maneuverability in the water. A review of seal stranding data from Cape Cod, Massachusetts, from 1999 to 2004 found that 622 pinniped strandings were recorded by the Cape Cod Stranding Network. Of these 622 strandings, 11 (approximately 2 percent) were found to be caused by boat strikes. Mortalities of pinnipeds (specifically harbor seals and gray seals) have initially been attributed to injuries sustained from ducted propellers on vessels such as workboats, tugs, and other support vessels (Bexton et al., 2012). However, further investigations have lead researchers to conclude that injuries that appeared to be the result of propellers were actually due to gray seal predation, cannibalism, and infanticide (Brownlow et al., 2016). Studies done in other areas have found similarly low trends—one study in the Salish Sea only found 27 instances of vessel strike out of 3,633 cases, with the majority of these cases found in pups (Olson et al., 2021).

#### F.7.2.1.4 Manatees

West Indian manatees respond to vessel movement via acoustic and possibly visual cues by moving away from the approaching vessel, increasing their swimming speed, and moving toward deeper water (Miksis-Olds et al., 2007; Nowacek et al., 2004b). The degree of the response varies with the individual manatee and may be more pronounced in deeper water where they are more easily able to locate the direction of the approaching vessel (Nowacek et al., 2004b). 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 (Nowacek et al., 2004b). Rycyk et al. (2018) found pronounced behavioral responses in tagged manatees when vessels passed within 10 m of the animal. While vessel speed did not have an effect on the occurrence, type, or number of behavioral changes observed in tagged manatees, results showed that manatees have more time to respond and changed their behavioral earlier when vessels approached slowly compared to vessels transiting on a plane at high speeds (approximately 20 miles per hour or greater) (Rycyk et al., 2018). 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). Manatees use nearshore boat channels and open water fairways as migratory and travel corridors, but have been shown to use the nearshore channel more frequently (Cloyed et al., 2019).

In addition to disturbance, West Indian manatees are particularly susceptible to vessel strikes (both strikes with the hull and propeller strikes) because they hover near the surface of the water, move very slowly, and spend most of their time in inshore waters where vessel traffic tends to be more concentrated (Calleson & Frohlich, 2007; Gerstein, 2002; Haubold et al., 2006; Runge et al., 2007). Recent modeling suggests that approximately 96 percent of adults, 70 percent of subadults, and 34 percent of calves have watercraft-related scars (Bassett et al., 2020). Vessel strikes are the direct agent of most human-caused deaths to adult West Indian manatees (Rommel et al., 2007), accounting for approximately 21 percent of all manatee deaths from 1974 to 2016 (Bassett et al., 2020), and 15 percent of all manatee injuries recorded in Florida between 2008 and 2012 (U.S. Fish and Wildlife Service, 2014). An analysis of a five-year subset (2000 to 2004) of historical mortality data suggests that a disproportionate number of propeller-caused watercraft-related mortalities could be attributed to propeller diameters greater than or equal to 17 inches (in.), suggesting that these were caused by watercraft greater than 40 ft. (Rommel et al., 2007). The USFWS indicates that manatees are probably struck by smaller watercraft more often, but the likelihood of mortality is dependent on the force of strike, which is a factor of the speed and size of the vessel. Martin et al. (2015) found that the expected number of manatee and boat encounters in a given area increased with vessel speed and distance traveled by the boat. The findings in Rycyk et al. (2018) on manatee response time to slower vessels suggest strikes with slow-moving vessels are less likely to be lethal compared to high-speed vessels.

Not all strikes are fatal, as evidenced by the fact that most West Indian manatees in Florida bear scars from previous boat strikes (Rommel et al., 2007). In fact, the Manatee Individual Photo-identification System identifies more than 3,000 Florida manatees by scar patterns mostly caused by boats, and most cataloged manatees have more than one scar pattern, indicative of multiple boat strikes (81 *Federal Register* 1000–1026, January 8, 2016). Non-lethal injuries may reduce the breeding success of females (Haubold et al., 2006) and may lower a manatee's immune response (Halvorsen & Keith, 2008).

#### F.7.2.2 Effects from Aircraft and Aerial Targets

Effects from aircraft and aerial targets are not applicable to marine mammals because they do not occur in airborne environments and will not be analyzed further in this section.

#### F.7.2.3 Effects from Military Expended Materials

The primary concern is the potential for a marine mammal to be hit with military expended material at or near the water's surface, which could result in injury or death. While disturbance or strike from an item falling through the water column is possible, it is not very likely because the objects generally sink slowly through the water and can be avoided by most marine mammals. Therefore, the discussion of military expended material strikes focuses on the potential of a strike at the surface of the water.

While no strike from military expended materials has ever been reported or recorded, the possibility of a strike still exists. Therefore, the potential for marine mammals to be struck by military expended materials was evaluated using statistical probability modeling to estimate potential direct strike exposures to a marine mammal under a worst-case scenario. Specific details of the modeling approach, including model selection and calculation methods, are presented in Appendix I (Military Expended Materials and Direct Strike Impact Analysis).

#### F.7.2.4 Effects from Seafloor Devices

The only seafloor device used during military readiness activities that has the potential to strike a marine mammal at or near the surface is an aircraft-deployed mine shape, which is used during aerial mine laying activities. These devices are identical to non-explosive practice bombs, and, therefore, the analysis of the potential effects from those devices is covered in Section F.7.2.3 (Effects from Military Expended Materials) and are not further analyzed in this section.

#### F.7.2.5 Effects from Pile Driving

Impact pile driving and vibratory pile removal as described in Chapter 2 (Description of Proposed Action and Alternatives) and Table 2.3-2 (Proposed Training Activities), was considered as a potential physical disturbance and strike stressor. This section addresses the physical presence of a temporary pier structure as a potential physical disturbance stressor and the potential for direct strike during pile driving.

Under Alternative 1 for training, pile driving use would occur during port repair activities at Naval Base Ventura County Port Hueneme.

Given the nearshore locations for this training activity and the temporary nature of the structures, it is not likely that marine mammals would experience physical disturbance from the presence of the temporary pier structure. Furthermore, it is not likely that any marine mammal would be struck by a piling during installation. Mitigation measures discussed in Chapter 5 (Mitigation) would be conducted to further reduce any potential for effects.

### F.7.3 Entanglement Stressors

This analysis includes the potential effects from three types of military expended materials: (1) wires and cables and (2) decelerators/parachutes. These materials, if encountered, may have the potential to entangle marine mammals in the Study Area at the surface, in the water column, or along the seafloor. Since potential effects depend on how a marine mammal encounters and reacts to items that pose an entanglement risk, the following subsections discuss research relevant to specific groups or species. Risk factors such as animal size, sensory capabilities, and foraging methods are also considered in the potential risk for entanglement. Most entanglements discussed are attributable to marine mammal encounters with fishing gear or other non-military materials that float or are suspended at the surface. Entanglement events are difficult to detect from land or from a boat as they may occur at considerable distances from shore and typically take place underwater. Smaller entangled animals are inherently less likely to be detected than larger ones, but larger animals may subsequently swim off while still entangled, towing lines or fishing gear behind them. Therefore, the likelihood of witnessing an entanglement event is typically low (Benjamins et al., 2014). However, the properties and size of these military expended materials, as described in the 2018 HSTT EIS/OEIS Section 3.0.3.3.5 (Entanglement Stressors) and Section 3.0.3.6.4 (Conceptual Framework for Assessing Effects from Entanglement), makes entanglement unlikely.

Since there has never been a reported or recorded instance of a marine mammal entangled in military expended materials (Henry et al., 2016; National Oceanic and Atmospheric Administration Marine Debris Program, 2014a), the Navy considered the available literature and reports on entanglement. These reports indicate that active and derelict fishing gear is the predominant cause of entanglement. The reason for this, and the ways that fishing gear may be different from military expended materials, are as follows: (1) fishing gear is most often used in areas of high productivity where whales may congregate and feed; whereas military expended materials are generally used in broad, diverse, open-ocean areas and expenditures are not concentrated; (2) fishing gear is designed to trap/entangle marine life and are made with a high breaking strength to withstand prolonged use in the ocean environment; military expended materials are not designed to persist in the ocean environment for long periods of time and are not designed to entangle or capture marine life; and (3) fishing gear and ropes are designed to float or be suspended in the water column for long periods of time, whereas most military expended materials sink immediately and rapidly.

#### F.7.3.1 Mysticetes

Mysticete species with documented entanglement reports include humpback whales, North Atlantic right whales, Rice's whales, minke whales, and bowhead whales (Cassoff et al., 2011; National Oceanic and Atmospheric Administration Marine Debris Program, 2014a). Aside from Rice's whales, the aforementioned species have records directly linking entanglement to marine debris as opposed to active fishing gear (Baulch & Perry, 2014; Laist, 1997). It has been estimated that a minimum of 52 percent and a maximum of 78 percent of whales have been non-lethally entangled in their lifetime in some populations (Neilson et al., 2009). In 2020, there were 25 reports of live entangled large whales along the east coast of the United States, and 33 in 2019 (National Marine Fisheries Service, 2022a, 2022b).

Entanglement of many large whales most often begins with rope being caught in its baleen plates. Based on feeding adaptations for mysticetes, oral entanglement may pose one of the greatest threats to survival, due to impaired foraging and possibly loss of function of the hydrostatic seal (formed when upper and lower lips come together and keep the mouth closed), requiring the whale to expend energy to actively keep the mouth closed during swimming (Cassoff et al., 2011). Impaired foraging could lead to deterioration of health, making the animal more susceptible to disease or eventual starvation over a long period of time, or chronic poor body condition which could result in suppressions to growth, age of sexual maturation and calving rates (Christiansen et al., 2020).

Compounding the issue, trailing lengths of rope or line may become wrapped around the animal's appendages as it struggles to free itself (Kozuck, 2003), limiting the animal's mobility and increasing drag. This reduced mobility can also reduce foraging success or even limit the animal's ability to surface. Notably, the single acute cause of entanglement mortalities has been associated with drowning from multiple body parts being entangled (Cassoff et al., 2011). Even if a whale is freed of an entanglement, the recovery time is estimated to be an average of 1.3–3 months (Moore et al., 2021; van der Hoop et al., 2017), extending the sub-lethal effects of an entanglement.

Common sources of entanglements for mysticetes include line and net fragments attached through the mouth or around the tail and flippers (National Oceanic and Atmospheric Administration Marine Debris

Program, 2014a). Rope diameter and breaking strengths may also determine an animal's ability to break free from entanglement. Increased rope strength has been found to be positively correlated with injury severity in right whales, but not for humpback whales (Knowlton et al., 2016). Minke whales were also found entangled in lower breaking strength ropes (10.47 kilonewtons [2,617 pound (lb.)-force]) than both humpback and right whales (17.13 and 19.30 kilonewtons [3,851 and 4,339 lb.-force], respectively) (Knowlton et al., 2016). These are significantly greater than the breaking strength of torpedo guidance wires (maximum 42 lb.-force) as described in the 2018 HSTT EIS/OEIS's Section 3.0.3.3.5.1 (Wires and Cables). Entanglement would be more likely for materials with similar physical properties as those described above.

In the western North Atlantic, entanglement in fishing gear is a known cause of humpback whale injury and mortality, with all components of both pot and gillnet gear documented during 30 separate humpback whale entanglement events (Johnson et al., 2005). This study also found one entanglement event involving a vessel anchor line rather than fishing gear. Overall, between 6 and 26 percent (average 12 percent) of the population exhibits evidence of new entanglement injuries every year (Robbins, 2009), though the proportion of entanglements due to fishing gear is unknown. Available data indicate that males typically have more entanglement scars than females and may become entangled more frequently. Juvenile whales were found to have a higher rate of entanglement and be more at risk of serious injury and mortality when entangled than mature animals of the same species (Robbins, 2009, 2010).

Military expended material is expected to sink to the ocean floor. It is possible that marine mammals could encounter these items within the water column as they sink to the bottom. Less buoyant items that sink faster are not as likely to become entangled with a marine mammal compared to more buoyant materials that would sink slower to the floor. Mysticetes that occupy the water column or skim feed along the water surface would have to encounter a military expended material at the same time and location it is either expended or as it sinks.

Almost 3 percent of all right whale sightings between 1980 and 2016, and over half of all cataloged North Atlantic right whales (58 percent) have been observed with seafloor sediment on their bodies, which suggests these whales make frequent contact with the seafloor (Hamilton & Kraus, 2019). Mysticete species that feed near or at the bottom in the areas where activities are conducted that result in military expended materials could encounter items that have already sunk and, therefore, do not have to be present at the precise time when items are expended.

#### F.7.3.2 Odontocetes

Odontocete species with documented records of marine debris entanglement, excluding fishing gear, are the sperm whale, bottlenose dolphin, harbor porpoise, and Dall's porpoise (National Oceanic and Atmospheric Administration Marine Debris Program, 2014a). Bottlenose dolphins are the most commonly entangled odontocete, with most entanglements involving monofilament line, net fragments, and rope attached to appendages (National Oceanic and Atmospheric Administration Marine Debris Program, 2014a). Heezen (1957) reported two confirmed instances of sperm whales entangled in the slack lengths of telegraph cable near cable repair sites along the seafloor. These whales likely became entangled while feeding along the bottom, as the cables were found wrapped around the jaw. Other sperm whale entanglements in gill nets have been reported, resulting in various behavioral responses, injuries and in some cases, mortalities (Haase & Felix, 1994; Jacobsen et al., 2010; Pace et al., 2008). Juvenile harbor porpoises exposed to 0.5-in. diameter white nylon ropes in both vertical and horizontal planes treated the ropes as barriers, more frequently swimming under than over them. However, porpoises feeding on fish in the area crossed the ropes more frequently and became less cautious, suggesting that rope poses a greater risk in a feeding area than in a transit area. For harbor porpoises feeding along the bottom, rope suspended near the seafloor is more likely to entangle than rope higher in the water column given the animals' natural tendency is to swim beneath barriers (Kastelein et al., 2005).

#### F.7.3.3 Pinnipeds

Entanglement is considered a serious threat to several populations of pinnipeds (Kovacs et al., 2012); 67 percent of pinniped species have been recorded as entangled (Kuhn et al., 2015). Younger pinnipeds appear to be more prone to entanglement than adults (Hofmeyr et al., 2006; Page et al., 2004). Seals can get entangled in nets and fishing line when young and then grow with the lines wrapped around their necks or appendages, causing deep wounds and eventually death. Death may occur by strangulation or severing of the arteries (Derraik, 2002). Between 2004 and 2008, the annual mean entanglement rate for gray seals at a haul-out site in Cornwall (in the United Kingdom) ranged from 3.6 to 5 percent, and mortality rates were likely higher for entangled animals (Allen et al., 2012). Gray and harbor seals also become entangled and drown in the U.S. Northeast Sink Gillnet Fishery (Johnston et al., 2015).

#### F.7.3.4 Polar Bear

In a review conducted by (Kühn & Van Franeker, 2020) on the interaction between marine debris and wildlife, only one occurrence of entanglement in polar bears was documented, but no further details regarding the material was provided.

#### F.7.3.5 West Indian Manatee

Entanglements have been documented for manatees (Beck & Barros, 1991; Forrester et al., 1975; O'Shea et al., 1985). Manatee foraging behaviors may predispose them to entanglement with fishery gear due to their tactile nature, meaning they need to be in close proximity or physically touching an object to gain extensive information about it (Adimey et al., 2014). In addition, manatees have limited abilities to detect finer objects, such as monofilament, until they have already come into contact with it, leading to an increased risk of entanglement (Bauer et al., 2012).

Fishery gear interactions with Florida manatees were analyzed from stranding records collected between 1997 and 2009 and results found that approximately 8 percent of the manatee cases were identified as fishery gear interactions (Adimey et al., 2014). Of the 380 reported cases, 76 percent consisted of hook and line interactions and 22 percent were from trap pot gear (Adimey et al., 2014).

### F.7.4 Ingestion Stressors

#### F.7.4.1 Mysticetes

Since baleen whales feed by filtering large amounts of water, they likely encounter and consume plastic debris at higher rates than other marine animals (National Oceanic and Atmospheric Administration Marine Debris Program, 2014b). Species that feed at the surface or in the water column include blue, fin, Bryde's, minke, and sei whales. While humpback whales may feed by lunging through the water after krill and fish, there are data confirming that humpback whales display bottom-feeding behaviors in areas of high concentrations of preferred prey, the northern sand lance (*Ammodytes dubius*) (Hain et al., 1995; Ware et al., 2014).

Baleen whales are believed to routinely encounter microplastics within the marine environment based on concentrations of these items and baleen whale feeding behaviors (Andrady, 2011). Observations of bowhead whale mouths have provided insights into potential threats to bowhead and right whales from oral entanglement of marine debris, including a greater probability of lethal consequences due to interference of the hydrostatic oral seal (Lambertsen et al., 2005). In a comprehensive review of documented ingestion of debris by marine mammals by Laist (1997), there are two species of mysticetes (bowhead and minke whale) with records of having ingested debris items that included plastic sheeting and a polythene bag. This effort was followed up by a comparative summary of the earlier review with additional information and the number of mysticete species with documented records of ingestion increased to seven species, including right whales, pygmy right whales, gray whales, and four rorqual species (Bergmann et al., 2015). Similarly, information compiled by (Williams et al., 2011) listed humpback whale, fin whale, minke whale as three species of mysticetes known to have ingested debris including items the authors characterized as fishing gear, polyethylene bag, plastic sheeting, plastic bags, rope, and general debris. Besseling et al. (2015) documented the first occurrence of microplastics in the intestines of a humpback whale. Anthropogenic debris have been found in the digestive tract of Southern right whales (Alzugaray et al., 2020), so it is probable that North Atlantic right whales also ingest marine debris.

Feeding behaviors of mysticete species suggest that potential encounters with ingestion stressors would only occur when debris items at the water's surface have spatial and temporal overlap with skim feeding animals, or while whales are engulfing prey in the water column as items sink to the bottom. Bottom-feeding humpback whales may also encounter ingestion stressors that have already sunk.

#### F.7.4.2 Odontocetes

In a comprehensive review of documented ingestion of debris by marine mammals, odontocetes had the most ingestion records, with 21 species represented (Laist, 1997). A follow-up to this review revealed an increase in odontocete ingestion of marine debris. Additionally, a follow-up to this review by Bergmann et al. (2015) revealed marine debris ingestion for odontocetes has increased, where 40 species now have documented records of ingestion.

Beaked whales use suction feeding to ingest benthic prey and may incidentally ingest other items (MacLeod et al., 2003). Both sperm whales and beaked whales are known to incidentally ingest foreign objects while foraging; however, this does not always result in negative consequences to health or vitality (Laist, 1997; Walker & Coe, 1990). While this incidental ingestion has led to sperm whale mortality in some cases, (Whitehead, 2003) suggested the scale to which this affects sperm whale populations was not substantial. Sperm whales are recorded as having ingested fishing net scraps, rope, wood, and plastic debris such as plastic bags and items from the seafloor (Jacobsen et al., 2010; Walker & Coe, 1990; Whitehead, 2003).

Weaned juveniles who are investigating multiple types of prey items may be particularly vulnerable to ingesting non-food items, as found in a study of juvenile harbor porpoises (Baird & Hooker, 2000). Similarly, a male pygmy sperm whale reportedly died from blockage of two stomach compartments by hard plastic, and a Blainville's beaked whale (*Mesoplodon densirostris*) washed ashore in Brazil with a ball of plastic thread in its stomach (Derraik, 2002). In one study, all 12 animals investigated from six odontocete species in the eastern Atlantic were found to have ingested microplastics, primarily fibers, and none larger than 5 mm (Montoto-Martínez et al., 2021).

#### F.7.4.3 Pinnipeds

Pinnipeds are opportunistic foragers, primarily feeding within the water column, but may also forage on the seafloor. Most of the seal species within the Study Area feed both within the water column and on the seafloor, while walruses feed primarily on benthic invertebrates (Bluhm & Grandinger, 2008). In a review of documented ingestion of debris by marine animals, 36 percent of seal species were found to have ingested plastics (Kuhn et al., 2015). Laist (1997) reported ingestion of Styrofoam cups by northern elephant seals and Steller sea lions, and (Bravo Rebolledo et al., 2013) reported plastics in the stomach contents of harbor seals. Plastic debris have been recorded in at least one hooded seal pup (Pinzone et al., 2021). The possibility of ingested debris transfer through predator-prey interactions has also been demonstrated by Eriksson and Burton (2003) in fur seals. As such, the risk of indirect ingestion of debris by marine mammals is dependent on the likelihood they are consuming contaminated prey.

#### F.7.4.4 Polar Bears

Polar bears feed primarily on other marine mammals (especially ringed seals, bearded seals, and harp seals) while on land and ice or out at sea (Bluhm & Grandinger, 2008). Plastics have also been found when assessing food items identified in scat samples (Iversen et al., 2013).

#### F.7.4.5 West Indian Manatee

Manatees feed on seagrass beds in relatively shallow coastal or estuarine waters. In a comprehensive review of documented ingestion of debris by marine mammals, the West Indian manatee had ingestion records that included monofilament line, plastic bags, string, twine, rope, fish hooks, wire, paper, cellophane, and rubber bands (Laist, 1997). Some researchers suggest that manatees incidentally ingest fishing gear and plastic while foraging on plants in shallow habitats where debris can accumulate and become entwined in the food resources (Adimey et al., 2014; Beck & Barros, 1991). Ingestion of fishing gear can cause impaction, abdominal infections, inversions of the intestine (Beck & Barros, 1991) and other indirect effects.

### F.7.5 Secondary Stressors

This section analyzes potential effects on marine mammals exposed to stressors indirectly through effects on their habitat (sediment or water quality) or prey. For the purposes of this analysis, indirect effects on marine mammals via sediment or water quality that do not require trophic transfer (e.g., bioaccumulation) to be observed are considered here. The invertebrates (Section 3.4), marine habitats (Section 3.5), and fish (Section 3.6) analyses indicated minimal to no effects on potential prey species of marine mammals due to bioaccumulation. It is important to note that the terms “indirect” and “secondary” do not imply reduced severity of environmental consequences but instead describe how the effect may occur in an organism. Bioaccumulation is considered in the *Ecosystem Technical Report for the Atlantic Fleet Training and Testing (AFTT) Final Environmental Impact Statement* (U.S. Department of the Navy, 2012b).

Stressors from Navy military readiness activities that could pose indirect effects on marine mammals via habitat or prey include: (1) explosives, (2) explosive byproducts and unexploded munitions, (3) metals, (4) chemicals, and (5) transmission of disease and parasites. Analyses of the potential effects on sediment and water quality are discussed in Section 3.2 (Sediment and Water Quality).

#### F.7.5.1 Explosives

Explosives may have an effect on marine mammal prey species. In addition to the physical effects of an underwater blast, prey might have behavioral reactions to underwater sound. For instance, prey species might exhibit a strong startle reaction to explosions that might include swimming to the surface or scattering away from the source. This startle and flight response is the most common secondary defense among animals (Hanlon & Messenger, 1996; Mather, 2004). The abundances of prey species near the detonation point could be diminished for a short period of time before being repopulated by animals from adjacent waters. Alternatively, any prey species that would be directly injured or killed by the blast could draw in scavengers from the surrounding waters that would feed on those organisms, and in turn could be susceptible to becoming directly injured or killed by subsequent explosions. Any of these scenarios would be temporary, only occurring during activities involving explosives, and no lasting effect on prey availability or the pelagic food web would be expected.

#### F.7.5.2 Explosion Byproducts and Unexploded Munitions

A series of research efforts that focused on World War II underwater munitions disposal sites in Hawaii (Briggs et al., 2016; Edwards et al., 2016; Kelley et al., 2016; Koide et al., 2016; University of Hawaii, 2010) and an intensively used live fire range in the Mariana Islands (Smith & Marx, 2016) provide information regarding the effects of undetonated materials and unexploded munitions on marine species. Section 3.2.3.1 (Explosives and Explosives Byproducts) and Section 3.2.3.2 (Metals) contain a summary of this literature which investigated water and sediment quality effects, on a localized scale, from munitions ocean disposal sites and ocean disposed dredge spoils sites. Findings from these studies indicate that there were no adverse effects on the local ecology from the presence of degrading munitions and there was no bioaccumulation of munitions-related chemicals in local marine species.

The island of Farallon De Medinilla (in the Mariana Islands) has been used as a target area since 1971. Between 1997 and 2012, there were 14 underwater scientific survey investigations around the island providing a long-term insight to the potential effects on the marine life from training and testing involving the use of munitions (Smith & Marx, 2016). Munitions use has included high-explosive rounds from gunfire, high-explosives bombs by Navy aircraft and U.S. Air Force B-52s, in addition to the expenditure of inert rounds and non-explosive practice bombs. Marine life assessed during these surveys included algae, corals, benthic invertebrates, sharks, rays, and bony fishes, and sea turtles. Over the 16-year period, investigators found no evidence that the condition of biological resources has been adversely affected to a significant degree by the training activities (Smith & Marx, 2016). Furthermore, they found that the health, abundance, and biomass of fishes, corals, and other marine resources were comparable, or superior to, those in similar habitats at other locations within the Mariana Archipelago.

These findings are consistent with other assessments such as the Potomac River Test Range at Dahlgren, Virginia, which was established in 1918 and is the nation’s largest fully instrumented, over-the-water gun-firing range. Munitions tested at Naval Surface Warfare Center, Dahlgren have included rounds from small-caliber guns up to the Navy’s largest (16-in. guns), bombs, rockets, mortars, grenades, mines, depth charges, and torpedoes (U.S. Department of the Navy, 2013e). Results from the assessment indicate that munitions expended at Naval Surface Warfare Center, Dahlgren have not contributed to significant concentrations of metals in the Potomac River. Sediment contribution of metals is orders of magnitude less than concentrations already present in the Potomac River from natural and manmade sources (U.S. Department of the Navy, 2013e).

#### F.7.5.3 Metals

Metals are introduced into seawater and sediments as a result of military readiness activities involving ship hulks, targets, munitions, and other military expended materials (Section 3.2.3.2, Metals) (Environmental Sciences Group, 2005a). Some metals bioaccumulate and physiological effects begin to occur only after several trophic transfers concentrate the toxic metals (Section 3.5, Habitats, and Chapter 4, Cumulative Effects). Evidence from a number of studies (Briggs et al., 2016; Kelley et al., 2016; Koide et al., 2016; U.S. Department of the Navy, 2013e; University of Hawaii, 2010) indicate metal contamination is very localized and that bioaccumulation resulting from munitions cannot be demonstrated. Specifically in sampled marine life living on or around munitions on the seafloor, metal concentrations could not be definitively linked to the munitions since comparison of metals in sediment next to munitions show relatively little difference in comparison to other "clean" marine sediments used as a control/reference (Koide et al., 2016). Research has demonstrated that some smaller marine organisms are attracted to metal munitions as a hard substrate for colonization or as shelter (Kelley et al., 2016; Smith & Marx, 2016), but this is unlikely to substantively affect marine mammal prey availability.

#### F.7.5.4 Chemicals

Several Navy military readiness activities introduce chemicals into the marine environment that are potentially harmful at higher concentrations; however, rapid dilution would occur, and toxic concentrations are unlikely to be encountered. Introduced chemicals are principally from flares and propellants from missiles and torpedoes. Properly functioning flares, missiles, and torpedoes combust most of their propellants, leaving benign, or readily diluted soluble combustion byproducts (e.g., hydrogen cyanide). Operational failures may allow propellants and their degradation products to be released into the marine environment. Flares and missiles that operationally fail may release perchlorate, which is highly soluble in water, persistent, and affect s metabolic processes in many plants and animals if in sufficient concentration. Such concentrations are not likely to persist in the ocean. Research has demonstrated that perchlorate did not bioconcentrate or bioaccumulate, which was consistent with the expectations for a water-soluble compound (Furin et al., 2013). Perchlorate from failed expendable items is therefore unlikely to compromise water quality to that point that it would act as a secondary stressor to marine mammals. It should also be noted that chemicals in the marine environment as a result of Navy military readiness activities would not occur in isolation and are typically associated with military expended materials that release the chemicals while in operation. Because marine mammal avoidance of an expended flare, missile, or torpedo in the water is almost certain, it would further reduce the potential for introduced chemicals to act as a secondary stressor.

#### F.7.5.5 Transmission of Marine Mammal Diseases and Parasites

The U.S. Navy deploys trained common bottlenose dolphins (*Tursiops truncatus*) and California sea lions (*Zalophus californianus*) for integrated training involving two primary mission areas: to find objects such as inert mine shapes, and to detect swimmers or other intruders around Navy facilities such as piers. When deployed, the animals are part of what the Navy refers to as Marine Mammal Systems. These Marine Mammal Systems include one or more motorized small boats, several crew members, and a trained marine mammal. Based on the standard procedures with which these systems are deployed, it is not reasonably foreseeable that use of these Marine Mammals Systems would result in the transmission of disease or parasites to cetaceans or pinnipeds in the Study Area based on the following.

Each trained animal is deployed under behavioral control to find the intruding swimmer or submerged object. Upon finding the target of the search, the animal returns to the boat and alerts the animal handlers that an object or swimmer has been detected. In the case of a detected object, the human handlers give the animal a marker that the animal can bite onto and carry down to place near the detected object. In the case of a detected swimmer, animals are given a localization marker or leg cuff that they are trained to deploy via a pressure trigger. After deploying the localization marker or leg cuff, the animal swims free of the area to return to the animal support boat. For detected objects, human divers or remote vehicles are deployed to recover the item. Swimmers that have been marked with a leg cuff are reeled in by security support boat personnel via a line attached to the cuff.

Marine Mammal Systems deploy approximately one to two weeks before the beginning of a training exercise to allow the animals to acclimate to the local environment. Four to 12 marine mammals are involved per exercise. Marine Mammal Systems typically participate in object detection and recovery, both participating in mine warfare exercises and assisting with the recovery of non-explosive mine shapes at the conclusion of an exercise. Marine Mammal Systems may also participate in port security and anti-terrorism/force protection exercises.

During the past 40 years, the Navy Marine Mammal Program has been deployed globally. To date, there have been no known instances of deployment-associated disease transfer to or from Navy marine mammals. Navy animals are maintained under the control of animal handlers and are prevented from having sustained contact with indigenous animals.

When not engaged in the training event, Navy marine mammals are either housed in temporary enclosures or aboard ships involved in training exercises. All marine mammal waste is disposed of in a manner approved for the specific holding facilities. When working, sea lions are transported in boats, and dolphins are transferred in boats or by swimming alongside the boat under the handler's control. Their open-ocean time is under stimulus control and is monitored by their trainers.

Navy marine mammals receive excellent veterinarian care (per Secretary of the Navy Instruction 3900.41E). Appendix A, Section 8, of the Swimmer Interdiction Security System Final EIS (U.S. Department of the Navy, 2009b) presents an overview of the veterinary care provided for the Navy's marine mammals. Appendix B (Activity Stressor Matrices), Section 2, of the Swimmer Interdiction Security System Final EIS presents detailed information on the health screening process for communicable diseases. The following is a brief summary of the care received by all of the Navy's marine mammals:

- Qualified veterinarians conduct routine and pre-deployment health examinations on the Navy's marine mammals; only animals determined as healthy are allowed to deploy.
- Restaurant-quality frozen fish are fed to prevent diseases that can be caused by ingesting fresh fish (e.g., parasitic diseases).
- Navy animals are routinely dewormed to prevent parasitic and protozoal diseases.
- If a valid and reliable screening test is available for a regionally relevant pathogen (e.g., polymerase chain reaction assays for morbillivirus), such tests are run on appropriate animal samples to ensure that animals are not shedding these pathogens.

The Navy Marine Mammal Program routinely does the following to further mitigate the low risk of disease transmission from captive to wild marine mammals during training exercises:

- Marine mammal waste is disposed of in an approved system dependent upon the animal's specific housing enclosure and location.
- Onsite personnel are made aware of the potential for disease transfer and report any sightings of wild marine mammals so that all personnel are alert to the presence of the animal.
- Marine mammal handlers visually scan for indigenous marine animals for at least five minutes before animals are deployed and maintain a vigilant watch while the animal is working in the water. If a wild marine mammal is seen approaching or within 100 m, the animal handler will

hold the marine mammal in the boat or recall the animal immediately if the animal has already been sent on the mission.
- The Navy obtains appropriate state agriculture and other necessary permits and strictly adheres to the conditions of the permit.

Due to the limited amount of time that the Navy marine mammals spend in the open ocean, the control that the trainers have over the animals, the collection and proper disposal of marine mammal waste, the exceptional screening and veterinarian care given to the Navy's animals, the visual monitoring for indigenous marine mammals, and more than 40 years with zero known incidents, there is no scientific basis to conclude that the use of Navy marine mammals during training activities will have an effect on wild marine mammals.

## F.8 Reptiles

### F.8.1 Energy Stressors

#### F.8.1.1 Effects from High-Energy Lasers

As discussed in Section 3.0.3.3.3.3 (High-Energy Lasers), high-energy laser weapons testing involves the use of up to 30 kilowatts of directed energy as a weapon against small surface vessels and airborne targets. These weapons systems are deployed from a surface ship to create small but critical failures in potential targets and used at short ranges from the target.

This section analyzes the potential effects of high-energy lasers on sea turtles. Sea snakes were not included in the model—it is generally assumed that sea snake occurrence within the Study Area is very rare. Because of the low density of sea snakes in open ocean areas where high-energy laser testing would occur, sea snakes are assumed to not be affected by high-energy laser strikes due to the extremely low likelihood of exposure. Therefore, sea snakes are not discussed further in the analysis for potential effects on reptiles by testing activities using high-energy lasers.

The primary concern for high-energy weapons testing is the potential for a sea turtle to be struck by a high-energy laser beam at or near the water's surface, which could result in injury or death, resulting from traumatic burns from the beam.

Sea turtles could be exposed to a laser only if the beam missed the target. Should the laser strike the sea surface, individual sea turtles at or near the surface could be exposed. The potential for exposure to a high-energy laser beam decreases as the water depth increases. Because laser platforms are typically helicopters and ships, sea turtles at sea would likely transit away or submerge in response to other stressors, such as ship or aircraft noise, although some sea turtles may not exhibit a response to an oncoming vessel or aircraft, increasing the risk of contact with the laser beam.

As discussed in Section 3.0.3.3.3.3 (High-Energy Lasers), high-energy laser use associated with training and testing activities would occur within the Hawaii Study Area and California Study Area. Training and testing activities have the potential to expose sea turtles that occur within these areas to this energy stressor.

Appendix I (Military Expended Materials and Direct Strike Impact Analyses) includes a conservative approach for estimating the probability of a direct laser strike on a sea turtle during testing and training activities. The Navy analysis assumes: (1) that all sea turtles would be at or near the surface 100 percent of the time, and would not account for the duration of time a sea turtle would be diving; and (2) that sea turtles are stationary, which does not account for any movement or any potential avoidance of the training or testing activity in response to other stressors (e.g., vessel noise).

The Navy compiled density data from several sources and developed a protocol to select the best available data sources based on species, area, time (season), and type of density model. The resulting GIS database, called the Navy Marine Species Density Database (U.S. Department of the Navy, 2017b), includes seasonal density values for sea turtle species present within the Study Area. When aerial surveys are used to collect data on sea turtle occurrence it is often difficult to distinguish between the different sea turtle species. To account for the known occurrence of multiple sea turtle species in the Study Area and the general lack of species-specific occurrence data for most species, a sea turtle guild, composed of green and hawksbill turtle sightings, was created to estimate sea turtle densities in the Hawaii Study Area. The sea turtle guild was not used to estimate sea turtle densities in the transit corridor (eastern or western portions) or for the California Study Area due to the scarcity of sea turtle sightings data in these areas.

While the analysis of sea turtle guild survey data applies to all species, it is more reflective of green turtles, which account for nearly all sightings in the Hawaii Study Area. The number of observations of hawksbill turtles would be so low as to render the data unusable for estimating density of this species. By considering the hawksbill and green turtle sightings together, a more powerful result can be provided for sea turtles as a guild. In theory, the guild also encompasses leatherback, olive ridley, and loggerhead turtles, but these species have not been identified during the collection of Navy monitoring data. The Navy's modeling results show a probability of 0.000064 strikes per year on a sea turtle. Based on the assumptions used in the statistical probability analysis, there is a high level of certainty in the conclusion that no sea turtle that occurs in the Study Area would be struck by a high-energy laser.

## F.8.2 Physical Disturbance and Strike Stressors

### F.8.2.1 Effects from Vessels and In-Water Devices

#### Vessels

Sea turtles spend a majority of their time submerged (Renaud & Carpenter, 1994; Sasso & Witzell, 2006), though Hazel et al. (2009) and Hazel et al. (2007) showed most species of sea turtles staying within the top 3 m of water despite deeper water being available. Any of the sea turtle species found in the Study Area can occur at or near the surface in open ocean and coastal areas, whether feeding or periodically surfacing to breathe. Distribution of species is not uniform, however. Typically in Hawaii, loggerheads and olive ridleys are not seen in nearshore habitats because they are either transiting (relatively briefly occurring within nearshore waters) or are in more pelagic habitats. Similarly for San Diego Bay, green sea turtles are regularly seen within the bay, but not other species. Green sea turtles are the most abundant sea turtles found in the nearshore environment of the Study Area, and in Hawaii, are observed to bask on land. Loggerheads, considered to be the most generalist of sea turtle species in terms of feeding and foraging behavior, apparently exhibit varied dive behavior that is linked to the quantity and quality of available resources. Foley et al. (2011) found that loggerheads spent 7.3 percent of time at the surface (associated with breathing), 42 percent of time under the surface but close to the surface within one body length, and 44 percent of time within the water column (the remaining time observed at or near the seafloor). Leatherback sea turtles are more likely to feed at or near the surface in open ocean areas. It is important to note that leatherbacks can forage for jellyfish at depth but bring them to the surface to ingest (Benson et al., 2007; Fossette et al., 2007; James & Herman, 2001). Basking on the water's surface is common for all species within the Study Area as a strategy to thermoregulate, and the reduced activity associated with basking may pose higher risks for sea turtle strikes because of a likely reduced capacity to avoid cues. Green, hawksbill, and loggerhead sea turtles are more likely to forage nearshore, and although they may feed along the seafloor, they surface periodically to breathe while feeding and moving between nearshore habitats.

In an attempt to determine traffic patterns for Navy and non-Navy vessels, the Center for Naval Analysis (Mintz, 2012; Mintz & Parker, 2006) conducted a review of historic data for commercial vessels, coastal shipping patterns, and Navy vessels. Within the Hawaii portion of the HCTT Study Area, significant commercial traffic is present as vessels bring shipments of goods to Hawaii as well as shipments between the islands. Trans-Pacific vessel traffic that passes through offshore waters near Hawaii are associated with transits between Asian ports and ports along the U.S. west coast or the Panama Canal. Commercial and non-Navy traffic, which included cargo vessels, bulk carriers, passenger vessels, and oil tankers (all over 20 m in length), was heaviest along the U.S. West Coast between San Diego and Seattle (Puget Sound) and between the Hawaiian Islands and the Panama Canal (Mintz & Parker, 2006). Well-defined International shipping lanes within the Study Area are also heavily traveled. Compared to coastal vessel activity, there was relatively little concentration of vessels in the other portions of the Study Area (Mintz & Parker, 2006). Vessel traffic data from 2009 shows that Navy vessels accounted for less than 10 percent of the total large vessel traffic (from estimated vessel hours) in the Study Area (Mintz, 2012). In the California Study Area where Navy vessel activity is concentrated within the exclusive economic zone, the Navy vessels accounted for 24 percent of the total large vessel traffic (Mintz, 2012).

A total of 298 sea turtle strandings were reported in the Hawaiian Islands, from all causes between 1982 and 2007. Based on an observed annual average of eight green sea turtles stranded in the Main Hawaiian Islands between 1982 and 2007, and after applying a correction factor for those that do not strand, NMFS estimated 25–50 green sea turtles are killed by vessel strike annually in the Main Hawaiian Islands (National Marine Fisheries Service, 2008a). A total of two hawksbill sea turtles were observed stranded with obvious boat strike injuries in the Main Hawaiian Islands between 1982 and 2008. The majority of strandings are likely the result of strikes with relatively small, but high speed fishing boats making thousands of trips through Hawaiian nearshore waters annually. As a term and condition for NMFS's Reinitiated Biological Opinion, the Navy prepared an analysis of all sea turtle strandings within the Hawaii Study Area (National Marine Fisheries Service, 2015). The reinitiated consultation included additional sea turtle information, in waters within Pearl Harbor and near the Pearl Harbor entrance, as well as waters surrounding Oahu in order to improve the understanding of sea turtle strikes. The vast majority of the strandings were green sea turtles (96 percent) with the remaining reported as hawksbill sea turtles or unidentifiable sea turtles. Most of these strandings were from Oahu (approximately 70 percent). Of all reported strandings, 7 percent were attributed to vessel strike, with most (34 percent) from unspecified causes and 27 percent from fisheries interactions. The remainder were attributed to disease, predation, entrapment, and natural mortality (National Marine Fisheries Service, 2015).

The frequency of vessel strike in open ocean waters surrounding Hawaii is much less clear. It is assumed if an animal is struck in waters further from shore, it is less likely to strand and be documented. There has been one recent report of a stranded turtle in Hawaii that appeared as though it may have been struck by a large propeller (such as those used by some Navy vessels) (National Marine Fisheries Service, 2008a). However, it is more likely turtles struck by large propellers would not strand because the damage to the carcass would be so extensive as to facilitate sinking or consumption by scavengers.

There is not a high level of sea turtle stranding data on the U.S. West Coast (National Marine Fisheries Service, 2008a). This does not necessarily indicate vessel strike is less common off the U.S. West Coast versus Hawaii. Ocean currents, vessel sizes, or other factors may simply affect the likelihood a struck turtle will strand. Regardless, this lack of stranding data makes estimating the frequency of sea turtle vessel strike off the U.S. West Coast difficult. Most observations of stranded sea turtles in Southern California since 1990 occurred within San Diego Bay, where a population of green sea turtles resides. Between 1990 and 2014, 10 green sea turtle strandings were observed with evidence of boat collision (National Marine Fisheries Service, 2008a). No other sea turtle species have stranded near or in the California Study Area that have had evidence of boat strike (National Marine Fisheries Service, 2008a). As a term and condition for NMFS's Reinitiated Biological Opinion, the Navy prepared an analysis of all sea turtle strandings within Southern California for 2015. Only seven strandings of sea turtles were reported in 2015. Four of these strandings were green sea turtles, two were loggerheads, and one was olive ridley. Only three sea turtles were reported as struck by vessels, all of whom were green sea turtles. These strandings were reported within San Diego Bay and were located in areas that are not used by the Navy (National Marine Fisheries Service, 2015).

Disturbance of sea turtles from vessel movements is expected to occur with more frequency than actual strikes. Visual cues from vessels nearby and vessel noise would likely induce short-term behavioral changes, such as cessation of foraging activities or moving away from the disturbance.

**In-Water Devices**

In-water devices are generally smaller (several inches to 111 ft.) than most Navy vessels. For a discussion on the types of activities that use in-water devices see Appendix B (Activity Stressor Matrices), and for information on where in-water devices are used, and how many exercises would occur under each alternative, see Section 3.0.3.3.4.1 (Vessels and In-Water Devices).

Devices that could pose a collision risk to sea turtles are those operated at high speeds and are unmanned. The Navy reviewed torpedo design features and a large number of previous anti-submarine warfare torpedo exercises to assess the potential of torpedo strikes on marine mammals, and its conclusions are also relevant to sea turtles. The acoustic homing programs of Navy torpedoes are sophisticated and would not confuse the acoustic signature of a marine mammal with a submarine/target. It is reasonable to assume that acoustic signatures of sea turtles would also not be confused with a submarine or target. All exercise torpedoes are recovered and refurbished for eventual re-use. Review of the exercise torpedo records indicates there has never been an effect on a sea turtle. In thousands of exercises in which torpedoes were fired or in-water devices used, there have been no recorded or reported instances of a marine species strike from a torpedo or any other in-water device.

Since some in-water devices are identical to support craft, (typically less than 15 m in length), sea turtles could respond to the physical presence of the device similar to how they respond to the physical presence of a vessel (see Table 3.0-17). Physical disturbance from the use of in-water devices is not expected to result in more than a momentary behavioral response. These responses would likely include avoidance behaviors (swimming away or diving) and cessation of normal activities (e.g., foraging). As with an approaching vessel, not all sea turtles would exhibit avoidance behaviors and be at higher risk of a strike.

In-water devices, such as unmanned underwater vehicles, that move slowly through the water are highly unlikely to strike sea turtles because the turtle could easily avoid the object. Towed devices are unlikely to strike a sea turtle because of the observers on the towing platform and other standard safety measures employed when towing in-water devices. Sea turtles that occur in areas that overlap with in-water device use within the Study Area may encounter in-water devices. It is possible that sea turtles may be disturbed by the presence of these activities, but any disturbance from the use of in-water devices is not expected to result in more than a temporary behavioral response.

Propulsion testing events occur infrequently but pose a higher strike risk because of the higher speeds at which the vessels need to achieve in order to complete the testing activity. These activities could occur in the Hawaii Study Area and California Study Area. However, there are just a few of these events proposed per year, so the increased risk is nominal compared to all vessel use proposed under Alternative 1.

As discussed in Section 3.0.3.3.4.1 (Vessels and In-Water Devices), testing activities involving the use of in-water devices would occur in the HCTT Study Area at any time of year. Unmanned surface vehicle use would occur within both the Hawaii Study Area and California Study Area.

As with training activities, the likelihood is low for testing activities to cause harmful interaction with a vessel or in-water device but cannot be wholly discounted. Sea turtle strikes in high vessel traffic areas (e.g., Pearl Harbor) have been reported. Potential effects of exposure to vessels may result in substantial changes in an individual's behavior, growth, survival, annual reproductive success, lifetime reproductive success (fitness), or species recruitment. Any strike at high speed is likely to result in significant injury. Potential effects of exposure to vessels are not expected to result in population-level effects for all sea turtle species. Under Alternative 1 testing activities, the Navy will continue to implement visual observation mitigation to avoid or reduce the potential for vessel and in-water device strike of sea turtles. Within a mitigation zone of a vessel or in-water device, trained observers will relay sea turtle locations to the operators, who are required to change course (no course change would be implemented if the vessel's safety is threatened, the vessel is restricted in its ability to maneuver [e.g., during launching and recovery of aircraft or landing craft, during towing activities, when mooring, etc.], or if the vessel is operated autonomously). A mitigation zone size is not specified for sea turtles to allow flexibility based on vessel type and mission requirements (e.g., small boats operating in a narrow harbor).

#### F.8.2.2 Effects from Military Expended Materials

The primary concern is the potential for a sea turtle to be struck with a military expended material at or near the water's surface, which could result in injury or death. For sea turtles, although disturbance or strike from an item as it falls through the water column is possible, it is not likely because the objects generally sink through the water slowly and can be avoided by most sea turtles. Materials will slow in their velocity as they approach the bottom of the water and will likely be avoided by any juvenile or adult sea turtles (e.g., olive ridley, green, loggerhead, or hawksbill turtles) that happen to be in the vicinity foraging in benthic habitats. Therefore, the discussion of military expended materials strikes focuses on the potential of a strike at the surface of the water.

There is a possibility that an individual turtle at or near the surface may be struck if they are in the target area at the point of physical impact at the time of non-explosive munitions delivery. Expended munitions may strike the water surface with sufficient force to cause injury or mortality. Adult sea turtles are generally at the surface for short periods, and spend most of their time submerged; however, hatchlings and juveniles spend more time at the surface while in ocean currents or at the surface while basking. The leatherback sea turtle is more likely to be foraging at or near the surface in the open ocean than other species, but the likelihood of being struck by a projectile remains very low because of the wide spatial distribution of leatherbacks relative to the point location of an activity. Furthermore, projectiles are aimed at targets, which will absorb the impact of the projectile.

While no strike from military expended materials has ever been reported or recorded on a reptile, the possibility of a strike still exists. Therefore, the potential for sea turtles to be struck by military expended materials was evaluated using statistical probability modeling to estimate potential direct strike exposures to a sea turtle. To estimate potential direct strike exposures, a worst-case scenario was calculated using the sea turtle with the highest average year-round density in areas with the highest military expended material expenditures in the Hawaii and California portions of the HCTT Study Area (see Appendix I, Military Expended Materials and Direct Strike Impact Analyses). The green sea turtle was used as a proxy for all sea turtle species because this species has the highest density estimates, which would provide the most conservative modeling output results. For estimates of expended materials in all areas, see Section 3.0.3.3.4.2 (Military Expended Materials). Input values include munitions data (frequency, footprint and type), size of the training or testing area, sea turtle density data, and size of the animal. To estimate the potential of military expended materials to strike a sea turtle, the impact area of all military expended materials was totaled over one year in the area with the highest combined amounts of military expended materials for the Proposed Action. The analysis of the potential for a sea turtle strike is influenced by the following assumptions:

- The model is two-dimensional, assumes that all sea turtles would be at or near the surface 100 percent of the time, and does not consider any time a sea turtle would be submerged.
- The model also does not take into account the fact that most of the projectiles fired during training and testing activities are fired at targets, and that most projectiles hit those targets, so only a very small portion of those would hit the water with their maximum velocity and force.
- The model assumes the animal is stationary and does not account for any movement of the sea turtle or any potential avoidance of the training or testing activity.

The potential of fragments from high-explosive munitions or expended material other than munitions to strike a sea turtle is likely lower than for the worst-case scenario calculated above because those events happen with much lower frequency. Fragments may include metallic fragments from the exploded target, as well as from the exploded munitions.

There is a possibility that an individual turtle at or near the surface may be struck if they are in the target area at the point of physical impact at the time of non-explosive munitions delivery. Expended munitions may strike the water surface with sufficient force to cause injury or mortality. Direct munitions strikes from non-explosive bombs, missiles, and rockets are potential stressors to some species. Some individuals at or near the surface may be struck directly if they are at the point of impact at the time of non-explosive practice munitions delivery. However, most missiles hit their target or are disabled before hitting the water. Thus, most of these missiles and aerial targets hit the water as fragments, which quickly dissipates their kinetic energy within a short distance of the surface.

Adult sea turtles are generally at the surface for short periods and spend most of their time submerged; however, hatchlings and juveniles of all sea turtle species spend more time at the surface while in ocean currents, and all sea turtle life stages bask on the surface. Leatherback sea turtles of all age classes are more likely to be foraging at or near the surface in the open ocean than other species, but the likelihood of being struck by a projectile remains very low because of the wide spatial distribution of leatherbacks relative to the point location of an activity. Furthermore, projectiles are aimed at targets, which will absorb the impact of the projectile. Other factors that further reduce the likelihood of a sea turtle being struck by an expended munition include the recovery of all non-explosive torpedoes as well as target-related materials that are intact after the activity. The Navy will implement mitigation (e.g., not conducting gunnery activities against a surface target when a specified distance from sea turtles) to avoid potential effects from military expended materials on sea turtles throughout the Study Area (see Section 5.6, Activity-based Mitigation).

#### F.8.2.3 Effects from Seafloor Devices

The types of activities that use seafloor devices include items placed on, dropped on, or that move along the seafloor such as mine shapes, anchor blocks, anchors, bottom-placed instruments, and bottom-crawling unmanned underwater vehicles. The likelihood of any sea turtle species encountering seafloor devices is considered low because these items are either stationary or move very slowly along the bottom. A benthic-foraging sea turtle would likely avoid the seafloor device. In the unlikely event that a sea turtle is in the vicinity of a seafloor device, the slow movement and stationary characteristics of these devices would not be expected to physically disturb or alter natural behaviors of sea turtles. Moreover, objects falling through the water column will slow in velocity as they sink toward the bottom and could be avoided by most sea turtles. Therefore, these items do not pose a significant strike risk to sea turtles. The only seafloor device used during military readiness activities that has the potential to strike a sea turtle at or near the surface is an aircraft deployed mine shape, which is used during aerial mine laying activities. These devices are identical to non-explosive practice bombs discussed above in Section F.8.2.2 (Effects from Military Expended Materials).

Based on the analysis, there is a reasonable level of certainty that no sea turtles would be struck by seafloor devices. The likelihood of a sea turtle encountering seafloor devices in benthic foraging habitats is considered low because these items are either stationary or move very slowly along the bottom. Seafloor devices are not likely to interfere with sea turtles resident to, or engaging in migratory, reproductive, and feeding behaviors within the range complexes of the Study Area. Further, seafloor devices would only affect sea turtle species that are foraging in benthic habitats (e.g., olive ridley, loggerhead, and green sea turtles). Sea turtles in coastal habitats can occur near the bottom when foraging or resting. Sea turtles encountering seafloor devices are likely to avoid them. Given the slow movement of seafloor devices, the effort expended by sea turtles to avoid them will be minimal, temporary, and not have fitness consequences.

### F.8.3 Entanglement Stressors

#### F.8.3.1 Effects from Wires and Cables

Fiber optic cables are flexible cables that can range in size up to 3,000 m in length. Longer cables present a higher likelihood of sea turtle interactions, and therefore present an increased risk of entanglement of a sea turtle. Other factors that increase risk of sea turtle interactions with fiber optic cables include the amount of time a fiber optic cable is in the same vicinity of a sea turtle; however, these cables will only be within the water column during the activity and while they sink, the likelihood of a sea turtle encountering and becoming entangled within the water column is extremely low. Fiber optic cables exhibit several physical qualities that reduce the risk of entanglement. Primarily, these cables are brittle and break easily. Because of the physical properties of fiber optic material, the cable is brittle and easily broken when kinked, twisted, or bent sharply. The cables are often designed with controlled buoyancy to minimize the cable's effect on vehicle movement. The fiber optic cable would be suspended within the water column during the activity, and then be expected to sink to the seafloor. Further, activities that use fiber optic cables occur in deep waters. These factors reduce the likelihood that a fiber optic cable would be in close proximity to a sea turtle—the cable is only buoyant during the training and testing activity, and subsequently sinks after use to rest in benthic habitats. If the isobaths is greater than the maximum benthic foraging ability (dive depth) of a sea turtle, then these cables would not present an entanglement risk. For example, as discussed previously, leatherbacks may dive to depths greater than 1,000 m in search of prey (e.g., jellyfish), while other species (e.g., loggerheads) may forage in benthic habitats as deep as approximately 200 m, and juvenile sea turtles (e.g., green sea turtles) resting and foraging in waters as deep as approximately 30 (Hochscheid, 2014; Rieth et al., 2011). In addition, because of the physical properties of the fiber optic material, the cable is unlikely to entangle a sea turtle body or appendage because the cable would likely break before an entangling loop would form. If a loop did form around an appendage or sea turtle body, the cable would subsequently break quickly on its own or in response to sea turtle movement. Therefore, fiber optic cables present an entanglement risk to sea turtles, but it is unlikely that an entanglement event would occur and any entanglement would be temporary (a few seconds) before the sea turtle could resume normal activities. As noted in Section H.6.1.3 (General Threats), entanglement by fishing gear is a serious global threat to sea turtles. The various types of marine debris attributed to sea turtle entanglement (e.g., commercial fishing gear, towed gear, stationary gear, or gillnets) have substantially higher (up to 500–2,000 lb.) breaking strengths at their "weak links." If fiber optic cables and fragments of cables sink to the seafloor in an area where the bottom is calm, they would remain there undisturbed. In an area with bottom currents or active tidal influence, the fiber optic strands may move along the seafloor, away from the location in which they were expended and potentially into sea turtle benthic foraging habitats. Over time, these strands may become covered by sediment in most areas or colonized by attaching and encrusting organisms, which would further stabilize the material and reduce the potential for reintroduction as an entanglement risk.

Similar to fiber optic cables discussed above, guidance wires may pose an entanglement threat to sea turtles either in the water column or after the wire has settled to the seafloor. The Navy previously analyzed the potential for entanglement of sea turtles by guidance wires and concluded that the potential for entanglement is low (U.S. Department of the Navy, 1996). These conclusions have also been carried forward in NMFS analyses of Navy training and testing activities (National Marine Fisheries Service, 2013). The likelihood of a sea turtle encountering and becoming entangled in a guidance wire depends on several factors. With the exception of a chance encounter with the guidance wire while it is sinking to the seafloor (at an estimated rate of 0.7 ft. per second), it is most likely that a sea turtle would only encounter a guidance wire once it had settled on the seafloor. Since the guidance wire will only be within the water column during the activity and while it sinks, the likelihood of a sea turtle encountering and becoming entangled within the water column is extremely low. Guidance wires have a relatively low tensile breaking strength; between 10 and 42 lb. and can be broken by hand (National Marine Fisheries Service, 2008a). In addition, based on degradation times, the guidance wires would break down within one to two years and therefore no longer pose an entanglement risk. As with fiber optic cables, guidance wire fragments may move with bottom currents or active tidal influence, and present an enduring entanglement risk if the wires were moved into benthic foraging habitats. Subsequent colonization by encrusting organisms, burying by sediment, and chemical breakdown of the copper filament would further reduce the potential for reintroduction as an entanglement risk. The length of the guidance wires varies, as described in Section 3.0.3.3.5.1 (Wires, Cables, and Nets), but greater lengths increase the likelihood that a sea turtle could become entangled. The behavior and feeding strategy of a species can determine whether it may encounter items on the seafloor, where guidance wires will most likely be available. There is potential for those species that feed on the seafloor to encounter guidance wires and potentially become entangled; however, the relatively few guidance wires being expended within the HCTT Study Area limits the potential for encounters.

Sonobuoys consist of a surface antenna and float unit and a subsurface hydrophone assembly unit. The two units are attached through a thin-gauge, dual-conductor, hard draw copper strand wire, which is then wrapped by a hollow rubber tubing or bungee in a spiral configuration. The tensile breaking strength of the sonobuoy wire and rubber tubing is no more than 40 lb. The length of the sonobuoy wire is housed in a plastic canister dispenser, which remains attached upon deployment. The length of cable that extends out is no more than 1,500 ft. and is dependent on the water depth and type of sonobuoy. Attached to the sonobuoy wire is a kite-drogue and damper disk stabilizing system made of non-woven nylon fabric. The nylon fabric is very thin and can be broken by hand. The sonobuoy wire runs through the stabilizing system and leads to the hydrophone components. The hydrophone components may be covered by thin plastic netting depending on type of sonobuoy. Each sonobuoy has a saltwater activated polyurethane float that inflates when the sonobuoy is submerged and keeps the sonobuoy components floating vertically in the water column below it. Sonobuoys remain suspended in the water column for no more than 30 hours, after which they sink to the seafloor. Several factors reduce the likelihood of sea turtle entanglement from sonobuoy components. The materials that present an entanglement risk in sonobuoys are weak, and if wrapped around an adult or juvenile sea turtle, would likely break soon after entanglement or break while bending into potentially entangling loops, although hatchlings would not likely be able to escape entrapment if entangled. These materials, however, are only temporarily buoyant and would begin sinking after use in an activity. The entanglement risk from these components would only occur when a sea turtle and these components were in close proximity, which is only in the water column. These materials would be expended in waters too deep for benthic foraging, so bottom foraging sea turtles would not interact with these materials once they sink. Some sonobuoy components, once they sink to the bottom, may be transported by bottom currents or active tidal influence, and present an enduring entanglement risk. In the benthic environment, subsequent colonization by encrusting organisms, burying by sediment, and chemical breakdown of the various materials would further reduce the potential for reintroduction as an entanglement risk.

Training activities under Alternative 1 would expend wires and cables throughout the Study Area.

Based on the numbers and geographic locations of their use, wires and cables used during training activities are analyzed for their potential to entangle sea turtles. Any species of sea turtle that occurs in the Study Area could at some time encounter expended cables or wires. The sink rates of cables and wires would rule out the possibility of these drifting great distances into nearshore and coastal areas where green, hawksbill, olive ridley, and loggerhead sea turtles are more likely to occur and feed on the bottom. The leatherback is more likely to co-occur with these activities, given its preference for open-ocean habitats, but this species is known to forage on jellyfish at or near the surface. Hatchlings and juveniles of some sea turtle species (e.g. greens and loggerheads), may occur in open-ocean habitats, too. Under Alternative 1, exposure to cables and wires used in training activities may cause short-term or long-term disturbance to an individual turtle because if a sea turtle were to become entangled in a cable or wire, it could free itself, or the entanglement could lead to injury or death. Potential effects of exposure to cable or wire may result in changes to an individual’s behavior, growth, survival, annual reproductive success, lifetime reproductive success (fitness), or species recruitment. However, wires and cables are generally not expected to cause disturbance to sea turtles because (1) sea turtles would only be exposed to potential entanglement risk as the wire or cable sinks through the water column; (2) due to their behavior, sea turtles are unlikely to become entangled in an object that is resting on the seafloor, and (3) there is a low concentration of expended wires and cables in the HCTT study area. Therefore, it is unlikely that an individual sea turtle would be in close proximity to a sinking wire or cable, and if so, would unlikely become entangled. Potential effects of exposure to cables and wires are not expected to result in population-level effects.

Testing activities under Alternative 1 would expend wires and cables within the Hawaii Study Area and California Study Area.

Based on the numbers and geographic locations of their use, wires and cables used during testing activities are analyzed for their potential to entangle sea turtles. Any species of sea turtle that occurs in the Study Area could at some time encounter expended cables or wires. The sink rates of cables and wires would rule out the possibility of these drifting great distances into nearshore and coastal areas where green, hawksbill, olive ridley, and loggerhead sea turtles are more likely to occur and feed on the bottom. The leatherback sea turtle is more likely to co-occur with these activities, given its preference for open-ocean habitats, but this species is known to forage on jellyfish at or near the surface. Under Alternative 1, exposure to cables and wires used in testing activities may cause short-term or long-term disturbance to an individual turtle because if a sea turtle were to become entangled in a cable or wire, it could free itself, or the entanglement could lead to injury or death. Potential effects of exposure to cable or wire may result in changes to an individual's behavior, growth, survival, annual reproductive success, lifetime reproductive success (fitness), or species recruitment. However, cables and wires are generally not expected to cause disturbance to sea turtles because of (1) the physical characteristics of the cables and wires, and (2) the behavior of the species, as sea turtles are unlikely to become entangled in an object that is resting on the seafloor. Potential effects of exposure to cables and wires are not expected to result in population-level effects.

The locations of training activities that expend wires and cables are the same under Alternatives 1 and 2. Table 3.0-24 shows the number and location of wires and cables expended during proposed training activities. Even though training activities under Alternative 2 occur at a slightly higher rate and frequency relative to Alternative 1, entanglement stress experienced by reptiles from guidance wires, fiber optic cables, and sonobuoy wires under Alternative 2 are not expected to be meaningfully different than those described under Alternative 1. The number of sonobuoys would increase under Alternative 2 training activities, thereby increasing the number of sonobuoy wires expended into the marine environment. However, wires and cables are generally not expected to cause disturbance to sea turtles because (1) sea turtles would only be exposed to potential entanglement risk as the wire or cable sinks through the water column; (2) due to their behavior, sea turtles are unlikely to become entangled in an object that is resting on the seafloor; and (3) there is a low concentration of expended wires and cables in the HCTT study area. Therefore, it is unlikely that an individual sea turtle would be in close proximity to a sinking wire or cable, and if so, would unlikely become entangled. Potential effects of exposure to cables and wires are not expected to result in population-level effects. Further, the differences in species overlap and potential effects from wires and cables on sea turtles during training activities would not be discernible from those described for training activities in Section 3.8.3.5 (Entanglement Stressors). As with Alternative 1, the use of wires and cables in training activities may cause short-term or long-term disturbance to an individual turtle, because if a sea turtle were to become entangled in a cable or wire, it could free itself or the entanglement could lead to injury or death. Potential effects of exposure to cable or wire may result in changes to an individual's behavior, growth, survival, annual reproductive success, lifetime reproductive success (fitness), or species recruitment. Potential effects of exposure to cables and wires are not expected to result in population-level effects.

The locations of testing activities that expend wires and cables are the same under Alternatives 1 and 2. Table 3.0-24 shows the number and location of wires and cables expended during proposed testing activities. Even though testing activities under Alternative 2 occur at a slightly higher rate and frequency relative to Alternative 1, entanglement stress experienced by reptiles from guidance wires, fiber optic cables, and sonobuoy wires under Alternative 2 are not expected to be meaningfully different than those described under Alternative 1. As with Alternative 1, the use of wires and cables in testing activities may cause short-term or long-term disturbance to an individual turtle, because if a sea turtle were to become entangled in a cable or wire, it could free itself or the entanglement could lead to injury or death. Potential effects of exposure to a cable or wire may result in changes to an individual's behavior, growth, survival, annual reproductive success, lifetime reproductive success (fitness), or species recruitment. Potential effects of exposure to cables and wires are not expected to result in population-level effects.

#### F.8.3.2 Effects from Decelerators/Parachutes

Section 3.0.3.3.5.2 (Decelerators/Parachutes) provides the number of training and testing exercises that involve the use of decelerators/parachutes and the geographic areas where they would be expended. Training and testing activities that introduce decelerators/parachutes into the water column can occur anywhere in the HCTT Study Area and may pose an entanglement risk to sea turtles. Potential effects from decelerators/parachutes as ingestion stressors to sea turtles are discussed in Section 3.8.3.6.1 (Effects from Military Expended Materials).

As described in Section 3.0.3.3.5.2 (Decelerators/Parachutes), decelerators/parachutes used during the proposed activities range in size from 18 in. up to 19–82 ft. in diameter. The vast majority of expended decelerators/parachutes are small (18 in.), cruciform shaped, and used with sonobuoys. Illumination flares use large decelerators/parachutes, up to 19 ft. in diameter. Drones use a larger decelerator/parachute system, ranging from 30 ft. to 82 ft. in diameter. Decelerators/parachutes have short attachment cords and upon impact with water may remain at the surface for 5–15 seconds before sinking to the seafloor, where they flatten. Sonobuoy decelerators/parachutes are designed to sink within 15 minutes, but the rate of sinking depends upon sea conditions and the shape of the decelerator/parachute, and the duration of the descent would depend on the water depth. Unlike the small- and medium-sized decelerators/parachutes, drone decelerators/parachutes do not have weights attached and may remain at the surface or suspended in the water column for some time prior to eventual settlement on the seafloor.

While in the water column, a sea turtle is less likely to become entangled because the decelerator/parachute would have to land directly on the turtle, or the turtle would have to swim into the decelerator/parachute before it sank. Prior to reaching the seafloor, it could be carried along in a current, or snagged on a hard structure near the bottom. Conversely, it could settle to the bottom, where it would be buried by sediment in most soft-bottom areas or colonized by attaching and encrusting organisms, which would further stabilize the material and reduce the potential for reintroduction as an entanglement risk. Decelerators/parachutes or decelerator/parachute lines may be a risk for sea turtles to become entangled, particularly while at the surface. A sea turtle would have to surface to breathe or grab prey from under the decelerator/parachute and swim into the decelerator/parachute or its lines.

If bottom currents are present, the canopy may billow and pose an entanglement threat to sea turtles that feed in benthic habitats (i.e., green, olive ridley, and loggerhead sea turtles). Bottom-feeding sea turtles tend to forage in nearshore areas rather than offshore, where these decelerators/parachutes are used; therefore, sea turtles are not likely to encounter decelerators/parachutes once they reach the seafloor. The potential for a sea turtle to encounter an expended decelerator/parachute at the surface or in the water column is extremely low, and is even less probable at the seafloor, given the general improbability of a sea turtle being near the deployed decelerator/parachute, as well as the general behavior of sea turtles. Depending on how quickly the decelerator/parachute may degrade, the risk may increase with time if the decelerator/parachute remains intact or if underwater currents delay settling of the decelerator/parachute on the seafloor (where they would likely be covered by sediment and encrusted). Factors that may influence degradation times include exposure to ultraviolet radiation and the extent of physical damage of the decelerator/parachute on the water's surface, as well as water temperature and sinking depth. It should be noted that no known instances of sea turtle entanglement with a decelerator/parachute assembly have been reported.

Training activities under Alternative 1 would expend decelerators/parachutes within the Hawaii Study Area, California Study Area, and the Transit Corridor. Based on the numbers and geographic locations of their use, decelerators/parachutes pose a risk of entanglement for all sea turtle species considered in this analysis. Any species of sea turtle that occurs in the Study Area could at some time encounter expended decelerator/parachute. The sink rates of a decelerator/parachute assembly would rule out the possibility of these drifting great distances into nearshore and coastal areas where green, hawksbill, olive ridley, and loggerhead sea turtles are more likely to occur and feed on the bottom. The leatherback is more likely to co-occur with these activities, given its preference for open-ocean habitats, but this species is known to forage on jellyfish at or near the surface. Early juveniles and hatchlings of other sea turtle species (e.g., green sea turtles and loggerheads) may also co-occur with these activities, as well. Under Alternative 1, exposure to decelerators/parachutes used in training activities may cause short-term or long-term disturbance to an individual turtle, because if a sea turtle were to become entangled in a decelerator/parachute, it could free itself, or the entanglement could lead to injury or death. Potential effects of exposure to decelerator/parachute may result in changes to an individual's behavior, growth, survival, annual reproductive success, lifetime reproductive success (fitness), or species recruitment. However, decelerators are generally not expected to cause disturbance to sea turtles because the decelerator/parachute would have to land directly on an animal, or an animal would have to swim into it before it sinks. Decelerators/parachutes have small footprints which further reduce the potential for entanglement. It is possible, however, that a benthic feeding sea turtle could become entangled when foraging in areas where decelerators/parachutes have settled on the seafloor. For example, if bottom currents are present, the canopy may temporarily billow and pose a greater entanglement threat.

Pursuant to the ESA, the use of decelerators/parachutes during training activities as described under Alternative 1 would have no effect on leatherback sea turtle critical habitat because activities would not occur within the designated critical habitat for this species, but may affect ESA-listed green, hawksbill, olive ridley, loggerhead, and leatherback sea turtles. The Navy has consulted with NMFS as required by section 7(a)(2) of the ESA.

Testing activities under Alternative 1 would expend decelerators/parachutes within the Hawaii Study Area and California Study Area. Based on the number of decelerators/parachutes expended under Alternative 1 testing activities, the small footprint of impact, and the low likelihood of a decelerator/parachute landing directly on a sea turtle, adverse effects on sea turtles are discountable (unlikely to occur). While entanglement is a serious stressor for sea turtles from a wide range of debris in the ocean, decelerators/parachutes used during military testing activities are an unlikely source. The leatherback is more likely to co-occur with these activities, given its preference for open-ocean habitats; this species is known to forage on jellyfish at or near the surface. Early juveniles and hatchlings of other sea turtle species (e.g., green sea turtles and loggerheads) may also co-occur with these activities, as well.

Pursuant to the ESA, the use of decelerators/parachutes in testing activities as described under Alternative 1 would have no effect on leatherback sea turtle critical habitat because activities would not occur within the designated critical habitat for this species, but may affect ESA-listed green, hawksbill, olive ridley, loggerhead, and leatherback sea turtles. The Navy has consulted with NMFS as required by section 7(a)(2) of the ESA.

### F.8.4 Ingestion Stressors

The potential effects from ingesting these materials is dependent upon the probability of the animal encountering these items in their environment, which is primarily contingent on where the items are expended and how a sea turtle feeds. Sea turtles commonly mistake debris for prey. The risk is prolific throughout sea turtle habitats, and ingestion of expended materials by sea turtles could occur in all large marine ecosystems and open ocean areas and can occur at the surface, in the water column, or at the seafloor, depending on the size and buoyancy of the expended object and the feeding behavior of the turtle. Susceptibility of sea turtles to ingestion risk is a factor of the life-stage of the individual sea turtle, foraging habits of the species, the location of the item within the water column, and the type of debris. For example, floating material could be eaten by turtles such as leatherbacks, juveniles, and hatchlings of all species that feed at or near the water surface, while materials that sink to the seafloor pose a risk to bottom-feeding turtles such as loggerheads. The variety of items eaten by juvenile and hatchling sea turtles of all species and adult leatherbacks that feed are prone to ingesting non-prey items (Fujiwara & Caswell, 2001; Hardesty & Wilcox, 2017; Mitchelmore et al., 2017; Schuyler et al., 2014; Schuyler et al., 2016).

The consequences of ingestion could range from temporary and inconsequential to long-term physical stress or even death. Ingestion of these items may not be directly lethal; however, ingestion of plastic and other fragments can restrict food intake and have sublethal effects caused by reduced nutrient intake (McCauley & Bjorndal, 1999). Poor nutrient intake can lead to decreased growth rates, depleted energy, reduced reproduction, and decreased survivorship. These long-term sublethal effects may lead to population-level effects, but this is difficult to assess because the affected individuals remain at sea and the trends may only arise after several generations have passed. Schuyler et al. (2014) determined that most sea turtles, at some point, ingest some amount of debris. Because bottom-feeding occurs in nearshore areas, materials that sink to the seafloor in the open ocean are less likely to be ingested due to their location. While these depths may be within the diving capabilities of most sea turtle species, especially leatherback sea turtles, bottom foraging species (i.e., greens, hawksbills, olive ridleys, and loggerheads) are more likely to forage in the shallower waters less than 100 m in depth. This overlaps with only a small portion of the depth range at which military materials are expended.

#### F.8.4.1 Effects from Military Expended Materials – Munitions

Many different types of explosive and non-explosive practice munitions are expended at sea during training and testing activities. Types of non-explosive practice munitions generally include projectiles, missiles, and bombs. Of these, only small- or medium-caliber projectiles would be small enough for a sea turtle to ingest. Small- and medium-caliber projectiles include all sizes up to and including 2.25 in. (57 mm) in diameter. These solid metal materials would quickly move through the water column and settle to the seafloor. Ingestion of non-explosive practice munitions is not expected to occur in the water column because the munitions sink quickly. Instead, they are most likely to be encountered by species that forage on the bottom. Types of high-explosive munitions that can result in fragments include demolition charges, projectiles, missiles, and bombs. Fragments would result from fractures in the munitions casing and would vary in size depending on the size of the net explosive weight and munitions type; however, typical sizes of fragments are unknown. These solid metal materials would quickly move through the water column and settle to the seafloor; therefore, ingestion is not expected by most species. Fragments are primarily encountered by species that forage on the bottom. Other military expended materials such as targets, large-caliber projectiles, intact training and testing bombs, guidance wires, 55-gallon drums, sonobuoy tubes, and marine markers are too large for sea turtles to consume.

Sublethal effects due to ingestion of munitions used in training and activities may cause short-term or long-term disturbance to an individual turtle because: (1) if a sea turtle were to incidentally ingest and swallow a projectile or solid metal high-explosive fragment, it could potentially disrupt its feeding behavior or digestive processes; and (2) if the item is particularly large in proportion to the turtle ingesting it, the item could become permanently encapsulated by the stomach lining, with a rare chance that this could impede the turtle's ability to feed or take in nutrients. Potential effects of exposure to munitions may result in changes to an individual's behavior, growth, survival, annual reproductive success, lifetime reproductive success (fitness), or species recruitment. In open ocean environments, munitions used in training activities are generally not expected to cause disturbance to sea turtles because: (1) sea turtles are not expected to encounter most small- and medium-caliber projectiles or high-explosive fragments on the seafloor because of the depth at which these would be expended; and (2) in some cases, a turtle would likely pass the projectile through their digestive tract and expel the item without affecting the individual. Because green, loggerhead, olive ridley, and hawksbill sea turtles feed along the seafloor, they are more likely to encounter munitions of ingestible size that settle on the bottom than leatherbacks that primarily feed at the surface and in the water column. Furthermore, these four species typically use nearshore feeding areas, while leatherbacks are more likely to feed in the open ocean. Given the very low probability of a leatherback encountering and ingesting materials on the seafloor, this analysis will focus on green, loggerhead, olive ridley, and hawksbill sea turtles and ingestible materials expended in offshore waters.

In open ocean waters and nearshore habitats, the amount of non-explosive practice munitions and high-explosive munitions fragments that an individual sea turtle would encounter is generally low based on the patchy distribution of both the projectiles and sea turtle feeding habits. In addition, a sea turtle would not likely ingest every projectile it encountered. Furthermore, a sea turtle may attempt to ingest a projectile or fragment and then reject it when it realizes it is not a food item. Therefore, potential effects of non-explosive practice munitions and fragments ingestion would be limited to the unlikely event in which a sea turtle might suffer a negative response from ingesting an item that becomes embedded in tissue or is too large to be passed through the digestive system. Sea snakes would have to mistake an item as prey, and would only be exposed in pelagic habitats, but would experience similar effects as sea turtles. The Navy considers the likelihood of ingestion of military expended materials by sea turtles or sea snakes to be very low.

The Navy will implement mitigation (e.g., not conducting gunnery activities within a specified distance of shallow-water coral reefs and precious coral beds) to avoid potential effects from military expended materials on seafloor resources in mitigation areas throughout the Study Area. This mitigation will consequently help avoid potential effects on benthic foraging sea turtles that feed on shallow-water coral reefs and precious coral beds.

#### F.8.4.2 Effects from Military Expended Materials Other Than Munitions

**Training**

As presented in Section 3.0.3.3.6 (Ingestion Stressors), military expended materials other than munitions would be expended during offshore training activities within the Hawaii Study Area and California Study Area.

Target-related material, chaff, flares, and decelerators/parachutes (and their subcomponents) have the potential to be ingested by a sea turtle, although that is considered unlikely since most of these materials would quickly drop through the water column, settle on the seafloor, or rapidly decay, and not present an ingestion hazard. Some Styrofoam, plastic endcaps, chaff, and other small items may float for some time before sinking.

While the smaller items discussed here may pose a hazard to sea turtles, as discussed for non-explosive practice munitions ingestion, the effects of ingesting these forms of expended materials on sea turtles would be minor because of the following factors:

- the limited geographic area where materials other than munitions are expended during a given event
- the limited period of time these military expended materials would remain in the water column
- the unlikely chance that a sea turtle might encounter and swallow these items on the seafloor, particularly given that many of these items would be expended over deep, offshore waters

The effects of ingesting military expended materials other than munitions would be limited to cases where an individual sea turtle might eat an indigestible item too large to be passed through the gut. The sea turtle would not be preferentially attracted to these military expended materials, with the possible exception of decelerators/parachutes that may appear similar to the prey of some sea turtle species and life stages that feed on jellyfish and similar organisms. For the most part, these military expended materials would most likely only be incidentally ingested by individuals feeding on the bottom in the precise location where these items were deposited. Non-munition military expended materials that would remain floating on the surface are too small to pose a risk of intestinal blockage to any sea turtle that happened to encounter it. Because leatherbacks and juveniles of some species (e.g., green sea turtles) are more likely to feed at or near the surface, they are more likely to encounter materials at the surface than are other species of turtles that primarily feed along the seafloor. Furthermore, leatherbacks typically feed in the open ocean, while other species are more likely to feed in nearshore areas. Though they are bottom-feeding species that generally feed nearshore, green, hawksbill, olive ridley, and loggerhead sea turtles may occur in the open ocean during migrations, as well as hatchling and juvenile stage turtles.

Sublethal effects due to ingestion of military expended materials other than munitions used in training activities may cause short-term or long-term disturbance to an individual turtle because (1) if a sea turtle were to incidentally ingest and swallow a decelerator/parachute, target fragment, chaff or flare component, it could potentially disrupt its feeding behavior or digestive processes; and (2) if the item is particularly large in proportion to the turtle ingesting it, the item could become permanently encapsulated by the stomach lining, with a rare chance that this could impede the turtle's ability to feed or take in nutrients. Potential effects of exposure to these items may result in changes to an individual's behavior, growth, survival, annual reproductive success, lifetime reproductive success (fitness), or species recruitment. However, decelerators/parachutes, target fragments, chaff, and flare components used in training activities are generally not expected to cause disturbance to sea turtles because (1) leatherbacks are likely to forage further offshore than within range complexes, and other sea turtles primarily forage on the bottom in nearshore areas; (2) in some cases, a turtle would likely pass the item through its digestive tract and expel the item without affecting the individual; and (3) chaff, if ingested, would occur in very low concentration and is similar to spicules, which sea turtles (species and life stages that consume sponges and other organisms containing spicules) ingest without harm. Potential effects of exposure to military expended materials other than munitions are not expected to result in population-level effects.

Pursuant to the ESA, the use of military expended materials other than munitions during training activities as described under Alternative 1 would have no effect on leatherback sea turtle critical habitat because activities would not occur within the designated critical habitat for this species, but may affect ESA-listed green, hawksbill, olive ridley, loggerhead, and leatherback sea turtles. The Navy has consulted with NMFS as required by section 7(a)(2) of the ESA.

**Testing**

As presented in Section 3.0.3.3.6 (Ingestion Stressors) military expended materials other than munitions would be expended during testing activities within the Hawaii Study Area and California Study Area. Target-related material, chaff, flares, and decelerators/parachutes (and their subcomponents) have the potential to be ingested by a sea turtle, although that is considered unlikely since most of these materials would quickly drop through the water column, and settle on the seafloor. Some Styrofoam, plastic endcaps, chaff, and other small items may float for some time before sinking.

While the smaller items discussed here may pose a hazard to sea turtles, as discussed for non-explosive practice munitions ingestion, the effects of ingesting these forms of expended materials on sea turtles would be minor because of the following factors:

- the limited geographic area where materials other than munitions are expended during a given event
- the limited period of time these military expended materials would remain in the water column
- the unlikely chance that a sea turtle might encounter and swallow these items on the seafloor, particularly given that many of these items would be expended over deep, offshore waters

The effects of ingesting military expended materials other than munitions would be limited to cases where an individual sea turtle might eat an indigestible item too large to be passed through the gut. The sea turtle would not be preferentially attracted to these military expended materials, with the possible exception of decelerators/parachutes that may appear similar to the prey of some sea turtle species and life stages that feed on jellyfish and similar organisms. For the most part, these military expended materials would most likely only be incidentally ingested by individuals feeding on the bottom in the precise location where these items were deposited. Non-munition military expended materials that would remain floating on the surface are too small to pose a risk of intestinal blockage to any sea turtle that happened to encounter it. Because leatherbacks and juveniles of some species (e.g., green sea turtles) are more likely to feed at or near the surface, they are more likely to encounter materials at the surface than are other species of turtles that primarily feed along the seafloor. Furthermore, leatherbacks typically feed in the open ocean, while other species are more likely to feed in nearshore areas. Though they are bottom-feeding species that generally feed nearshore, green, hawksbill, olive ridley, and loggerhead sea turtles may occur in the open ocean during migrations, as well as hatchling and juvenile stage turtles.

Pursuant to the ESA, the use of military expended materials other than munitions during testing activities as described under Alternative 1 would have no effect on leatherback sea turtle critical habitat because activities would not occur within the designated critical habitat for this species, but may affect ESA-listed green, hawksbill, olive ridley, loggerhead, and leatherback sea turtles. The Navy has consulted with NMFS as required by section 7(a)(2) of the ESA.

**Target-Related Materials**

At-sea targets are usually remotely operated airborne, surface, or subsurface traveling units, most of which are designed to be recovered for reuse. If they are severely damaged or displaced, targets may sink before they can be retrieved. Expendable targets include air-launched decoys, marine markers (smoke floats), cardboard boxes, and 10 ft. diameter red balloons tethered by a sea anchor. Most target fragments would sink quickly in the sea. Floating material, such as Styrofoam, may be lost from target boats and remain at the surface for some time, however during target recovery, personnel would collect as much floating debris and Styrofoam as possible.

**Chaff**

Chaff is an electronic countermeasure designed to reflect radar waves and obscure aircraft, vessels, and other equipment from radar tracking sources. Chaff is composed of an aluminum alloy coating on glass fibers of silicon dioxide (U.S. Air Force, 1997). It is released or dispensed in cartridges or projectiles that contain millions of chaff fibers. When deployed, a diffuse cloud of fibers undetectable to the human eye is formed. Chaff is a very light material that can remain suspended in air anywhere from 10 minutes to 10 hours and can travel considerable distances from its release point, depending on prevailing atmospheric conditions (Arfsten et al., 2002; U.S. Air Force, 1997). Doppler radar has tracked chaff plumes containing approximately 900 grams (g) of chaff drifting 200 mi. from the point of release, with the plume covering greater than 400 cubic miles (1,667 cubic kilometers) (Arfsten et al., 2002).

The chaff concentrations that sea turtles could be exposed to following release of multiple cartridges (e.g., following a single day of training) are difficult to accurately estimate because it depends on several unknown factors. First, specific release points are not recorded and tend to be random, and chaff dispersion in air depends on prevailing atmospheric conditions. After falling from the air, chaff fibers would be expected to float on the sea surface for some period, depending on wave and wind action. The fibers would be dispersed further by sea currents as they float and slowly sink toward the bottom. Chaff concentrations in benthic habitats following release of a single cartridge would be lower than the values noted in this section, based on dispersion by currents and the enormous dilution capacity of the receiving waters.

Several literature reviews and controlled experiments have indicated that chaff poses little risk, except at concentrations substantially higher than those that could reasonably occur from military training (Arfsten et al., 2002; Spargo, 1999; U.S. Air Force, 1997). Nonetheless, some sea turtle species within the Study Area could be exposed to chaff through direct body contact and ingestion. Chemical alteration of water and sediment from decomposing chaff fibers is not expected to result in exposure. Based on the dispersion characteristics of chaff, it is likely that sea turtles would occasionally come in direct contact with chaff fibers while at the water's surface and while submerged, but such contact would be inconsequential. Chaff is similar to fine human hair (U.S. Air Force, 1997). Because of the flexibility and softness of chaff, external contact would not be expected to affect most wildlife (U.S. Air Force, 1997), and the fibers would quickly wash off shortly after contact. Given the properties of chaff, skin irritation is not expected to be a problem (U.S. Air Force, 1997). Arfsten et al. (2002); Spargo (1999); U.S. Air Force

(1997) reviewed the potential effects of chaff inhalation on humans, livestock, and animals and concluded that the fibers are too large to be inhaled into the lung. The fibers are predicted to be deposited in the nose, mouth, or trachea and are either swallowed or expelled; however, these reviews did not specifically consider sea turtles.

Although chaff fibers are too small for sea turtles to confuse with prey and forage, there is some potential for chaff to be incidentally ingested along with other prey items, particularly if the chaff attaches to other floating marine debris. If ingested, chaff is not expected to affect sea turtles due to the low concentration that would be ingested and the small size of the fibers. While no similar studies to those discussed in Section 3.0.3.3.6.3 (Military Expended Materials) on the effects of chaff have been conducted on sea turtles, they are also not likely to be affected by incidental ingestion of chaff fibers. For instance, some sea turtles ingest spicules (small spines within the structure of a sponge) in the course of eating the sponges, without harm to their digestive system. Since chaff fibers are of similar composition and size as these spicules (Spargo, 1999), ingestion of chaff should be inconsequential for sea turtles.

Chaff cartridge plastic end caps and pistons would also be released into the marine environment, where they would persist for long periods and could be ingested by sea turtles while initially floating on the surface and sinking through the water column. Chaff end caps and pistons would eventually sink in saltwater to the seafloor (Spargo, 2007), which reduces the likelihood of ingestion by sea turtles at the surface or in the water column.

**Flares**

Flares are designed to burn completely. The only material that would enter the water would be a small, round, plastic compression pad or piston (0.45–4.1 g depending on flare type). The flare pads and pistons float in sea water.

An extensive literature review and controlled experiments conducted by the United States Air Force demonstrated that self-protection flare use poses little risk to the environment or animals (U.S. Air Force, 1997). For sea turtles and sea snakes, these types of flares are large enough to not be considered an ingestion hazard. Nonetheless, sea turtles within the vicinity of flares could be exposed to light generated by the flares. It is unlikely that sea turtles or sea snakes would be exposed to any chemicals that produce either flames or smoke since these components are consumed in their entirety during the burning process. Animals are unlikely to approach or get close enough to the flame to be exposed to any chemical components.

**Decelerators/Parachutes**

As noted previously in Section 3.0.3.3.5.2 (Decelerators/Parachutes), decelerators/parachutes are classified into four different categories based on size: small, medium, large, and extra-large. The majority of expended decelerators/parachutes are in the small category. Decelerators/parachutes in the three remaining size categories (medium – up to 19 ft. in diameter, large – between 30 and 50 ft. in diameter, and extra-large – up to 80 ft.in diameter) are likely too big to be mistaken for prey items and ingested by a sea turtle or sea snake.

The majority of decelerators/parachutes are weighted and by design must sink below the surface within five minutes of contact with the water. Once on the seafloor, decelerators/parachutes become flattened (Environmental Sciences Group, 2005b). Ingestion of a decelerator/parachute by a sea turtle or sea snake at the surface or within the water column would be unlikely, since the decelerator/parachute would not be available for very long before it sinks. Once on the seafloor, if bottom currents are present, the canopy may temporarily billow and be available for potential ingestion by a sea turtle feeding on or near the seafloor (sea snakes are not benthic foragers, and therefore would not be exposed to ingestion risk of decelerators/parachutes on the seafloor). Conversely, the decelerator/parachute could be buried by sediment in most soft bottom areas or colonized by attaching and encrusting organisms, which would further stabilize the material and reduce the potential for an ingestion risk. Some decelerators/parachutes may be too large to be a potential prey item for certain age classes (e.g., hatchlings and pre-recruitment juveniles), although degradation of the decelerator/parachute may create smaller items that are potentially ingestible. The majority of these items (from sonobuoys), however, would be expended in deep offshore waters. Bottom-feeding sea turtles (e.g., green, hawksbill, olive ridley, and loggerhead turtles) tend to forage in nearshore and coastal areas rather than offshore, where the majority of these decelerators/parachutes are used. Since these materials would most likely be expended in offshore waters too deep for benthic foraging, it would be unlikely for bottom-foraging sea turtles to interact with these materials once they sink; therefore, it is unlikely that sea turtles would encounter decelerators/parachutes once they reach the seafloor.

### F.8.5 Secondary Stressors

This section analyzes potential effects on reptiles exposed to stressors indirectly through effects on habitat and prey availability.

**Explosives**

Underwater explosions could affect other species in the food web, including sea turtle and sea snake prey species, and could disrupt ecological relationships and conditions that would lead to decreased availability of forage. The effects of explosions would differ depending on the type of prey species in the area of the blast. As described in Chapter 2 (Description of Proposed Action and Alternatives), Table 2.6-1 through Table 2.6-5, training and testing activities resulting in underwater explosions will occur in the Study Area.

In addition to the physical effects of an underwater blast (e.g., injury or mortality from the blast pressure wave), prey might have behavioral reactions to underwater sound. For instance, prey might exhibit a strong startle reaction to detonations that might include swimming to the surface or scattering away from the source. This startle and flight response is the most common secondary defense among animals (Mather, 2004). The abundance of prey near the detonation point could be diminished for a short period before being repopulated by animals from adjacent waters (Berglind et al., 2009; Craig, 2001). Many sea turtle prey items, such as jellyfish, sponges, and molluscs, have limited mobility and ability to react to pressure waves; therefore, mobile prey species for sea turtles and sea snakes would be less affected because of their ability to respond to other stressors preceding an underwater blast (e.g., vessel noise or visual cues). Any of these scenarios would be temporary, only occurring during activities involving explosives, and no lasting effect on prey availability or the pelagic food web would be expected. For example, if prey were removed from an area resulting from a stressor introduced by a training or testing activity, prey species would be expected to return to or recolonize rapidly in the area because there would be little or no permanent change to the habitat.

The Navy will implement mitigation (e.g., not conducting gunnery activities within a specified distance of shallow-water coral reefs) to avoid potential effects from explosives and physical disturbance and strike stressors on seafloor resources in mitigation areas throughout the Study Area. This mitigation will consequently help avoid potential effects from explosives on sea turtle and sea snake prey species that inhabit shallow-water coral reefs, live hard bottom, precious coral beds, artificial reefs, and shipwrecks.

**Explosion Byproducts and Unexploded Munitions**

High-order explosions consume most of the explosive material, creating typical combustion products. In the case of Royal Demolition Explosive, also known as cyclonite and hexogen, 98 percent of the byproducts are common seawater constituents, and the remainder is rapidly diluted below threshold effect level (Section 3.2, Sediments and Water Quality). Explosion byproducts associated with high-order detonations present no indirect stressors to sea turtles or sea snakes through sediment or water. Furthermore, most explosions occur in depths exceeding those which normally support seagrass beds and coral reefs, areas that are commonly used by green and hawksbill sea turtles. For example, most detonations would occur in waters greater than 200 ft. in depth, and greater than 3 NM from shore, although mine warfare, demolition, and some testing detonations would occur in shallow water close to shore. These low-order detonations and unexploded munitions present elevated likelihood of secondary effects on sea turtles. For sea snakes, deep diving to these depths is not likely, and they would not be exposed to indirect stressors from explosion byproducts or unexploded munitions.

Deposition of undetonated explosive materials into the marine environment can be reasonably well estimated by the known failure and low-order detonation rates of high explosives (Section 3.2, Sediments and Water Quality, Table 3.2-10). While it is remotely possible for sea turtles to come into contact with an undetonated explosive, to have contact with unexploded materials in the sediment or water, and or to ingest unexploded materials in sediments, it is very unlikely. For sea snakes, benthic foraging in pelagic environments is unlikely to occur, and interactions with undetonated explosives is highly unlikely.

Indirect effects by explosives and unexploded munitions to sea turtles via sediment contamination would only be possible only if a sea turtle ingested the sediment. Degradation of explosives proceeds through several pathways, as discussed in Section 3.2.3.1 (Explosives Stressors). Degradation products of Royal Demolition Explosive are not toxic to marine organisms at realistic exposure levels (Rosen & Lotufo, 2010). Relatively low solubility of most explosives and their degradation products means that concentrations of these contaminants in the marine environment are relatively low and readily diluted. Furthermore, while explosives and their degradation products were detectable in marine sediment approximately 6–12 in. away from degrading munitions, the concentrations of these compounds were not statistically distinguishable from background beyond 3–6 ft. from the degrading munitions (Section 3.2.3.1, Explosives Stressors). Taken together, it is possible that sea turtles could be exposed to degrading explosives, but it would be within a very small radius of the explosive (1 to 6 ft.). Sea snakes, with shallow water pelagic habits, would not likely interact with sediments.

A series of research efforts focused on World War II underwater munitions disposal sites in Hawaii (Edwards et al., 2016; Kelley et al., 2016; Koide et al., 2016; University of Hawaii, 2010) and an intensively used live fire range in the Mariana Islands (Smith & Marx, 2016) provide information in regard to the effects of undetonated materials and unexploded munitions on marine life. Section 3.2.3.1 (Explosives and Explosives Byproducts) and Section 3.2.3.2 (Metals) contains a summary of this literature that investigated water and sediment quality effects, on a localized scale, from munitions ocean disposal sites and ocean disposed dredge spoils sites. Findings from these studies indicate that there were no adverse effects on the local ecology from the presence of degrading munitions and there was no bioaccumulation of munitions-related chemicals in local marine species.

The island of Farallon De Medinilla (in the Mariana Islands) has been used as a target area since 1971. Between 1997 and 2012, there were 14 underwater scientific survey investigations around the island providing a long-term look at potential effects on the marine life from training and testing involving the use of munitions (Smith & Marx, 2016). Munitions use has included high-explosive rounds from gunfire, high-explosives bombs by Navy aircraft and U.S. Air Force B-52s, in addition to the expenditure of inert rounds and non-explosive practice bombs. Marine life assessed during these surveys included algae, corals, benthic invertebrates, sharks, rays, and bony fishes, and sea turtles. The investigators found no evidence over the 16-year period that the condition of the biological resources had been adversely affected to a significant degree by the training activities (Smith & Marx, 2016). Furthermore, they found that the health, abundance, and biomass of fishes, corals and other marine resources were comparable to or superior to those in similar habitats at other locations within the Mariana Archipelago.

These findings are consistent with other assessments such as that done for the Potomac River Test Range at Dahlgren, Virginia, which was established in 1918 and is the nation's largest fully instrumented, over-the-water gun-firing range. Munitions tested at Naval Surface Warfare Center, Dahlgren have included rounds from small-caliber guns up to the Navy's largest (16 in. guns), bombs, rockets, mortars, grenades, mines, depth charges, and torpedoes (U.S. Department of the Navy, 2013f). Results from the assessment indicate that munitions expended at Naval Surface Warfare Center, Dahlgren have not contributed to significant concentrations of metals to the Potomac River water and sediments, given those contributions are orders of magnitude lower than concentrations already present in the Potomac River from natural and manmade sources (U.S. Department of the Navy, 2013f).

The concentration of munitions/explosions, expended material, or devices in any one location in the HCTT Study Area would be a small fraction of that from a World War II dump site, or a target island used for 45 years, or a water range in a river used for almost 100 years. Based on findings from much more intensively used locations, the water quality effects from the use of munitions, expended material, or devices resulting from any of the proposed actions would be negligible by comparison. As a result, explosion by-products and unexploded munitions would have no meaningful effect on water quality and would therefore not constitute a secondary indirect stressor for sea turtles or sea snakes.

**Metals**

Metals are introduced into seawater and sediments as a result of training and testing activities involving ship hulks, targets, munitions, and other military expended materials (see Section 3.2.3.2, Metals) (Environmental Sciences Group, 2005b). Some metals bioaccumulate and physiological effects begin to occur only after several trophic transfers concentrate the toxic metals (Section 3.2 Sediments and Water Quality; and Chapter 4, Cumulative Effects). Evidence from a number of studies (Briggs et al., 2016; Edwards et al., 2016; Kelley et al., 2016; Koide et al., 2016) indicate metal contamination is very localized and that bioaccumulation resulting from munitions cannot be demonstrated. Specifically, in sampled marine life living on or around munitions on the seafloor, metal concentrations could not be definitively linked to the munitions since comparison of metals in sediment next to munitions show relatively little difference in comparison to other "clean" marine sediments used as a control/reference (Koide et al., 2016). Research has demonstrated that some smaller marine organisms are attracted to metal munitions as a hard substrate for colonization or as shelter (Kelley et al., 2015) (Smith & Marx, 2016). Although this would likely increase prey availability for some benthic foraging sea turtles that feed on molluscs (e.g., loggerheads), the relatively low density of metals deposited by training and testing activities compared to concentrated dump and range sites would not likely substantively benefit sea turtles. As stated above, pelagic habits and shallow water diving would not likely present any opportunities for sea snake interactions with metal contaminated sediments.

**Chemicals**

Several Navy training and testing activities introduce chemicals into the marine environment that are potentially harmful in higher concentrations; however, rapid dilution would occur, and toxic concentrations are unlikely to be encountered. Chemicals introduced are principally from flares and propellants for missiles and torpedoes. Properly functioning flares, missiles, and torpedoes combust most of their propellants, leaving benign or readily diluted soluble combustion byproducts (e.g., hydrogen cyanide). Operational failures may allow propellants and their degradation products to be released into the marine environment. Flares and missile that operationally fail may release perchlorate, which is highly soluble in water, persistent, and affects metabolic processes in many plants and animals if in sufficient concentration. Such concentrations are not likely to persist in the ocean. Research has demonstrated that perchlorate did not bioconcentrate or bioaccumulate, which was consistent with the expectations for a water-soluble compound (Furin et al., 2013). Perchlorate from failed expendable items is therefore unlikely to compromise water quality to that point that it would act as a secondary stressor to sea turtles. It should also be noted that chemicals in the marine environment as a result of Navy training and testing activities would not occur in isolation and are typically associated with military expended materials that release the chemicals while in operation. Because sea turtles and sea snakes would likely avoid expended flares, missiles, or torpedoes in the water (because of other cues such as visual and noise disturbance), avoidance would further reduce the potential for introduced chemicals to act as a secondary stressor.

#### F.8.5.1 Effects on Habitat

As presented in Section F.7.5 (Secondary Stressors), Navy activities that introduce explosive byproducts and unexploded munitions, metals, and chemicals into the marine environment have not demonstrated long-term effects on sediment and water quality. Explosive byproducts and unexploded munitions from ongoing Navy activities have not resulted in water quality effects, and the likelihood of sea turtles or sea snakes being in contact with sediments contaminated from degrading explosives is low, given the small radius of impact around the location of the explosive. Furthermore, there is no evidence of bioconcentration or bioaccumulation of chemicals introduced by Navy activities that would alter water quality to an extent that would result in overall habitat degradation for sea turtles or sea snakes.

As stated previously, most detonations would occur in waters greater than 200 ft. in depth, and greater than 3 NM from shore, although mine warfare, demolition, and some testing detonations would occur in shallow water close to shore. In deep waters, explosions would not likely affect habitat for sea turtles or sea snakes because the explosion would not be on or proximate to the sea floor. In nearshore waters, explosions would typically occur in the same locations, limiting the removal of habitat to previously disturbed areas. Therefore, habitat loss from training and testing activities that use explosions would not substantially remove habitats available to sea turtles and sea snakes and not affect sea turtle or sea snake individuals or populations.

Pursuant to the ESA, training and testing activities that introduce secondary stressors would have no effect on leatherback sea turtle critical habitat because activities would not occur within the designated critical habitat for this species, but may affect ESA-listed green, hawksbill, olive ridley, loggerhead, and leatherback sea turtles through minor and localized indirect effects on these species' habitat. The Navy has consulted with NMFS as required by section 7(a)(2) of the ESA.

#### F.8.5.2 Effects on Prey Availability

As presented above in Section F.7.5 (Secondary Stressors), Navy activities that introduce explosives, metals, and chemicals into the marine environment have not demonstrated long-term effects on prey availability for sea turtles or sea snakes. Bioaccumulation of metals from munitions in prey species has not been demonstrated, and no effects to prey availability from metals and chemicals are known to occur.

Training and testing activities in the HCTT Study Area would be unlikely to affect coral reefs (a direct and indirect source of prey and forage items for sea turtles) because the Navy implements mitigation for shallow-water coral reefs. These mitigation measures would continue under both Alternative 1 and Alternative 2.

Pursuant to the ESA, training and testing activities that introduce secondary stressors would have no effect on leatherback sea turtle critical habitat because activities would not occur within the designated critical habitat for this species but may affect ESA-listed sea turtles through minor and localized indirect effects on these species' prey availability. The Navy has consulted with NMFS as required by section 7(a)(2) of the ESA.

#### F.8.5.3 Summary of Potential Effects on Reptiles

**Additive Stressors–** There are generally two ways that a sea turtle or sea snake could be exposed to multiple additive stressors. The first would be if an animal were exposed to multiple sources of stress from a single event or activity (e.g., a mine warfare event may include the use of a sound source and a vessel).

The potential for a combination of these effects from a single activity would depend on the range to effects of each of the stressors and the response or lack of response to that stressor. Most of the activities proposed under Alternative 1 generally involve the use of moving platforms (e.g., ships, torpedoes, aircraft) that may produce one or more stressors; therefore, it is likely that if a sea turtle or sea snake were within the potential effects range of those activities, it may be affected by multiple stressors simultaneously. Individual stressors that would otherwise have minimal to no effect, may combine to cause a response. However, due to the wide dispersion of stressors, speed of the platforms, general dynamic movement of many training and testing activities, and behavioral avoidance exhibited by sea turtles, it is very unlikely that a sea turtle or sea snake would remain in the potential effects range of multiple sources or sequential exercises. Exposure to multiple stressors is more likely to occur at an instrumented range where training and testing using multiple platforms may be concentrated during a particular event. In such cases involving a relatively small area on an instrumented range, a behavioral reaction resulting in avoidance of the immediate vicinity of the activity would reduce the likelihood of exposure to additional stressors. Nevertheless, the majority of the proposed activities are unit-level training and small testing activities which are conducted in the open ocean. Unit level exercises occur over a small spatial scale (one to a few square miles) and with few participants (usually one or two vessels) or short duration (the order of a few hours or less).

Secondly, a sea turtle or sea snake could be exposed to multiple training and testing activities over the course of its life, however, training and testing activities are generally separated in space and time in such a way that it would be unlikely that any individual sea turtle or sea snake would be exposed to stressors from multiple activities within a short timeframe. However, sea turtles with a home range intersecting an area of concentrated Navy activity have elevated exposure risks relative to sea turtles that simply transit the area through a migratory corridor. Sea snakes in open ocean environments within the Study Area are more associated with currents without home ranges in pelagic areas; therefore, activities concentrated in repeated geographic locations would not present a risk to pelagic roaming sea snakes. This limited potential for exposure of individuals is not anticipated to affect populations.

**Synergistic Stressors–**Multiple stressors may also have synergistic effects on sea turtles. Assumed to rarely occur in the Study Area, and not occurring within groups, sea snakes would likely not experience synergistic effects. Sea turtles that react to a sound source (behavioral response) or experience injury from acoustic stressors could be more susceptible to physical strike and disturbance stressors via a decreased ability to detect and avoid threats. Sea turtles that experience behavioral and physiological consequences of ingestion stressors could be more susceptible to entanglement and physical strike stressors via malnourishment and disorientation. Similarly, sea turtles that may be weakened by disease (e.g., fibropapillomatosis) or other factors that are not associated with Navy training and testing activities may be more susceptible to stressors analyzed in this EIS. These interactions are speculative, and without data on the combination of multiple Navy stressors, the synergistic effects from the combination of Navy stressors are difficult to predict in any meaningful way. Research and monitoring efforts have included before, during, and after-event observations and surveys, data collection through conducting long-term studies in areas of Navy activity, occurrence surveys over large geographic areas, biopsy of animals occurring in areas of Navy activity, and tagging studies where animals are exposed to Navy stressors. These efforts are intended to contribute to the overall understanding of what effects may be occurring overall to animals in these areas. To date, the findings from the research and monitoring and the regulatory conclusions from previous analyses by NMFS (National Oceanic and Atmospheric Administration, 2013; National Oceanic and Atmospheric Administration, 2015) are that majority of effects from Navy training and testing activities are not expected to have deleterious effects on the fitness of any individuals or long-term consequences to populations of sea turtles.

Although potential effects on certain sea turtle species from training and testing activities under Alternative 1 may include injury to individuals, those injuries are not expected to lead to consequences for populations. The potential effects anticipated from Alternative 1 are summarized in Sections 3.8.4 (Endangered Species Act Determinations). For a discussion of cumulative effects, see Chapter 4 (Cumulative Effects).

## F.9 Birds

### F.9.1 Acoustic Stressors

#### F.9.1.1 Background

The sections below include a survey and synthesis of best-available-science published in peer-reviewed journals, technical reports, and other scientific sources pertinent to effects on birds potentially resulting from sound-producing Navy training and testing activities. Effects on birds depends on the sound source and context of exposure. Possible effects include auditory or non-auditory trauma, hearing loss resulting in temporary or permanent hearing threshold shift, auditory masking, physiological stress, or changes in behavior, including changing habitat use and activity patterns, increasing stress response, decreasing immune response, reducing reproductive success, increasing predation risk, and degrading communication (Larkin et al., 1996). Numerous studies have documented that birds and other wild animals respond to human-made noise (Bowles et al., 1994; Larkin et al., 1996; National Park Service, 1994). The manner in which birds respond to noise could depend on species physiology life stage, characteristics of the noise source, loudness, onset rate, distance from the noise source, presence/absence of associated visual stimuli, and previous exposure. Noise may cause physiological or behavioral responses that reduce the animals' fitness or ability to grow, survive, and reproduce successfully.

The types of birds exposed to sound-producing activities depend on where training and testing activities occur. Birds in the study area can be divided into three groups based on breeding and foraging habitat: (1) those species such as albatrosses, petrels, frigatebirds, tropicbirds, boobies, alcids, and some terns that forage over the ocean and nest on oceanic islands; (2) species such as pelicans, cormorants, gulls, and some terns that nest along the coast and forage in nearshore areas; and (3) those few species such as skuas, jaegers, Franklin's gull, Bonaparte's gulls, ring-billed gulls, black terns, and ducks and loons that nest and forage in inland habitats and come to the coastal areas during nonbreeding seasons. In addition, birds that are typically found inland, such as songbirds, may be present flying in large numbers over open ocean areas during annual spring and fall migration periods.

Birds could be exposed to sounds from a variety of sources. While above the water surface, birds may be exposed to airborne sources such as pile driving, weapons noise, vessel noise, and aircraft noise. While foraging and diving, birds may be exposed to underwater sources such as sonar, pile driving, air guns, and vessel noise. While foraging birds will be present near the water surface, migrating birds may fly at various altitudes. Some species such as sea ducks and loons may be commonly seen flying just above the water's surface, but the same species can also be spotted flying high enough (5,800 ft.) that they are barely visible through binoculars (Lincoln et al., 1998). While there is considerable variation, the favored altitude for most small birds appears to be between 500 ft. (152 m) and 1,000 ft. (305 m). Radar studies have demonstrated that 95 percent of the migratory movements occur at less than 10,000 ft. (3,050 m), with the bulk of the movements occurring under 3,000 ft. (914 m) (Lincoln et al., 1998).

Seabirds use a variety of foraging behaviors that could expose them to underwater sound. Most seabirds plunge-dive from the air into the water or perform aerial dipping (the act of taking food from the water surface in flight); others surface-dip (swimming and then dipping to pick up items below the surface) or jump-plunge (swimming, then jumping upward and diving underwater). Birds that feed at the surface by surface or aerial dipping with limited to no underwater exposure include petrels, jaegers, and phalaropes. Birds that plunge-dive are typically submerged for short durations, and any exposure to underwater sound would be very brief. Birds that plunge-dive include albatrosses, some tern species, masked boobies, gannets, shearwaters, and tropicbirds. Some birds, such as cormorants, seaducks, alcids, and loons pursue prey under the surface, swimming deeper and staying underwater longer than other plunge-divers. Some of these birds may stay underwater for up to several minutes and reach depths between 50 ft. (15 m) and 550 ft. (168 m) (Alderfer, 2003; Durant et al., 2003; Jones, 2001; Lin, 2002; Ronconi, 2001). Birds that forage near the surface would be exposed to underwater sound for shorter periods of time than those that forage below the surface. Exposures of birds that forage below the surface may be reduced by destructive interference of reflected sound waves near the water surface (see Appendix D, Acoustic and Explosive Concepts). Sounds generated underwater during training and testing would be more likely to affect birds that pursue prey under the surface, although as previously stated, little is known about seabird hearing ability underwater.

#### F.9.1.1.1 Injury

Auditory structures can be susceptible to direct mechanical injury due to 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 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. There are no data on damage to the middle ear structures of birds due to acoustic exposures. Because birds are known to regenerate auditory hair cells, studies have been conducted to purposely expose birds to very high sound exposure levels (SELs) in order 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 below F.8.1.1.2 (Hearing Loss).

Because there is 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 changes near non-explosive impulsive underwater 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 [see Popper et al. (2014)]. 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. Proximal 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 3.8.3.2, Explosive Stressors). Unlike in water, most acoustic energy will reflect off the surface of an animal's body in air. Plus, 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.

#### F.9.1.1.2 Hearing Loss

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 exposure frequency, received sound pressure level (SPL), temporal pattern, and duration. Hearing loss could impair a bird's ability to hear biologically important sounds within the affected frequency range. Biologically important sounds come from social groups, potential mates, offspring, or parents; environmental sounds; prey; or predators.

Because in-air measures of hearing loss and recovery in birds due to an acoustic exposure are limited [e.g., quail, budgerigars, canaries, and zebra finches (Ryals et al., 1999); budgerigar (Hashino et al., 1988); parakeet (Saunders & Dooling, 1974); quail (Niemiec et al., 1994)] and no studies exist of bird hearing loss due to underwater sound exposures, auditory threshold shift in birds is considered to be consistent with general knowledge about noise-induced hearing loss described in the Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities (see Section F.1.1). The frequencies affected by hearing loss would vary depending on the exposure frequency. The limited data on hearing loss in birds shows that the frequency of exposure is the hearing frequency most likely to be affected (Saunders & Dooling, 1974).

Hearing loss can be due to biochemical (fatiguing) processes or 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). Unlike many other animals, birds have the ability to 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).

Hearing loss is typically quantified in terms of threshold shift—the amount (in dB) that hearing thresholds at one or more specified frequencies are elevated, compared to their pre-exposure values, at some specific time after the noise exposure. The amount of threshold shift measured usually decreases with increasing recovery time — the amount of time that has elapsed since a noise exposure. If the threshold shift eventually returns to zero (i.e., the hearing threshold returns to the pre-exposure value), the threshold shift is called a TTS. If the threshold shift does not completely recover (the threshold remains elevated compared to the pre-exposure value), the remaining threshold shift is called a PTS. By definition, TTS is a function of the recovery time, therefore comparing the severity of noise exposures based on the amount of induced TTS can only be done if the recovery times are also considered. For example, a 20 dB TTS measured 24 hours post-exposure indicates a more hazardous exposure than one producing 20 dB of TTS measured only two minutes after exposure; if the TTS is 20 dB after 24 hours, the TTS measured after two minutes would have likely been much higher. Conversely, if 20 dB of TTS was measured after two minutes, the TTS measured after 24 hours would likely have been much smaller.

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 in birds.

**Hearing Loss due to Non-Impulsive Sounds**

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). Although not directly comparable, this SPL would be perceived as extremely loud but just under the threshold of pain for humans per the American Speech-Language-Hearing Association. 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 µPa$^2$-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 threshold shift 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 [see (Southall et al., 2007)]. 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– 50 dB after 12-hour exposures to 112 dB and 118 dB re 20 µPa SPL tones at 2.86 kHz (158 – 164 dB re 20 µPa$^2$-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.

**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, the few studies of hearing loss in birds due to exposures to non-impulsive sound are the only other avian data upon which to assess bird 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.

#### F.9.1.1.3 Masking

Masking occurs when one sound, distinguished as the 'noise,' 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 the Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities (Section F.1.1), 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.

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 $\mu Pa^2$ /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., Dooling (1980), Noirot et al. (2011), Dooling and Popper (2000), and Crowell (2016)] and inter-species 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, increase 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.

#### F.9.1.1.4 Physiological Stress

Animals in the marine environment 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. Anthropogenic sound-producing activities have the potential to provide additional stressors beyond those that naturally occur, as described in the Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities (see Section F.1.1).

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 Navy conservatively assumes 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.

#### F.9.1.1.5 Behavioral Reactions

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 manner in which 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 F.1.1, 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). 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).

**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 effects 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 Behavioral Reactions to Aircraft.

**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.

**Behavioral Reactions to Aircraft**

There are multiple possible factors involved in behavioral responses 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 would 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 sound level exposures greater than 85 A-weighted decibels (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 101–116 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 ft. (500 m) distance from raptors were observed to elicit no response (Ellis, 1981). The effects of low-altitude military training flights on wading bird colonies in Florida were estimated using colony distributions and turnover rates. There were no demonstrated effects 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.

#### F.9.1.1.6 Long Term Consequences

Long term consequences to birds due to acoustic exposures are considered following the Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities (see Section F.1.1).

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 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, although intermittent exposures are assumed to be less likely than prolonged exposures to have lasting consequences.

### F.9.1.2 Effects from Pile Driving

Noise from the installation and removal of piles has a potential to affect animals in the vicinity of the training event. Impact pile driving creates repetitive impulsive sound. An impact pile driver generally operates in the range of 36–50 blows per minute. Vibratory pile extraction creates a nearly continuous sound made up of a series of short duration rapid impulses at a much lower source level than impact pile driving. The sounds are emitted both in the air and in the water in nearshore areas where some birds forage. It is expected that most birds would exhibit avoidance behavior and leave the pile driving location. However, if prey species such as fish are killed or injured as a result of pile driving, some birds may continue to forage close to the construction area, or may be attracted to the area, and be exposed to associated noise. Behavioral responses and displacement from the area are expected to be temporary for the duration of the pile driving and extraction activities.

Impulses from the impact hammer are broadband and carry most of their energy in the lower frequencies. The underwater SPLs produced by impact pile driving during Navy activities are below the conservatively estimated injury thresholds recommended for other small animals with similar sized air cavities (sea turtles and fish; see Popper et al. (2014)). Therefore, the risk of barotrauma to any diving birds is negligible. Impulses from the impact hammer attenuate more quickly in air than in water and birds are likely to avoid the area during impact driving. Therefore, the risk of barotrauma to birds in air or at the water surface is negligible.

Pursuit divers may remain underwater for minutes, increasing the chance of underwater sound exposure. However, the short duration of driving or extracting a single pile would limit the likelihood of exposure, especially since a bird that is disturbed by pile driving while underwater may respond by swimming to the surface. Although it is not known what duration or intensity of underwater sound exposure would put a bird at risk of hearing loss, birds are less susceptible to both temporary and PTS than mammals (Saunders & Dooling, 1974). Diving birds have adaptations to protect the middle ear and tympanum from pressure changes during diving that may affect hearing (Dooling & Therrien, 2012). While some adaptions may exist to aid in underwater hearing, other adaptations to protect in-air hearing may limit aspects of underwater hearing (Hetherington, 2008). Because of these reasons, the likelihood of a diving bird experiencing an underwater exposure to impact pile driving that could affect hearing is considered low. Vibratory pile extraction sound levels are low and are not considered to pose a risk to bird hearing in air or in water.

Because diving birds may rely more on vision for foraging, there is no evidence that diving birds rely on underwater acoustic communication for foraging, and individual pile driving and extraction occurs only over a few minutes, the masking of important acoustic signals underwater by pile driving is unlikely. The potential for masking of calls in air would also likely be limited because of the short duration of individual pile driving and extraction and the likelihood that birds would avoid the area around pile driving activities.

Responses by birds to noise from pile driving would be short-term behavioral or physiological responses (e.g., alert response, startle response, and temporary increase in heart rate). Startle or alert reactions are not likely to disrupt major behavior patterns, such as migrating, breeding, feeding, and sheltering, or to result in serious injury to any birds. Some birds may be attracted to the area to forage for prey species killed or injured as a result of pile driving and be exposed to noise from pile driving temporarily. Birds may be temporarily displaced and there may be temporary increases in stress levels; however, behavior and use of habitat would return shortly after the training is complete.

## F.9.2 Explosive Stressors

### F.9.2.1 Effects from Explosives

#### F.9.2.1.1 Injury

If a bird is close to an explosive detonation, the exposure to high pressure levels and sound impulse can cause barotrauma. Barotrauma is physical injury due to a difference in pressure between an air space inside the body and the surrounding air or water. Sudden very high pressures can also cause damage at tissue interfaces due to the way pressure waves travel differently through tissues with different material properties. Damage could also occur to the structure of the ear, considered to be the body part most susceptible to pressure damage.

Detonations that occur underwater could injure, kill, or disturb diving birds, particularly pursuit divers that spend more time underwater than other foraging birds (Danil & St Leger, 2011). Studies show that birds are more susceptible to underwater explosions when they are submerged versus partially submerged on the surface. Two species of duck were exposed to explosive blasts while submerged 0.61 m and while sitting on the water surface. Onset of mortality ($LD_1$) was predicted to occur at an impulse exposure of 248 pascal seconds (Pa-s) (36 pounds per square inch per millisecond [psi-ms]) for birds underwater and 690 Pa-s (100 psi-ms) for birds at the water surface (Yelverton & Richmond, 1981). No injuries would be expected for birds underwater at blast pressures below 41 Pa-s (6 psi-ms) and for birds on the surface at blast pressures below 207 Pa-s (30 psi-ms) (Yelverton & Richmond, 1981). Tests of underwater explosive exposures to other taxa (fish, mammals) have shown that susceptibility to injury is related to animal mass, with smaller animals being more susceptible to injury (Yelverton & Richmond, 1981). It is reasonable to assume that this relationship would apply to birds as well. The range to these thresholds would be based on several factors including charge size, depth of the detonation, and how far the bird is beneath the water surface.

Detonations in air or at the water surface could also injure birds while either in flight or at the water surface. Experiments that exposed small, medium, and large birds to blast waves in air were conducted to determine the exposure levels that would be injurious (Damon et al., 1974). Birds were assessed for internal injuries to air sacs, organs, and vasculature, as well as injury to the auditory tympanum, but internal auditory damage was not assessed. Results indicated that peak pressure exposure of 5 psi would be expected to produce no blast injuries, 10 psi would produce slight to extensive injuries, and 20 psi would produce 50 percent mortality. These results also suggested that birds with higher mass may be less susceptible to injury. In addition to the risk of direct blast injury, exposure to an explosion in air may cause physical displacement of a bird that could be injurious. The same study examined displacement injuries to birds (Damon et al., 1974). Results indicated that impulse exposures below 5 psi-ms would not be expected to result in injuries.

One experiment was conducted with birds in flight, showing how birds can withstand relatively close exposures to in-air explosions (Damon et al., 1974). Flying pigeons were exposed to a 64-pound (lb.) net explosive weight explosion. Birds at 44–126 ft. from the blast exhibited no signs of injury, while serious injuries were sustained at ranges less than 40 ft. The no injury zone in this experiment was also for exposures less than 5 psi-ms impulse, similar to the results of the displacement injury study.

Ranges to the no injury threshold for a range of in-air explosives are shown in Table F-7.

**Table F-7: Range to No Blast Injury for Birds Exposed to Aerial Explosives**

| ***Net explosive weight*** | ***Range to 5 psi*** |
|---|---|
| 5 pounds (lb.) | 21 feet (ft.) |
| 10 lb. | 26 ft. |
| 100 lb. | 57 ft. |

Note: Ranges calculated using the methods in U.S. Department of the Navy (1975).

Another risk of explosions in air is exposure to explosive fragmentation, in which pieces of the casing of a cased explosive are ejected at supersonic speeds from the explosion. The risk of direct strike by fragmentation would decrease exponentially with distance from the explosion, as the worst case for strike at any distance is the surface area of the casing fragments, which ultimately would decrease their outward velocity under the influence of drag. It is reasonable to assume that a direct strike in air or at the water surface would be mortal. Once in water, the drag on any fragments would quickly reduce their velocity to non-hazardous levels (Swisdak & Montanaro, 1992).

The initial detonation in a series of detonations may deter birds from subsequent exposures via an avoidance response, however, birds have been observed taking interest in surface objects related to detonation events and subsequently being killed by a following detonation [Stemp, R. in Greene et al. (1985)].

#### F.9.2.1.2 Hearing Loss

Exposure to intense sound may result in hearing loss which persists after cessation of the noise exposure. There are no data on hearing loss in birds specifically due to explosives; therefore, the limited data on hearing loss due to impulsive sounds, apply to explosive exposures.

#### F.9.2.1.3 Physiological Stress

Birds 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. Exposures to explosives have the potential to provide additional stressors beyond those that naturally occur, as described in the Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities (see Section F.1.1).

There are no data on physiological stress in birds specifically due to explosives; therefore, the limited data on physiological stress due to impulsive sounds, apply to explosive exposures.

#### F.9.2.1.4 Masking

Masking occurs when one sound, distinguished as the 'noise,' interferes with the detection or recognition of another sound. Exposure to explosives may result in masking. There are no data on masking in birds specifically due to explosives; therefore, the limited data on masking due to impulsive sounds, apply to explosive exposures. Due to the very brief duration of an explosive sound, any masking would be brief during an explosive activity.

#### F.9.2.1.5 Behavioral Reactions

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 limited data on behavioral reactions due to impulsive sounds, apply to explosive exposures.

Because data on behavioral responses by birds to explosions is limited, information on bird responses to other impulsive sounds may be informative. Seismic surveys had no noticeable effects 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. The sensitivity of birds to disturbance may also vary during different stages of the nesting cycle. Similar noise levels may be more likely to cause nest abandonment during incubation of eggs than during brooding of chicks because birds have invested less time and energy and have a greater chance of re-nesting (Knight & Temple, 1986).

#### F.9.2.1.6 Long Term Consequences

Long-term consequences to birds due to explosive exposures are considered following the Conceptual Framework for Assessing Effects from Acoustic and Explosive Activities (Section F.1.1.6).

Long-term consequences to a population are determined by examining changes in the population growth rate. Physical effects that could lead to a reduction in the population growth rate include mortality or injury, which could remove animals from the reproductive pool, and permanent hearing impairment, which could affect foraging and communication. The 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 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. More research is needed to better understand the long-term consequences of anthropogenic stressors, although intermittent exposures to explosive noise are assumed to be less likely to have lasting consequences.

## F.9.3 Energy Stressors

### F.9.3.1 Effects from In-Water Electromagnetic Devices

The kinetic energy weapon referred to as a rail gun is an in-water electromagnetic device that will be tested and eventually used in training events aboard surface vessels, firing non-explosive projectiles at land- or sea-based targets. This system charges for approximately two minutes and discharges in less than a second. The duration of the firing event is extremely short (about eight milliseconds), which makes it quite unlikely that a bird would fly over at the precise moment of firing. The short duration of each firing event also means that the likelihood of affecting any animal using magnetic fields for orientation is extremely small. Further, the high magnetic field levels experienced within 80 ft. of the launcher quickly dissipate and return to background levels beyond 80 ft. The magnetic field levels outside of the 80 ft. buffer zone would be below the most stringent guidelines for humans (i.e., people with pacemakers or active implantable medical devices). Therefore, the electromagnetic effects would be temporary in nature and not expected to result in effects on organisms (U.S. Department of the Navy, 2009a), and are not analyzed further in this section.

Birds are known to use the Earth's magnetic field as a navigational cue during seasonal migrations (Akesson & Hedenstrom, 2007; Fisher, 1971; Wiltschko & Wiltschko, 2005). Birds use numerous other orientation cues to navigate in addition to magnetic fields. These include position of the sun, celestial cues, visual cues, wind direction, and scent (Akesson & Hedenstrom, 2007; Fisher, 1971; Haftorn et al., 1988; Wiltschko & Wiltschko, 2005). It is believed that birds are able to successfully navigate long distances by using a combination of these cues. A magnetite-based (magnetic mineral) receptor mechanism in the upper beak of birds provides information on position and compass direction (Wiltschko & Wiltschko, 2005). Towed in-water electromagnetic device effects on birds would only occur underwater and would only affect diving species or species on the surface in the immediate area where the device is deployed. There is no information available on how birds react to electromagnetic fields underwater.

### F.9.3.2 Effects from In-Air Electromagnetic Devices

Currently, questions exist about far-field, non-thermal effects from low-power, in-air electromagnetic devices. Manville (2016) performed a literature review of this topic. Although findings are not always consistent, Manville (2016) reported that several peer-reviewed studies have shown non-thermal effects can include (1) affecting behavior by preventing birds from using their magnetic compass, which may in turn affect migration; (2) fragmenting the DNA of reproductive cells, decreasing the reproductive capacity of living organisms; (3) increasing the permeability of the blood-brain barrier; (4) other behavioral effects; (5) other molecular, cellular, and metabolic changes; and (6) increasing cancer risk.

Cucurachi et al. (2013) also performed a literature review of 113 studies and reported that (1) few field studies were performed (the majority were conducted in a laboratory setting); (2) 65 percent of the studies reported ecological effects both at high as well as low dosages (i.e., those that are compatible with real field situations, at least on land); (3) no clear dose-effect relationship could be discerned but that studies finding an effect applied higher durations of exposure and focused more on mobile phone frequency ranges; and (4) a lack of standardization and a limited number of observations limited the possibility of generalizing results from an organism to an ecosystem level.

Many bird species return to the same stopover, wintering, and breeding areas every year and often follow the exact same or very similar migration routes (U.S. Department of the Navy, 2002), and ample evidence exists that displaced birds can successfully reorient and find their way when one or more cues are removed (U.S. Department of the Navy, 2009a). For example, Haftorn et al. (1988) found that after removal from their nests and release into a different area, snow petrels (*Pagodrama nivea*) were able to successfully navigate back to their nests even when their ability to smell was removed. Furthermore, Wiltschko and Wiltschko (2005) report that in-air electromagnetic pulses administered to birds during an experimental study on orientation do not deactivate the magnetite-based receptor mechanism in the upper beak altogether but instead cause the receptors to provide altered information, which in turn causes birds to orient in different directions. However, these effects were temporary, and the ability of the birds to correctly orient themselves eventually returned. Similar results were found by a subsequent study by Wiltschko et al. (2011) on European robins (*Erithacus rubecula*) that tested the effects of exposure to specific wavelengths of visible light. Therefore, in the unlikely event that a bird is temporarily disoriented by an electromagnetic device, it is expected that it would still be able to reorient using its internal magnetic compass to aid in navigation once the stressor ceases or the bird and stressor are separated by sufficient distance. Therefore, any temporary disorientation experienced by birds from electromagnetic changes caused by training activities in the Study Area may be considered a short-term effect and would not hinder bird navigation abilities. Furthermore, other orientation cues may include position of the sun and moon, visual cues, wind direction, infrasound, and scent; these cues would not be affected by in-air electromagnetic devices.

The Environmental Assessment (EA) for the Upgraded AEGIS Combat System concluded that the rapid increase of the bird population around a newly constructed radar installation “indicates that any negative effects of the radiation zone overhead have been negligible.” Another study on the effects of extremely low-frequency in-air electromagnetic fields on breeding and migrating birds around the Navy’s extra-low-frequency communication system antenna in Wisconsin found no evidence that bird distribution or abundance was affected by in-air electromagnetic fields produced by the antenna. In addition, radars, including X-band systems, are frequently used to track bird movements as it has been demonstrated that they do not affect bird behavior. Moreover, previous studies have consistently determined that the chances that a bird will move in the same direction and at the same speed as a constant beam of electromagnetic radiation (e.g., while an in-air electromagnetic device tracks a target), and therefore be exposed to radiation that could cause thermal damage, are extremely small.

California least terns could be exposed to intermittent in-air electromagnetic stressors in nearshore areas where training activities occur. If present in the open water areas where training activities involving in-air electromagnetic stressors occur, Hawaiian petrel, short-tailed albatross, marbled murrelet, Newell’s shearwater, or band-rumped storm-petrel could be temporarily disturbed while foraging or migrating.

Given (1) the information provided above; (2) the dispersed nature of Navy testing and training activities at sea; and (3) the relatively low-level and dispersed use of these systems at sea, the following conclusions are reached:

- The chance that in-air electromagnetic devices would cause thermal damage to an individual bird is extremely low;
- It is possible, although unlikely, that some bird individuals would be exposed to levels of electromagnetic radiation that would cause discomfort, in which case they would likely avoid the immediate vicinity of testing and training activities;
- The strength of any avoidance response would decrease with increasing distance from the in-air electromagnetic device; and
- No long-term or population-level effects would occur.

### F.9.4 Physical Disturbance and Strike Stressors

#### F.9.4.1 Effects from Vessels

Direct collisions with most Navy vessels (or a vessel's rigging, cables, poles, or masts) are unlikely but may occur, especially at night. Many bird species are attracted to artificial lighting, particularly Procellariiformes. Newell's shearwater and Hawaiian petrel fledglings are particularly attracted to light, which can cause exhaustion and increase potential for collision with land-based structures (Reed et al., 1985). Lighting on boats and vessels has also contributed to bird fatalities in open-ocean environments when birds are attracted to these lights, usually in inclement weather conditions (Merkel & Johansen, 2011). Birds can become disoriented at night in the presence of artificial light (Favero et al., 2011; Hamilton, 1958; Hyrenbach, 2001, 2006), and lighting on vessels may attract some birds, increasing the potential for harmful encounters. Other effects would be the visual and behavioral disturbance from a vessel. Birds respond to moving vessels in various ways. Some birds, including certain species of gulls, storm petrels, and albatrosses, commonly follow vessels (Favero et al., 2011; Hyrenbach, 2001, 2006); while other species such as plovers, curlews, frigatebirds, and sooty terns seem to avoid vessels (Borberg et al., 2005; Hyrenbach, 2006). There could be a slightly increased risk of effects during the winter, or fall/spring migrations when migratory birds use celestial clues during night time flight and are concentrated in coastal areas. However, despite this concentration, most birds would still be able to avoid collision with a vessel. Vessel movements could elicit short-term behavioral or physiological responses (e.g., alert response, startle response, fleeing the immediate area, temporary increase in heart rate).

Navy aircraft carriers, surface combatant vessels, and amphibious warfare ships are minimally lighted for tactical purposes. For vessels of this type there are two white lights that shine forward and one that shines aft; these lights must be visible for at least 6 NM. A single red and a single green light are located on the port and starboard sides of vessels, respectively. These lights are visible for a minimum of 3 NM. Solid white lighting appears more problematic for birds, especially nocturnal migrants (Gehring et al., 2009; Poot et al., 2008). Navy vessel lights are mostly solid, but sometimes may not appear solid because of the constant movement of the vessel (wave action), making vessel lighting potentially less problematic for birds in some situations.

While some potential exists for birds to be struck by vessels as they are foraging, resting, or flying near the water surface, most birds would be expected to see or hear an oncoming vessel and to fly or swim away to avoid a potentially harmful encounter. Injury or mortality could occur if a bird were struck, but most bird encounters with vessels would be expected to result in a brief behavioral and physiological response as described above. It should be noted that such responses involve at the least a temporary displacement of birds from foraging areas, resulting in energetic costs to the birds (Velando & Munilla, 2011). Birds would be expected to return and resume foraging soon after the vessel passed through the area, or to forage elsewhere, and the fitness of individual birds would probably not be compromised.

Other harmful bird-vessel interactions are commonly associated with commercial fishing vessels because birds are attracted to concentrated food sources around these vessels (Dietrich & Melvin, 2004; Melvin & Parrish, 2001). However, concentrated food sources are not associated with Navy vessels, so birds following Navy vessels would be very unlikely.

Amphibious vessel movements could elicit short-term behavioral or physiological responses such as alert response, startle response, cessation of feeding, fleeing the immediate area, nest abandonment, and a temporary increase in heart rate. There could be a slightly increased risk of effects during the winter, or fall/spring migrations and during nesting season when migratory birds are concentrated in coastal areas where amphibious vessels have the potential to disturb nesting or foraging shorebirds such as the ESA-listed California least tern. The general health of individual birds would not be compromised, unless a direct strike occurred. However, it is highly unlikely that a bird would be struck in this scenario because most foraging shorebirds in the vicinity of the approaching amphibious vessel would likely be dispersed by the noise of its approach before it could come close enough to strike a bird (Section 3.9.3.4.1, Effects from Vessels and In-Water Devices).

## F.9.5 Entanglement Stressors

### F.9.5.1 Effects from Decelerators/Parachutes

If the decelerator/parachute and its lines sink to the seafloor in an area where the bottom is calm, it would remain there undisturbed. Over time, it may become covered by sediment in most areas or colonized by attaching and encrusting organisms, which would further stabilize the material and reduce the potential for reintroduction as an entanglement risk. If bottom currents are present, the canopy may billow and pose an entanglement threat to birds that feed in benthic habitats. Bottom-feeding birds tend to forage in nearshore areas rather than offshore, where these decelerators/parachutes are used; therefore, birds are not likely to encounter decelerators/parachutes once they reach the seafloor. The potential for a bird to encounter an expended decelerator/parachute at the surface or in the water column is extremely low, it is even less probable at the seafloor given the general improbability of a bird being near the deployed decelerator/parachute as well as the general behavior of birds. Depending on how quickly the decelerator/parachute may degrade, the risk may increase with time if the decelerator/parachute remains intact. Factors that may influence degradation times include exposure to ultraviolet radiation and the extent of physical damage of the decelerator/parachute on the water's surface, as well as water temperature and sinking depth.

## F.9.6 Ingestion Stressors

### F.9.6.1 Effects from Military Expended Materials Other Than Munitions

The analysis in this section includes the potential ingestion of military expended materials other than munitions, all of which are expended away from nearshore habitats and close to the water surface. Tables 3.0-20, 3.0-21, 3.0-24, and 3.0-26describe the annual quantities and locations where these materials would be generated by training and testing activities under Alternatives 1 and 2. Appendix A (Navy Activity Descriptions) provides more specific information on the activities that may result in ingestion stressors, and the typical locations where these activities occur.

While it has been widely documented that a wide range of marine organisms (including zooplankton, baleen whales, and seabirds) will ingest plastic, the mechanism that causes these organisms to do so was discovered only recently (Savoca, 2016; Savoca et al., 2016). Procellariiformes, or tube-nosed seabirds (e.g., albatrosses, shearwaters and petrels) utilize a highly developed sense of smell to find food that is patchily distributed in offshore and open ocean environments. Specifically, these birds are attracted to dimethyl sulfide, which is produced when the cell walls of algae are damaged (e.g., when marine herbivores such as krill eat it), thereby alerting the seabirds that food (e.g., krill) are nearby. Through a literature review, Savoca et al. (2016) demonstrated that seabirds that utilize dimethyl sulfide as a foraging cue consumed plastic nearly six times more frequently than species that were not attracted to dimethyl sulfide. Savoca et al. (2016) also performed field studies that confirmed that algae growing on three of the most common types of plastic debris (polypropylene and low- and high-density polyethylene) can produce dimethyl sulfide within three weeks at concentrations at least four orders of magnitude above the behavioral detection threshold for Antarctic prions (*Pachyptila desolata*), thereby creating an "olfactory trap."

Birds could potentially ingest expended materials other than munitions used by the Navy during training and testing activities within the Study Area. The Navy expends the following types of materials that could become ingestion stressors for birds during training and testing in the Study Area: missile components, target fragments, chaff and flare endcaps/pistons, and decelerators/parachutes.

Ingestion of expended materials by birds could occur in all large marine ecosystems and open ocean areas and would occur either at the surface or just below the surface portion of the water column, depending on the size and buoyancy of the expended object and the feeding behavior of the birds. Floating material of ingestible size could be eaten by birds that feed at or near the water surface, while materials that sink pose a potential risk to diving birds that feed just below the water's surface (Titmus & Hyrenbach, 2011). Some items, such as decelerators/parachutes or sonobuoys are too large to be ingested and will not be discussed further. Also, decelerators/parachutes sink rapidly to the seafloor.

Physiological effects on birds from ingestion include blocked digestive tracts and subsequent food passage, blockage of digestive enzymes, lowered steroid hormone levels, delayed ovulation (egg maturation), reproductive failure, nutrient dilution (nonnutritive debris displaces nutritious food in the gut), exposure to indirect effects from harmful chemicals found in and on the plastic material, and altered appetite satiation (the sensation of feeling full), which can lead to starvation (Azzarello & Van Vleet, 1987). While ingestion of marine debris has been linked to bird mortalities, sublethal effects are more common (Moser & Lee, 1992).

Many species of seabirds are known to ingest floating plastic debris and other foreign matter while feeding on the surface of the ocean (Auman et al., 1997; Yamashita et al., 2011). Evidence indicates that physical and toxicological effects from plastic ingestion by seabirds are widespread among species and pervasive in terms of the number of individuals affected, and that effects are increasing (Kain et al., 2016; Wilcox et al., 2015). For example, 21 of 38 seabird species (55 percent) collected off the coast of North Carolina from 1975 to 1989 contained plastic particles (Moser & Lee, 1992). The mean particle sizes of ingested plastic were positively correlated with the birds' size though the mean mass of plastic found in the stomachs and gizzards of 21 species was below 3 grams. In Hawaii, the proportion of necropsied Newell's shearwaters and wedge-tailed shearwaters (*Ardenna pacifica*) that were found to have ingested plastic more than doubled from 2007 to 2014 (Kain et al., 2016). The number of plastic particles found in the stomachs of northern fulmars in the North Sea increased by two-to-three-fold from the mid-1980s to mid-1990s and has remained at about 30 particles per bird since then. Since the 1980s, concentrations of industrial plastics in the ocean waters and their consumption by fulmars has decreased by about 75 percent, while the abundance of user plastics and their consumption has shown no obvious trend (van Franeker & Law, 2015). Some seabirds have used plastic and other marine debris for nest building which may lead to ingestion of that debris (Votier et al., 2011). Indirect ingestion of plastic also occurs from consuming prey (such as fishes) that ingest plastic.

Plastic is often mistaken for prey, and the incidence of plastic ingestion appears to be related to a bird's feeding mode and diet (Henry et al., 2011; Provencher et al., 2014). Seabirds that feed by pursuit-diving, surface-seizing, and dipping tend to ingest plastic, while those that feed by plunging or piracy typically do not ingest plastic (Azzarello & Van Vleet, 1987; Provencher et al., 2014). Birds of the order Procellariiformes, which include petrels, shearwaters, and albatrosses, tend to accumulate more plastic than other species (Azzarello & Van Vleet, 1987; Moser & Lee, 1992; Pierce et al., 2004; Provencher et al., 2014). Some birds, including gulls and terns, commonly regurgitate indigestible parts of their food items such as shell and fish bones. However, the structure of the digestive systems of most Procellariiformes makes it difficult to regurgitate solid material such as plastic (Azzarello & Van Vleet, 1987; Moser & Lee, 1992; Pierce et al., 2004). Two species of albatross (*Diomedeidae*) have also been reported to ingest plastic while feeding at sea. While such studies have not conclusively shown that plastic ingestion is a significant source of direct mortality, it may be a contributing factor to other causes of albatross mortality (Naughton et al., 2007).

As summarized by Pierce et al. (2004), Auman et al. (1997) and Azzarello and Van Vleet (1987) documented consequences of plastic ingestion by seabirds include blockage of the intestines and ulceration of the stomach, reduction in the functional volume of the gizzard leading to a reduction of digestive capability, and distention of the gizzard leading to a reduction in hunger. Dehydration has also been documented in seabirds that have ingested plastic (Sievert & Sileo, 1993). Studies have found negative correlations between body weight and plastic load, as well as between body fat (a measure of energy reserves) and the number of pieces of plastic in a seabird's stomach (Auman et al., 1997; Ryan, 1987; Sievert & Sileo, 1993). Other possible concerns that have been identified include toxic plastic additives and toxic contaminants that could be adsorbed to the plastic from ambient seawater. Pierce et al. (2004) described two cases where plastic ingestion caused seabird mortality from starvation. The examination of a deceased adult northern gannet revealed that a 1.5 in. diameter plastic bottle cap lodged in its gizzard blocked the passage of food into the small intestine, which resulted in its death from starvation. Northern gannets are substantially larger, and dive deeper than the ESA-listed birds in the Study Area. Also, since gannets typically utilize flotsam in nest building (Votier et al., 2011), they may be more susceptible to ingesting marine debris than other species as it gathers that material. Dissection of an adult greater shearwater’s gizzard revealed that a 1.5 by 0.5 in. fragment of plastic blocked the passage of food in the digestive system, which also resulted in death from starvation.

Species such as storm-petrels, albatrosses, shearwaters, fulmars, and noddies that forage by picking prey from the surface may have a greater potential to ingest any floating plastic debris (Donnelly-Greenan et al., 2014). Ingestion of plastic military expended material by any species from the taxonomic groups found within the Study Area has the potential to affect individual birds. The risk of plastic ingestion and impaction in chicks of many species of seabirds may be different from the risks to adults. Albatross chicks appear to be at greater risk than adults, because of their high rates of ingestion and apparent low frequency of regurgitative casting of indigestible material. Hyrenbach et al. (2015) demonstrated that almost 100 percent of chicks of black-footed and Laysan albatrosses breeding in the Northwestern Hawaiian Islands ingest plastics during the pre-fledging period when they are dependent upon food brought to the breeding colony by parents. Floating plastic items are ingested by adult albatrosses and regurgitated to chicks along with normal food items. Negative effects of plastic ingestion may result from impaction of the upper gastrointestinal tract and interference with passage of food through the digestive system, contributing to reduced resistance to disease and lowered post-fledging survival. Significant correlations between plastic loads and body condition or growth rates, were not found, however (Hyrenbach et al., 2015). Flesh-footed shearwater (*Puffinus carnipes*) fledglings in eastern Australia have been found to contain relatively large amounts of plastics, which is correlated with poor body condition and tissue contaminant loads (Lavers et al., 2014).

The distribution of floating expended items would be irregular in both space and time, as training and testing activities do not occur in the same place each time. The random distribution of items across the large Study Area yields very low probabilities that seabirds will encounter a floating item. However, when a seabird does encounter a floating item of ingestible size, an ingestion risk may exist. Although most military expended material components are expected to sink to the seafloor and spend limited periods within the water column, some items remain buoyant for an extended period. Expended training and testing material, such as missile components or target fragments that float, may be encountered by seabirds in the waters of the Study Area, increasing the potential for ingestion of smaller components. Ocean currents concentrate plastic debris, making seabirds that feed along frontal zones more susceptible (Azzarello & Van Vleet, 1987). While some seabird ingestion of expended materials could occur, these factors indicate that a small number of birds would be affected and that population level effects would not be expected.

**Target-Related Materials**

As described in Section 3.0.3.3.6.3 (Military Expended Materials), at-sea targets are usually remotely operated airborne, surface, or subsurface traveling units, most of which are designed to be recovered for reuse. However, if they are used during activities that use high explosives then they may result in fragments. Expendable targets that may result in fragments would include air-launched decoys, surface targets (e.g., marine markers, paraflares, cardboard boxes, and 10 ft. diameter red balloons), and mine shapes. Most target fragments would sink quickly to the seafloor. Floating material, such as Styrofoam, may be lost from target boats and remain at the surface for some time. Only targets that may result in smaller fragments that do not immediately sink are included in the analyses of ingestion potential.

There are additional types of targets discussed previously, but only surface targets, subsurface targets, air targets, Sinking Exercise ship hulks, and mine shapes would be expected to result in fragments when high-explosive munitions are used.

**Chaff**

As described in Section 3.0.3.3.6.3 (Military Expended Materials), large areas of air space and open water within the Study Area would be exposed to chaff at very low concentrations. This same section also provides a general discussion of chaff as an ingestion stressor and concludes that chaff poses little risk to organisms, except at concentrations substantially higher than those that could reasonably occur from military training. Additional information is provided below.

It is unlikely that chaff would be selectively ingested (U.S. Department of the Air Force, 1997). Ingestion of chaff fibers is not expected to cause physical damage to a bird’s digestive tract based on the fibers’ small size (ranging in lengths of 0.25 to 3 in. with a diameter of about 40 micrometers) and flexible nature, as well as the small quantity that could reasonably be ingested. In addition, concentrations of chaff fibers that could reasonably be ingested are not expected to be toxic to seabirds. Scheuhammer (1987) reviewed the metabolism and toxicology of aluminum in birds and mammals and found that intestinal adsorption of orally ingested aluminum salts was very poor, and the small amount adsorbed was almost completely removed from the body by excretion. Dietary aluminum normally has minor effects on healthy birds and mammals, and often high concentrations (greater than 1,000 milligrams per kilogram) are needed to induce effects such as impaired bone development, reduced growth, and anemia (U.S. Department of the Navy, 1999). A bird weighing 2.2 lb. would need to ingest more than 83,000 chaff fibers per day to receive a daily aluminum dose equal to 1,000 per kilogram; this analysis was based on chaff consisting of 40 percent aluminum by weight and a 5 ounce chaff canister containing 5 million fibers. As an example, an adult herring gull weighs about 1.8–2.7 lb. (Cornell Lab of Ornithology, 2009). It is highly unlikely that a bird would ingest a toxic dose of chaff based on the anticipated environmental concentration of chaff (i.e., 1.8 fibers per square foot for an unrealistic, worst-case scenario of 360 chaff cartridges simultaneously released at a single drop point).

### Flares

A general discussion of flares as an ingestion stressor is presented in Section 3.0.3.3.4.2 (Military Expended Materials). Ingestion of flare compression pads or pistons 1.3 in. in diameter and 0.13 in. thick (U.S. Department of the Air Force, 1997) by birds may result in gastrointestinal obstruction or reproductive complications. Based on the information presented above, if a seabird were to ingest a compression pads or pistons, the response would vary based on the species and individual bird. The responses could range from none, to sublethal (reduced energy reserves), to lethal (digestive tract blockage leading to starvation). Ingestion of compression pads or pistons by species that regularly regurgitate indigestible items would likely have no adverse effects. However, compression pads or pistons are similar in size to those plastic pieces described above that caused digestive tract blockages and eventual starvation. Therefore, ingestion of compression pads or pistons could be lethal to some individual seabirds. Species with small gizzards and anatomical constrictions that make it difficult to regurgitate solid material would likely be most susceptible to blockage (such as Procellariiformes). Based on available information, it is not possible to accurately estimate actual ingestion rates or responses of individual birds.

## F.9.7 Secondary Stressors

This section analyzes potential effects on birds exposed to stressors indirectly through effects on habitat and prey availability.

### F.9.7.1 Effects on Habitat

The potential of water, air quality, and abiotic habitat stressors associated with training and testing activities to indirectly affect birds, as a secondary stressor, was analyzed. The assessment of potential water, air quality, and abiotic habitat stressors is discussed in previous sections in this Draft EIS/OEIS (Section 3.1, Air Quality; and Section 3.2, Sediments and Water Quality). These analyses address specific activities in local environments that may affect bird habitats. At-sea activities that may affect water and air include general emissions, and at-sea activities that may affect habitats include explosives and physical disturbance and strike.

As noted in Section 3.1 (Air Quality), and Section 3.2 (Sediments and Water Quality), implementation of the No Action Alternative, Alternative 1, or Alternative 2 would minimally affect sediments, water, air quality, or habitats, and therefore would not indirectly affect seabirds as secondary stressors. Furthermore, any physical effects on seabird habitats would be temporary and localized because training and testing activities would occur infrequently. These activities would not be expected to adversely affect seabirds or seabird habitats.

Indirect effects on sediments, water or air quality under Alternative 1 or Alternative 2 would have no effect on ESA-listed bird species due to: (1) the temporary nature of effects on sediments, water, or air quality, (2) the distribution of temporary sediments, water, or air quality effects, (3) the wide distribution of birds in the Study Area, and (4) the dispersed spatial and temporal nature of the training and testing activities that may have temporary sediments, water, or air quality effects. No long-term or population-level effects are expected.

Pursuant to the ESA, secondary effects on habitat during training or testing activities as described under Alternative 1 and Alternative 2 may affect least terns, Hawaiian petrels, short-tailed albatrosses, marbled murrelets, Newell's shearwaters, and band-rumped storm petrels. The Navy has consulted with the USFWS as required by section 7(a)(2) of the ESA.

#### F.9.7.2 Effects on Prey Availability

As noted in Section 3.4 (Invertebrates) and Section 3.5 (Fishes), implementation of the No Action Alternative, Alternative 1, or Alternative 2 would not adversely affect populations of invertebrate or fish prey resources (e.g., crustaceans, bivalves, worms, sand lance, herring, etc.) of birds and therefore would not indirectly affect birds as secondary stressors. Any effects on seabird prey resources would be temporary and localized. Furthermore, as discussed above, these activities are expected to have minimal effects on bird habitats. Additional detail is provided below.

As discussed in Section 3.4.3.7 (Secondary Stressors), effects on invertebrate prey availability resulting from explosives, explosives byproducts, unexploded munitions, metals, and chemicals would likely be negligible overall and population-level effects on marine invertebrates are not expected. Because individuals of many invertebrate taxa prey on other invertebrates, mortality resulting from explosions or exposure to metals or chemical materials would reduce the number of invertebrate prey items available. A few species prey upon fish, and explosions and exposure to metals and chemical materials could result in a minor reduction in the number of fish available. However, the effect is expected to be small and discountable. Any vertebrate or invertebrate animal killed or significantly impaired by Navy activities could potentially represent an increase in food availability for scavenging invertebrates. None of the effects described above would likely be detectable at the population or subpopulation level.

As noted in Section 3.6.3.7.2 (Fishes, Effects on Prey Availability), prey species might exhibit a strong startle reaction to detonations that might include swimming to the surface or scattering away from the source. This startle and flight response is the most common secondary defense among animals (Hanlon & Messenger, 1996). The sound from underwater explosions might induce startle reactions and temporary dispersal of schooling fishes if they are within close proximity to an explosion (Popper et al., 2014; Wright, 1982), which in turn could make them more visible to predators (Kastelein et al., 2008). The abundances of fish and invertebrate prey species near the detonation point could be diminished for a short period of time before being repopulated by animals from adjacent waters. Alternatively, any prey species that would be directly injured or killed by the blast could draw in scavengers from the surrounding waters that would feed on those organisms, who in turn could be susceptible to becoming directly injured or killed by subsequent explosions. Any of these scenarios would be temporary, only occurring during activities involving explosives, and no lasting effect on prey availability or the food web would be expected. Indirect effects of underwater detonations and high explosive munitions use under the Proposed Action would not result in a decrease in the quantity or quality of fish populations in the Study Area.

Based on Sections 3.4 (Invertebrates) and 3.6 (Fishes), project-related stressors would not affect populations of invertebrates and fishes that support birds in the Study Area. Therefore, no secondary effects associated with prey availability are expected. Furthermore, the Navy will implement mitigation (e.g., not conducting gunnery activities within a specified distance of shallow-water coral reefs) to avoid potential effects from explosives and physical disturbance and strike stressors on seafloor resources in mitigation areas throughout the Study Area (see Sections 5.7.1, Shallow-Water Coral Reef and Precious Coral Bed Mitgation Areas; and Section 5.7.2, Artificial Reef, Hard Bottom Substrate, and Shipwreck Mitigation Areas). This mitigation will consequently help avoid potential effects on bird prey that inhabits shallow-water coral reefs, live hard bottom, precious coral beds, artificial reefs, and shipwrecks.

Pursuant to the ESA, secondary effects on prey availability during training or testing activities as described under Alternative 1 and Alternative 2 may affect least terns, Hawaiian petrels, short-tailed albatrosses, marbled murrelets, Newell's shearwaters, and band-rumped storm petrels. The Navy has consulted with the USFWS as required by section 7(a)(2) of the ESA.

# References

Able, K. W. and M. P. Fahay. (1998). *The First Year in the Life of Estuarine Fishes in the Middle Atlantic Bight*. New Brunswick, NJ: Rutgers University Press.

Abramson, L., S. Polefka, S. Hastings, and K. Bor. (2011). *Reducing the Threat of Ship Strikes on Large Cetaceans in the Santa Barbara Channel Region and Channel Islands National Marine Sanctuary: Recommendations and Case Studies* (Marine Sanctuaries Conservation Series). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Ocean Service, National Marine Sanctuary Program.

Adimey, N. M., C. A. Hudak, A. R. Powell, K. Bassos-Hull, A. Foley, N. A. Farmer, L. White, and K. Munch. (2014). Fishery gear interactions from stranded bottlenose dolphins, Florida manatees and sea turtles in Florida, U.S.A. *Marine Pollution Bulletin 81* 103–115. DOI:10.1016/j.marpolbul.2014.02.008

Agency for Toxic Substances and Disease Registry. (2007). *Toxicological Profile for Lead*. Atlanta, GA: U.S. Department of Health and Human Services.

Agency for Toxic Substances and Disease Registry. (2008). *Toxicological Profile for Perchlorates*. Atlanta, GA: U.S. Department of Health and Human Services.

Aggio, J. F., R. Tieu, A. Wei, and C. D. Derby. (2012). Oesophageal chemoreceptors of blue crabs, *Callinectes sapidus*, sense chemical deterrents and can block ingestion of food. *The Journal of Experimental Biology 215* (10): 1700–1710. DOI:10.1242/jeb.065854

Aguilar de Soto, N., N. Delorme, J. Atkins, S. Howard, J. Williams, and M. Johnson. (2013). Anthropogenic noise causes body malformations and delays development in marine larvae. *Scientific Reports 3* (2831): 2831. DOI:10.1038/srep02831

Aguilar, N., M. Carrillo, I. Delgado, F. Diaz, and A. Brito. (2000). *Fast ferries impact on cetacean in Canary Islands: Collisions and displacement* [Abstract]. Presented at the European Research on Cetaceans - 14. Cork, Ireland.

Akesson, S. and A. Hedenstrom. (2007). How migrants get there: Migratory performance and orientation. *BioScience 57* (2): 123–133.

Alderfer, J. (2003). Auks, murres, puffins. In M. Baughman (Ed.), *National Geographic Refererence Atlas to the Birds of North America* (pp. 176–185). Washington, DC: National Geographic Society.

Allen, R., D. Jarvis, S. Sayer, and C. Mills. (2012). Entanglement of grey seals *Halichoerus grypus* at a haul out site in Cornwall, UK. *Marine Pollution Bulletin 64* 2815–2819.

Alzugaray, L., M. Di Martino, L. Beltramino, V. J. Rowntree, M. Sironi, and M. M. Uhart. (2020). Anthropogenic debris in the digestive tract of a southern right whale (*Eubalaena australis*) stranded in Golfo Nuevo, Argentina. *Marine Pollution Bulletin 161* 111738. DOI:https://doi.org/10.1016/j.marpolbul.2020.111738

Andersen, D. E., O. J. Rongstad, and W. R. Mytton. (1990). Home-range changes in raptor exposed to increased human activity levels in southeastern Colorado. *Wildlife Society Bulletin 18* 134–142.

Andrady, A. L. (2011). Microplastics in the marine environment. *Marine Pollution Bulletin 62* 1595–1605. DOI:10.1016/j.marpolbul.2011.05.030

André, M., M. Solé, M. Lenoir, M. Durfort, C. Quero, A. Mas, A. Lombarte, M. van der Schaar, M. Lopez-Bejar, M. Morell, S. Zaugg, and L. Houégnigan. (2011). Low-frequency sounds induce acoustic trauma in cephalopods. *Frontiers in Ecology and the Environment 9* (9): 489–493. DOI:10.1890/100124

Andriguetto-Filho, J. M., A. Ostrensky, M. R. Pie, U. A. Silva, and W. A. Boeger. (2005). Evaluating the impact of seismic prospecting on artisanal shrimp fisheries. *Continental Shelf Research 25* 1720–1727. DOI:10.1016/j.csr.2005.05.003

Ankley, G. T. (1996). Evaluation of metal/acid-volatile sulfide relationships in the prediction of metal bioaccumulation by benthic macroinvertebrates. *Environmental Toxicology and Chemistry 15* 2138–2146.

Aplin, J. A. (1947). The effect of explosives on marine life. *California Fish and Game 33* 23–30.

Arfsten, D. P., C. L. Wilson, and B. J. Spargo. (2002). Radio frequency chaff: The effects of its use in training on the environment. *Ecotoxicology and Environmental Safety 53* 1–11.

Arifin, L., A. Bakti, A. Virgota, L. P. A. Damayanti, T. H. U. Radiman, A. Retnowulan, Hernawati, A. Sabil, and D. Robbe. (2012). Biorock Reef Restoration in Gili Trawangan, North Lombok, Indonesia. In T. J. Goreau & R. K. T. Trench (Eds.), *Innovative Methods of Marine Ecosystem Restoration* (pp. 59–80). Boca Raton, FL: CRC Press.

Armstrong, J. D., D.-C. Hunter, R. J. Fryer, P. Rycroft, and J. E. Orpwood. (2015). Behavioural responses of Atlantic salmon to mains frequency magnetic fields. *Scottish Marine and Freshwater Science 6* 1–17. DOI:10.7489/1621-1

Aschettino, J. M., D. T. Engelhaupt, A. G. Engelhaupt, A. Dimatteo, T. Pusser, M. F. Richlen, and J. T. Bell. (2020). Satellite telemetry reveals spatial overlap between vessel high-traffic areas and humpback whales (*Megaptera novaeangliae*) near the mouth of the Chesapeake Bay. *Frontiers in Marine Science 7*.

Au, W. W. L. and M. Green. (2000). Acoustic interaction of humpback whales and whale-watching boats. *Marine Environmental Research 49* (5): 469–481.

Auman, H. J., J. P. Ludwig, J. P. Giesy, and T. Colborn. (1997). Plastic ingestion by Laysan Albatross chicks on Sand Island, Midway Atoll, in 1994 and 1995. In G. Robinson & R. Gales (Eds.), *Albatross Biology and Conservation* (pp. 239–244). Chipping Norton, Australia: Surrey Beatty & Sons.

Azzarello, M. Y. and E. S. Van Vleet. (1987). Marine birds and plastic pollution. *Marine Ecology - Progress Series 37* 295–303.

Baird, R. W. and S. K. Hooker. (2000). Ingestion of plastic and unusal prey by a juvenile harbour porpoise. *Marine Pollution Bulletin 40* (8): 719–720.

Barron, D. G., J. D. Brawn, L. K. Butler, L. M. Romero, and P. J. Weatherhead. (2012). Effects of military activity on breeding birds. *The Journal of Wildlife Management 76* (5): 911–918.

Bassett, B. L., J. A. Hostetler, E. Leone, C. P. Shea, B. D. Barbeau, G. L. Lonati, A. L. Panike, A. Honaker, and L. I. Ward-Geiger. (2020). Quantifying sublethal Florida manatee watercraft interactions by examining scars on manatee carcasses. *Endangered Species Research 43* 395-408.

Bauer, G. B., M. Fuller, A. Perry, J. R. Dunn, and J. Zoeger. (1985). Magnetoreception and Biomineralization of Magnetite in Cetaceans. In *Magnetite Biomineralization and Magnetoreception in Organisms: A New Biomagnetism* (pp. 487–507). New York, NY: Plenum Press.

Bauer, G. B., J. C. Gaspard, III, D. E. Colbert, J. B. Leach, A. Stamper, and R. Reep. (2012). Tactile discrimination of textures by Florida manatees (*Trichechus manatus latirostris*). *Marine Mammal Science 28* (4): 456–471.

Baulch, S. and C. Perry. (2014). Evaluating the impacts of marine debris on cetaceans. *Marine Pollution Bulletin 80* (1–2): 210–221. DOI:10.1016/j.marpolbul.2013.12.050

Bazzi, A. O. (2014). Heavy metals in seawater, sediments and marine organisms in the Gulf of Chabahar, Oman Sea. *Journal of Oceanography and Marine Science 5* (3): 20–29. DOI:10.5897/JOMS2014.0110

Beck, C. A. and N. B. Barros. (1991). The impact of debris on the Florida manatee. *Marine Pollution Bulletin 22* (10): 508–510.

Becker, N. M. (1995). *Fate of Selected High Explosives in the Environment: A Literature Review* (LAUR-95-1018). Los Alamos, NM: Los Alamos National Laboratories.

Bejder, L., A. Samuels, H. Whitehead, and N. Gales. (2006). Interpreting short-term behavioural responses to disturbance within a longitudinal perspective. *Animal Behaviour 72* 1149–1158. DOI:10.1016/j.anbehav.2006.04.003

Benjamins, S., V. Harnois, H. C. M. Smith, L. Johanning, L. Greenhill, C. Carter, and B. Wilson. (2014). *Understanding the Potential for Marine Megafauna Entanglement Risk from Marine Renewable Energy Developments* (Commissioned Report No. 791). Battleby, Scotland: Scottish Natural Heritage.

Benson, S. R., K. A. Forney, J. T. Harvey, J. V. Carretta, and P. H. Dutton. (2007). Abundance, distribution, and habitat of leatherback turtles (*Dermochelys coriacea*) off California, 1990–2003. *Fishery Bulletin 105* (3): 337–347.

Berglind, R., D. Menning, R. Tryman, A. Helte, P. Leffler, and R. M. Karlsson. (2009). *Environmental Effects of Underwater Explosions: A Literature Study*. Kjeller, Norway: Forsvarets Forskningsinstitut.

Bergmann, M., L. Gutow, and M. Klages. (2015). *Marine Anthropogenic Litter*. New York, NY and London, United Kingdom: Springer.

Berlett, B. S. and E. R. Stadtman. (1997). Protein oxidation in aging, disease, and oxidative stress. *The Journal of Biological Chemistry 272* (33): 20313–20316.

Berman-Kowalewski, M., F. M. D. Gulland, S. Wilkin, J. Calambokidis, B. Mate, J. Cordaro, D. Rotstein, J. S. Leger, P. Collins, K. Fahy, and S. Dover. (2010). Association Between Blue Whale (*Balaenoptera musculus*) Mortality and Ship Strikes Along the California Coast. *Aquatic Mammals 36* (1): 59–66. DOI:10.1578/am.36.1.2010.59

Besseling, E., E. M. Foekema, J. A. Van Franeker, M. F. Leopold, S. Kuhn, E. L. B. Rebolledo, E. Hebe, L. Mielke, J. Ijzer, P. Kamminga, and A. A. Koelmans. (2015). Microplastic in a macro filter feeder: Humpback whale *Megaptera novaeangliae*. *Marine Pollution Bulletin 95* (1): 248–252. DOI:10.1016/j.marpolbul.2015.04.007

Bexton, S., D. Thompson, A. Brownlow, J. Barley, R. Milne, and C. Bidwell. (2012). Unusual mortality of pinnipeds in the United Kingdom associated with helical (corkscrew) injuries of anthropogenic origin. *Aquatic Mammals 38* (3): 229–240. DOI:10.1578/AM.38.3.2012.229

Bickel, S. L., J. D. Malloy Hammond, and K. W. Tang. (2011). Boat-generated turbulence as a potential source of mortality among copepods. *Journal of Experimental Marine Biology and Ecology 401* (1–2): 105–109. DOI:10.1016/j.jembe.2011.02.038

Bilkovic, D., M. Mitchell, J. Davis, E. Andrews, A. King, P. Mason, J. Herman, N. Tahvildari, and J. Davis. (2017). *Review of boat wake wave impacts on shoreline erosion and potential solutions for the Chesapeake Bay* (STAC Publication Number 17-002). Edgewater, MD: Chesapeake Bay Program Scientific and Technical Advisory Committee.

Bishop, M. J. (2008). Displacement of epifauna from seagrass blades by boat wake. *Journal of Experimental Marine Biology and Ecology 354* (1): 111–118. DOI:10.1016/j.jembe.2007.10.013

Black, B. B., M. W. Collopy, H. F. Percival, A. A. Tiller, and P. G. Bohall. (1984). *Effects of Low Level Military Training Flights on Wading Bird Colonies in Florida*. Gainesville, FL: Florida Cooperative Fish and Wildlife Research Unit School of Forest Resources and Conservation University of Florida.

Bleckmann, H. and R. Zelick. (2009). Lateral line system of fish. *Integrative Zoology 4* (1): 13–25.

Bloom, P. and M. Jager. (1994). The injury and subsequent healing of a serious propeller strike to a wild bottlenose dolphin (*Tursiops truncatus*) resident in cold waters off the Northumberland coast of England. *Aquatic Mammals 20.2* 59–64.

Bluhm, B. A. and R. Grandinger. (2008). Regional variability in food availability for Arctic marine mammals. *Ecological Applications 18* (2): S77–S96.

Bocconcelli, A. (2009). *Fine-Scale Focal Dtag Behavioral Study in the Gulf of Maine* (Marine Mammals & Biological Oceanography Annual Reports: FY09). State College, PA and Scituate, MA: Office of Naval Research.

Boehlert, G. W. and A. B. Gill. (2010). Environmental and ecological effects of ocean renewable energy development: A current synthesis. *Oceanography 23* (2): 68–81.

Boerger, C. M., G. L. Lattin, S. L. Moore, and C. J. Moore. (2010). Plastic ingestion by planktivorous fishes in the North Pacific Central Gyre. *Marine Pollution Bulletin 60* (12): 2275–2278. DOI:10.1016/j.marpolbul.2010.08.007

Borberg, J. M., L. T. Ballance, R. L. Pitman, and D. G. Ainley. (2005). A test for bias attributable to seabird avoidance of ships during surveys conducted in the tropical Pacific. *Marine Ornithology 33* 173–179.

Botsford, L. W., D. R. Brumbaugh, C. Grimes, J. B. Kellner, J. Largier, M. R. O'Farrell, S. Ralston, E. Soulanille, and V. Wespestad. (2009). Connectivity, sustainability, and yield: Bridging the gap between conventional fisheries management and marine protected areas. *Reviews in Fish Biology and Fisheries 19* (1): 69–95. DOI:10.1007/s11160-008-9092-z

Bowles, A. E. (1995). Chapter 8: Responses of Wildlife to Noise. In R. L. Knight & K. J. Gutzwiller (Eds.), *Wildlife and Recreationists: Coexistence Through Management and Research*. Washington, DC: Island Press.

Bowles, A. E., F. T. Awbrey, and J. R. Jehl. (1991). *The Effects of High-Amplitude Impulsive Noise on Hatching Success: A Reanalysis of the Sooty Tern Incident*. Wright Patterson Airforce Base, OH: Noise and Sonic Boom Impact Technology Program.

Bowles, A. E., M. Knobler, M. D. Seddon, and B. A. Kugler. (1994). *Effects of Simulated Sonic Booms on the Hatchability of White Leghorn Chicken Eggs.* Brooks Air Force Base, TX: Systems Research Laboratories.

Braun, C. D., G. B. Skomal, S. R. Thorrold, and M. L. Berumen. (2015). Movements of the reef manta ray (*Manta alfredi*) in the Red Sea using satellite and acoustic telemetry. *Marine Biology 162* 2351–2362.

Bravo Rebolledo, E. L., J. A. Van Franeker, O. E. Jansen, and S. M. Brasseur. (2013). Plastic ingestion by harbour seals (*Phoca vitulina*) in The Netherlands. *Marine Pollution Bulletin 67* (1–2): 200–202. DOI:10.1016/j.marpolbul.2012.11.035

Brierley, A. S., P. G. Fernandes, M. A. Brandon, F. Armstrong, N. W. Millard, S. D. McPhail, P. Stevenson, M. Pebody, J. Perrett, M. Squires, D. G. Bone, and G. Griffiths. (2003). An investigation of avoidance by Antarctic krill of RRS *James Clark Ross* using the *Autosub*-2 autonomous underwater vehicle. *Fisheries Research 60* 569–576.

Briggs, C., S. M. Shjegstad, J. A. K. Silva, and M. H. Edwards. (2016). Distribution of chemical warfare agent, energetics, and metals in sediments at a deep-water discarded military munitions site. *Deep Sea Research Part II: Topical Studies in Oceanography 128* 63–69.

Brix, K., P. Gillette, A. Pourmand, T. Capo, and M. Grosell. (2012). The effects of dietary silver on larval growth in the echinoderm *Lytechinus variegatus*. *Archives of Environmental Contamination and Toxicology 63* (1): 95–100. DOI:10.1007/s00244-012-9757-4

Brown, A. L. (1990). Measuring the effect of aircraft noise on sea birds. *Environmental International 16* 587–592.

Brown, B. T., G. S. Mills, C. Powels, W. A. Russell, G. D. Therres, and J. J. Pottie. (1999). The influence of weapons-testing noise on bald eagle behavior. *Journal of Raptor Research 33* (3): 227–232.

Brown, R. J., T. S. Galloway, D. Lowe, M. A. Browne, A. Dissanayake, M. B. Jones, and M. H. Depledge. (2004). Differential sensitivity of three marine invertebrates to copper assessed using multiple biomarkers. *Aquatic Toxicology 66* 267–278. DOI:10.1016/j.aquatox.2003.10.001

Brownlow, A., J. Onoufriou, A. Bishop, N. Davison, and D. Thompson. (2016). Corkscrew Seals: Grey Seal (*Halichoerus grypus*) Infanticide and Cannibalism May Indicate the Cause to Spiral Lacerations in Seals. *PLoS ONE 11* (6): e0156464. DOI:10.1371/journal.pone.0156464

Buchman, M. (2008). *National Oceanic and Atmospheric Administration Screening Quick Reference Tables, National Oceanic and Atmospheric Administration OR&R Report 08-1*. Seattle, WA: Office of Response and Restoration Division National Oceanic and Atmospheric Administration.

Buckingham, C. A., L. W. Lefebvre, J. M. Schaefer, and H. I. Kochman. (1999). Manatee response to boating activity in a thermal refuge. *Wildlife Society Bulletin 27* (2): 514–522.

Bullock, T. H., D. A. Bodznick, and R. G. Northcutt. (1983). The phylogenetic distribution of electroreception—Evidence for convergent evolution of a primitive vertebrate sense modality. *Brain Research Reviews 6* (1): 25–46.

Bureau of Ocean Energy Management. (2011). *Effects of EMFs from Undersea Power Cables on Elasmobranchs and Other Marine Species*. Camarillo, CA: U.S. Department of the Interior, Bureau of Ocean Energy Management, Regulation, and Enforcement, Pacific OCS Region.

Burger, J. (1981). Behavioural responses of herring gulls, *Larus argentatus,* to aircraft noise. *Environmental Pollution Series A, Ecological and Biological 24* (3): 177–184.

Burt, J., A. Bartholomew, P. Usseglio, A. Bauman, and P. F. Sale. (2009). Are artificial reefs surrogates of natural habitats for corals and fish in Dubai, United Arab Emirates? *Coral Reefs 28* 663–675. DOI:10.1007/s00338-009-0500-1

Caceres, C., F. G. Taboada, J. Hofer, and R. Anadon. (2013). Phytoplankton Growth and Microzooplankton Grazing in the Subtropical Northeast Atlantic. *PLoS ONE 8* (7).

Cairns, S. D. (2007). Deep-water corals: An overview with special reference to diversity and distribution of deep-water Scleractinian corals. *Bulletin of Marine Science 81* (3): 311–322.

Calambokidis, J. (2012). *Summary of Ship-Strike Related Research on Blue Whales in 2011*. Olympia, WA: Cascadia Research.

Calleson, C. S. and R. K. Frohlich. (2007). Slower boat speeds reduce risks to manatees. *Endangered Species Research 3* 295–304. DOI:10.3354/esr00056

Camargo, F. S. and C. Bellini. (2007). Report on the collision between a spinner dolphin and a boat in the Fernando de Noronha Archipelago, Western Equatorial Atlantic, Brazil. *Biota Neotropica 7* (1): 209–211.

Carr, R. S. and M. Nipper. (2003). *Assessment of Environmental Effects of Ordnance Compounds and their Transformation Products in Coastal Ecosystems*. Port Hueneme, CA: Naval Facilities Engineering Service Center.

Caruso, F., L. Hickmott, J. D. Warren, P. Segre, G. Chiang, P. Bahamonde, S. Español-Jiménez, S. LI, and A. Bocconcelli. (2021). Diel differences in blue whale (*Balaenoptera musculus*) dive behavior increase nighttime risk of ship strikes in northern Chilean Patagonia. *Integrative Zoology 16* (4): 594–611. DOI:https://doi.org/10.1111/1749-4877.12501

Cassoff, R. M., K. M. Moore, W. A. McLellan, S. G. Barco, D. S. Rotstein, and M. J. Moore. (2011). Lethal entanglement in baleen whales. *Diseases of Aquatic Organisms 96* 175–185. DOI:10.3354/dao02385

Celi, M., F. Filiciotto, D. Parrinello, G. Buscaino, M. A. Damiano, A. Cuttitta, S. D'Angelo, S. Mazzola, and M. Vazzana. (2013). Physiological and agonistic behavioural response of *Procambarus clarkii* to an acoustic stimulus. *The Journal of Experimental Biology 216* 709–718. DOI:10.1242/jeb.078865

Celi, M., F. Filiciotto, M. Vazzana, V. Arizza, V. Maccarrone, M. Ceraulo, S. Mazzola, and G. Buscaino. (2015). Shipping noise affecting immune responses of European spiny lobster (*Palinurus elephas*). *Canadian Journal of Zoology 93* 113–121. DOI:dx.doi.org/10.1139/cjz-2014-0219

Chan, A. A. Y. H., P. Giraldo-Perez, S. Smith, and D. T. Blumstein. (2010a). Anthropogenic noise affects risk assessment and attention: The distracted prey hypothesis. *Biology Letters 6* (4): 458–461. DOI:10.1098/rsbl.2009.1081

Chan, A. A. Y. H., W. D. Stahlman, D. Garlick, C. D. Fast, D. T. Blumstein, and A. P. Blaisdell. (2010b). Increased amplitude and duration of acoustic stimuli enhance distraction. *Animal Behaviour 80* 1075–1079.

Chan, I., L. C. Tseng, S. Ka, C. F. Chang, and J. S. Hwang. (2012). An experimental study of the response of the gorgonian coral *Subergorgia suberosa* to polluted seawater from a former coastal mining site in Taiwan. *Zoological Studies 51* (1): 27–37.

Chapman, C. J. and A. D. Hawkins. (1973). Field study of hearing in cod, *Gadus morhua* L. *Journal of Comparative Physiology 85* (2): 147–167. DOI:10.1007/bf00696473

Chappell, S. (2018, August 23, 2018). Personal commuication to Combs, R. on AFTT Invertebrates Text.

Charifi, M., M. Sow, P. Ciret, S. Benomar, and J. Massabuau. (2017). The sense of hearing in the Pacific oyster, *Magallana gigas*. *PLoS ONE 12* (10): 19.

Chaturvedi, S. and P. N. Dave. (2015). Solid propellants: AP/HTPB composite propellants. *Arabian Journal of Chemistry web*. DOI:10.1016/j.arabjc.2014.12.033

Chavez, E. (2009). *2008 San Diego Bay Eelgrass Inventory and Bathymetry Update*. San Diego, CA: San Diego Unified Port District Environmental Advisory Committee.

Chesapeake Biological Laboratory. (1948). *Effects of Underwater Explosions on Oysters, Crabs and Fish: a Preliminary Report*. Solomons Island, MD: Department of Research and Education.

Chiarelli, R. and M. C. Roccheri. (2014). Marine invertebrates as bioindicators of heavy metal pollution. *Open Journal of Metal 4* 93–106. DOI:http://dx.doi.org/10.4236/ojmetal.2014.44011

Choy, C. A. and J. C. Drazen. (2013). Plastic for dinner? Observations of frequent debris ingestion by pelagic predatory fishes from the central North Pacific. *Marine Ecology Progress Series 485* 155–163. DOI:10.3354/meps10342

Christian, J. R., A. Mathieu, D. H. Thomson, D. White, and R. A. Buchanan. (2003). *Effect of Seismic Energy on Snow Crab (Chionoecetes opilio)* (Environmental Research Funds Report). Calgary, Canada: LGL Ltd., Environmental Research Associates.

Christiansen, F., S. M. Dawson, J. W. Durban, H. Fearnbach, C. A. Miller, L. Bejder, M. Uhart, M. Sironi, P. Corkeron, W. Rayment, E. Leunissen, E. Haria, R. Ward, H. A. Warick, I. Kerr, M. S. Lynn, H. M. Pettis, and M. J. Moore. (2020). Population comparison of right whale body condition reveals poor state of the North Atlantic right whale. *Marine Ecology Progress Series 640* 1–16.

Chuenpagdee, R., L. E. Morgan, S. M. Maxwell, E. A. Norse, and D. Pauly. (2003). Shifting gears: Addressing the collateral impacts of fishing methods in U.S. waters. *Frontiers in Ecology and the Environment 1* (10): 517–524.

Clarke, M. E., C. E. Whitmire, and M. M. Yoklavich. (2015). *State of Deep-Sea Coral and Sponge Ecosystems of the U.S. West Coast: 2015* (Vol. X NOAA Technical Memorandum). Silver Spring, MD: National Oceanic and Atmospheric Administration.

Cloyed, C. S., E. E. Hieb, M. K. Collins, K. P. DaCosta, and R. H. Carmichael. (2019). Linking use of ship channels by West Indian manatees (*Trichechus manatus*) to seasonal migration and habitat use. *Frontiers in Marine Science 6*. DOI:10.3389/fmars.2019.00318

Cole, M., P. Lindeque, E. Fileman, C. Halsband, R. Goodhead, J. Moger, and T. S. Galloway. (2013). Microplastic ingestion by zooplankton. *Environmental Science & Technology 47* (12): 6646–6655. DOI:10.1021/es400663f

Coles, S. L., R. C. DeFelice, L. G. Eldredge, J. T. Carlton, R. L. Pyle, and A. Suzumoto. (1997). *Biodiversity of Marine Communities in Pearl Harbor, Oahu, Hawaii with Observations on Introduced Exotic Species* (Bishop Museum Technical Report No. 10). Honolulu, HI: Bishop Museum Press.

Collin, S. P. and D. Whitehead. (2004). The functional roles of passive electroreception in non-electric fishes. *Animal Biology 54* (1): 1–25.

Conn, P. B. and G. K. Silber. (2013). Vessel speed restrictions reduce risk of collision-related mortality for North Atlantic right whales. *Ecosphere 4* (4): 1–16. DOI:10.1890/es13-00004.1

Conomy, J. T., J. A. Dubovsky, J. A. Collazo, and W. J. Fleming. (1998). Do black ducks and wood ducks habituate to aircraft disturbance? *Journal of Wildlife Management 62* (3): 1135–1142.

Cornell Lab of Ornithology. (2009). *All About Birds: Herring Gull*. Retrieved 2011, from http://www.allaboutbirds.org/guide/Herring_Gull/id.

Couturier, L. I. E., A. D. Marshall, F. R. A. Jaine, T. Kashiwagi, S. J. Pierce, K. A. Townsend, S. J. Weeks, M. B. Bennett, and A. J. Richardson. (2012). Biology, ecology and conservation of the Mobulidae. *Journal of Fish Biology 80* 1075–1119.

Craig, J. C., Jr. (2001). *Appendix D, Physical Impacts of Explosions on Marine Mammals and Turtles Final Environmental Impact Statement, Shock Trial of the Winston Churchill (DDG 81)*. Washington, DC: U.S. Department of the Navy, Naval Sea Systems Command.

Crocker, F. H., K. J. Indest, and H. L. Fredrickson. (2006). Biodegradation of the cyclic nitramine explosives RDX, HMX, and CL-20. *Applied Microbiology and Biotechnology 73* 274–290. DOI:10.1007/s00253-006-0588-y

Crowell, S. C. (2016). Measuring in-air and underwater hearing in seabirds. *Advances in Experimental Medicine and Biology 875* 1155–1160. DOI:10.1007/978-1-4939-2981-8_144

Crum, L., M. Bailey, J. Guan, P. Hilmo, S. Kargl, and T. Matula. (2005). Monitoring bubble growth in supersaturated blood and tissue *ex vivo* and the relevance to marine mammal bioeffects. *Acoustics Research Letters Online 6* (3): 214–220. DOI:10.1121/1.1930987

Crum, L. and Y. Mao. (1996). Acoustically enhanced bubble growth at low frequencies and its implications for human diver and marine mammal safety. *The Journal of the Acoustical Society of America 99* (5): 2898–2907.

Cruz-Uribe, O., D. P. Cheney, and G. L. Rorrer. (2007). Comparison of TNT removal from seawater by three marine macroalgae. *Chemsphere 67* 1469–1476. DOI:10.1016/j.chemosphere.2007.01.001

Cucurachi, S., W. Tamis, M. Vijver, W. Peijnenburg, J. Bolte, and G. Snoo. (2013). A review of the ecological effects of radiofrequency electromagnetic fields (RF-EMF). *Environmental International 51* 116–140.

Currie, J. J., S. H. Stack, and G. D. Kaufman. (2017). Modelling whale-vessel encounters: The role of speed in mitigating collisions with humpback whales (*Megaptera novaeangliae*). *Journal of Cetacean Research Management 17* 57–63.

Cusano, D. A., L. A. Conger, S. M. Van Parijs, and S. E. Parks. (2019). Implementing conservation measures for the North Atlantic right whale: considering the behavioral ontogeny of mother-calf pairs. *Animal Conservation 22* (3): 228–237. DOI:https://doi.org/10.1111/acv.12457

Czech-Damal, N. U., A. Liebschner, L. Miersch, G. Klauer, F. D. Hanke, C. Marshall, G. Dehnhardt, and W. Hanke. (2011). Electroreception in the Guiana dolphin (*Sotalia guianensis*). *Proceedings of the Royal Society B: Biological Sciences 279* (1729): 663–668. DOI:10.1098/rspb.2011.1127

Damon, E. G., D. R. Richmond, E. R. Fletcher, and R. K. Jones. (1974). *The Tolerance of Birds to Airblast* (Contract Number DASA 01-70-C-0075). Springfield, VA: Lovelace Foundation for Medical Education and Research.

Danil, K. and J. A. St Leger. (2011). Seabird and dolphin mortality associated with underwater detonation exercises. *Marine Technology Society Journal 45* (6): 89–95.

Danner, G. R., J. Chacko, and F. Brautigam. (2009). Voluntary ingestion of soft plastic fishing lures affects brook trout growth in the laboratory. *North American Journal of Fisheries Management 29* (2): 352–360. DOI:10.1577/M08-085.1

Dantas, D. V., M. Barletta, and M. F. da Costa. (2012). The seasonal and spatial patterns of ingestion of polyfilament nylon fragments by estuarine drums (Sciaenidae). *Environmental Science and Pollution Research 19* (2): 600–606.

Dasgupta, P. K., P. K. Martinelango, W. A. Jackson, T. A. Anderson, K. Tian, R. W. Tock, and S. Rajagopalan. (2005). The origin of naturally occurring perchlorate: The role of atmospheric processes. *Environmental Science and Technology 39* (6): 1569–1575. DOI:10.1021/es048612x

Davis, A. R. (2009). The role of mineral, living and artificial substrata in the development of subtidal assemblages. In M. Wahl (Ed.), *Marine Hardbottom Communities: Patterns, Dynamics, Diversity and Change* (Vol. 206, pp. 19–37). New York, NY: Springer-Verlag.

Davis, A. R., A. Broad, W. Gullett, J. Reveley, C. Steele, and C. Schofield. (2016). Anchors away? The impacts of anchor scour by ocean-going vessels and potential response options. *Marine Policy 73* 1–7.

Davison, P. and R. G. Asch. (2011). Plastic ingestion by mesopelagic fishes in the North Pacific Subtropical Gyre. *Marine Ecological Progress Series 432* 173–180.

Dawes, C. J., J. Andorfer, C. Rose, C. Uranowski, and N. Ehringer. (1997). Regrowth of the seagrass, *Thalassia testudinum,* into propeller scars. *Aquatic Botany 59* (1–2): 139–155. DOI:10.1016/s0304-3770(97)00021-1

Day, R. D., R. D. McCauley, Q. P. Fitzgibbon, and J. M. Semmens. (2016). Seismic air gun exposure during early-stage embryonic development does not negatively affect spiny lobster *Jasus edwardsii* larvae (Decapoda: Palinuridae). *Scientific Reports 6* (22723): 9. DOI:10.1038/srep22723

Deakos, M. H., J. D. Baker, and L. Bejder. (2011). Characteristics of a manta ray *Manta alfredi* population off Maui, Hawaii, and implications for management. *Marine Ecology Progress Series 429* 245–260.

Dempster, T. and M. Taquet. (2004). Fish aggregation device (FAD) research: Gaps in current knowledge and future direction for ecological studies. *Reviews in Fish Biology and Fisheries 14* (1): 21–42.

Derraik, J. G. B. (2002). The pollution of the marine environment by plastic debris: A review. *Marine Pollution Bulletin 44* 842–852.

Dietrich, K. and E. Melvin. (2004). *Annotated Bibliography: Seabird Interactions with Trawl Fishing Operations and Cooperative Research* (Washington Sea Grant Program). Seattle, WA: University of Washington Board of Regents.

Donnelly-Greenan, E., J. Harvey, H. Nevins, M. Hester, and W. Walker. (2014). Prey and plastic ingestion of Pacific Northern Fulmars (*Fulmarus glacialis rogersii*) from Monterey Bay, California. *Marine Pollution Bulletin 85* 214–224. DOI:10.1016/j.marpolbul.2014.05.046

Dooling, R. J. (1980). Behavior and Psychophysics of Hearing in Birds. In A. N. Popper & R. R. Fay (Eds.), *Comparative Studies of Hearing in Vertebrates* (pp. 261–288). New York, NY: Springer-Verlag.

Dooling, R. J. and A. N. Popper. (2000). Hearing in birds and reptiles. In R. J. Dooling, R. R. Fay, & A. N. Popper (Eds.), *Comparative Hearing in Birds and Reptiles* (Vol. 13, pp. 308–359). New York, NY: Springer-Verlag.

Dooling, R. J. and S. C. Therrien. (2012). Hearing in birds: What changes from air to water. *Advances in Experimental Medicine and Biology 730* 77–82. DOI:10.1007/978-1-4419-7311-5_17

Dossot, G. A., J. S. Krumholz, D. M. Hudson, D. L. Pochtar, N. C. Dickenso, E. B. Baker, and T. E. Moll. (2017). *Simulated anthropogenic noise exposure to marine invertebrates using a standing wave tube* [Type]. Presented at the Acoustical Society of America Meeting. New Orleans, LA.

Douglas, A. B., J. Calambokidis, S. Raverty, S. J. Jeffries, D. M. Lambourn, and S. A. Norman. (2008). Incidence of ship strikes of large whales in Washington State. *Journal of the Marine Biological Association of the United Kingdom 88* (6): 1121–1132. DOI:10.1017/S0025315408000295

Durant, J. M., T. Anker-Nilssen, and N. C. Stenseth. (2003). Trophic interaction under climate fluctuations: The Atlantic puffin as an example. *Proceedings of the Royal Society of London 270(B)* (1): 461–466.

Dyndo, M., D. M. Wisniewska, L. Rojano-Donate, and P. T. Madsen. (2015). Harbour porpoises react to low levels of high frequency vessel noise. *Scientific Reports 5* 11083. DOI:10.1038/srep11083

Edwards, M. H., S. M. Shjegstad, R. Wilkens, J. C. King, G. Carton, D. Bala, B. Bingham, M. C. Bissonnette, C. Briggs, N. S. Bruso, R. Camilli, M. Cremer, R. B. Davis, E. H. DeCarlo, C. DuVal, D. J. Fornari, I. Kaneakua-Pia, C. D. Kelley, S. Koide, C. L. Mah, T. Kerby, G. J. Kurras, M. R. Rognstad, L. Sheild, J. Silva, B. Wellington, and M. V. Woerkom. (2016). The Hawaii undersea military munitions assessment. *Deep Sea Research Part II: Topical Studies in Oceanography 128* 4–13. DOI:10.1016/j.dsr2.2016.04.011

Ellis, D. H. (1981). *Responses of Raptorial Birds to Low Level Military Jets and Sonic Booms* (Results of the 1980-1981 joint U.S. Air Force-U.S. Fish and Wildlife Service Study). Oracle, AZ: Institute for Raptor Studies.

Elvin, S. S. and C. T. Taggart. (2008). Right whales and vessels in Canadian waters. *Marine Policy 32* 379–386. DOI:doi:10.1016/j.marpol.2007.08.001

Environmental Sciences Group. (2005a). *Canadian Forces Maritime Experimental and Test Range Environmental Assessment Update 2005*. Kingston, Canada: Environmental Sciences Group, Royal Military College.

Environmental Sciences Group. (2005b). *Canadian Forces Maritime Experimental and Test Ranges: Environmental Assessment Update 2005*. Kingston, Canada: Environmental Sciences Group, Royal Military College.

Erbe, C., C. Reichmuth, K. Cunningham, K. Lucke, and R. Dooling. (2016). Communication masking in marine mammals: A review and research strategy. *Marine Pollution Bulletin 103* (1–2): 15–38. DOI:10.1016/j.marpolbul.2015.12.007

Eriksson, C. and H. Burton. (2003). Origins and biological accumulation of small plastic particles in fur seals from Macquarie Island. *Ambio 32* (6): 380–384.

Espinoza, M. and I. S. Wehrtmann. (2008). Stomach content analyses of the threadfin anglerfish *Lophiodes spilurus* (Lophiiformes: Lophiidae) associated with deepwater shrimp fisheries from the central Pacific of Costa Rica. *Revista de Biología Tropical 56* (4): 1959–1970.

European Flame Retardants Association. (2002). *Flame retardant fact sheet: Red phosphorus*. Retrieved February 24, 2011, from http://www.cefic-efra.com/Objects/2/Files/RedPhosphorusFactSheet.pdf.

Fahlman, A., P. L. Tyack, P. J. O. Miller, and P. H. Kvadsheim. (2014). How man-made interference might cause gas bubble emboli in deep diving whales. *Frontiers in Physiology 5* (13): 1–6. DOI:10.3389/fphys.2014.00013

Farrell, R. E. and S. D. Siciliano. (2007). *Environmental Effects of Radio Frequency Chaff Released During Military Training Exercises: A Review of the Literature*. Saskatoon, Canada: Goose Bay Office of the Department of National Defense.

Favero, M., G. Blanco, G. Garcia, S. Copello, J. P. S. Pon, E. Frere, F. Quintana, P. Yorio, F. Rabuffetti, G. Canete, and P. Gandini. (2011). Seabird mortality associated with ice trawlers in the Patagonian shelf: Effect of discards on the occurrence of interactions with fishing gear. *Animal Conservation 14* (2): 131–139.

Felix, F. and K. Van Waerebeek. (2005). Whale mortality from ship strikes in Ecuador and West Africa. *Latin American Journal of Aquatic Mammals 4* (1): 55–60. DOI:10.5597/lajam00070

Fewtrell, J. L. and R. D. McCauley. (2012). Impact of air gun noise on the behaviour of marine fish and squid. *Marine Pollution Bulletin 64* (5): 984–993. DOI:10.1016/j.marpolbul.2012.02.009

Fields, R. D. (2007). The Shark's Electric Sense: An astonishingly sensitive detector of electric fields helps sharks zero in on prey. *Scientific American 297* (2): 75–81.

Filiciotto, F., M. Vazzana, M. Celi, V. Maccarrone, M. Ceraulo, G. Buffa, V. Arizza, G. de Vincenzi, R. Grammauta, S. Mazzola, and G. Buscaino. (2016). Underwater noise from boats: Measurement of its influence on the behaviour and biochemistry of the common prawn (*Palaemon serratus*, Pennant 1777). *Journal of Experimental Marine Biology and Ecology 478* 24–33. DOI:10.1016/j.jembe.2016.01.014

Filiciotto, F., M. Vazzana, M. Celi, V. Maccarrone, M. Ceraulo, G. Buffa, V. Di Stefano, S. Mazzola, and G. Buscaino. (2014). Behavioural and biochemical stress responses of *Palinurus elephas* after exposure to boat noise pollution in tank. *Marine Pollution Bulletin 84* (1–2): 104–114. DOI:10.1016/j.marpolbul.2014.05.029

Finneran, J. J. (2015). Noise-induced hearing loss in marine mammals: A review of temporary threshold shift studies from 1996 to 2015. *The Journal of the Acoustical Society of America 138* (3): 1702–1726. DOI:10.1121/1.4927418

Firestone, J. (2009). Policy considerations and measures to reduce the likelihood of vessel collisions with great whales. *Environmental Affairs 36* 389–400.

Fisher, H. I. (1971). Experiments on homing in Laysan Albatrosses, *Diomedea immutabilis*. *Condor 73* (4): 389–400.

Fitzpatrick, J. L., J. K. Desjardins, K. A. Stiver, R. Montgomerie, and S. Balshine. (2006). Male reproductive suppression in the cooperatively breeding fish *Neolamprologus pulcher*. *Behavioural Ecology 17* 25–33.

Foderaro, L. W. (2015). Group Petitions to Save a Prehistoric Fish From Modern Construction. *New York Times*. Retrieved from http://www.nytimes.com/2015/07/22/nyregion/group-petitions-to-save-a-prehistoric-fish-from-modern-construction.html?_r=0.

Foley, H. J., R. C. Holt, R. E. Hardee, P. B. Nilsson, K. A. Jackson, A. J. Read, D. A. Pabst, and W. A. McLellan. (2011). Observations of a western North Atlantic right whale (*Eubalaena glacialis*) birth offshore of the protected southeast U.S. critical habitat. *Marine Mammal Science 27* (3): E234–E240. DOI:10.1111/j.1748-7692.2010.00452

Fonnesbeck, C. J., L. P. Garrison, L. I. Ward-Geiger, and R. D. Baumstark. (2008). Bayesian hierarchichal model for evaluating the risk of vessel strikes on North Atlantic right whales in the SE United States. *Endangered Species Research 6* 87–94. DOI:10.3354/esr00134

Forrester, D. J., F. H. White, J. C. Woordard, and N. P. Thompson. (1975). Intussusception in a Florida manatee. *Journal of Wildlife Diseases 11* 566–568.

Fossette, S., S. Ferraroli, H. Tanaka, Y. Ropert-Coudert, N. Arai, K. Sato, Y. Naito, Y. Le Maho, and J. Georges. (2007). Dispersal and dive patterns in gravid leatherback turtles during the nesting season in French Guiana. *Marine Ecology Progress Series 338* 233–247.

Fossette, S., A. C. Gleiss, J. Chalumeau, T. Bastian, C. D. Armstrong, S. Vandenabeele, M. Karpytchev, and G. C. Hays. (2015). Current-oriented swimming by jellyfish and its role in bloom maintenance. *Current Biology 25* 342–347.

Fox, H. E. and R. L. Caldwell. (2006). Recovery from blast fishing on coral reefs: A tale of two scales. *Ecological Applications 16* (5): 1631–1635.

Fujiwara, M. and H. Caswell. (2001). Demography of the endangered North Atlantic right whale. *Nature 414* 537–541. DOI:10.1038/35107054

Furin, C. G., F. A. von Hippel, B. Hagedorn, and T. M. O'Hara. (2013). Perchlorate trophic transfer increases tissue concentrations above ambient water exposure alone in a predatory fish. *Journal of Toxicology and Environmental Health. Part A 76* (18): 1072–1084. DOI:10.1080/15287394.2013.836693

Gannier, A. and G. Marty. (2015). Sperm whales ability to avoid approaching vessels is affected by sound reception in stratified waters. *Marine Pollution Bulletin 95* (1): 283–288. DOI:10.1016/j.marpolbul.2015.03.029

Gaspin, J. B., G. B. Peters, and M. L. Wisely. (1976). *Experimental Investigations of the Effects of Underwater Explosions on Swimbladder Fish*. Silver Spring, MD: Naval Ordnance Lab.

Gehring, J., P. Kerlinger, and A. M. Manville, II. (2009). Communication towers, lights, and birds: Successful methods of reducing the frequency of avian collisions. *Ecological Applications 19* (2): 505–514.

Gende, S. M., A. N. Hendrix, K. R. Harris, B. Eichenlaub, J. Nielsen, and S. Pyare. (2011). A Bayesian approach for understanding the role of ship speed in whale-ship encounters. *Ecological Applications 21* (6): 2232–2240.

Germanov, E. S. and A. D. Marshall. (2014). Running the gauntlet: Regional movement patterns of *Manta alfredi* through a complex of parks and fisheries. *PLoS ONE 9* (10): e110071.

Gerstein, E. R. (2002). Manatees, bioacoustics and boats: Hearing tests, environmental measurements and acoustic phenomena may together explain why boats and animals collide. *American Scientist 90* (2): 154–163. DOI:10.1511/2002.2.154

Gerstein, E. R., J. E. Blue, and S. E. Forysthe. (2005). The acoustics of vessel collisions with marine mammals. *Proceedings of OCEANS 2005 MTS/IEEE 2* 1190–1197. DOI:10.1109/OCEANS.2005.1639917

Gill, A. B. (2005). Offshore renewable energy: Ecological implications of generating electricity in the coastal zone. *Journal of Applied Ecology 42* 605–615.

Gill, A. B., I. Gloyne-Philips, J. Kimber, and P. Sigray. (2014). Marine Renewable Energy, Electromagnetic (EM) Fields and EM-Sensitive Animals. In M. Shields & A. Payne (Eds.), *Marine Renewable Energy Technology and Environmental Interactions. Humanity and the Sea* (pp. 61–79). Dordrecht, Netherlands: Springer.

Goldstein, M. C. and D. S. Goodwin. (2013). Gooseneck barnacles (Lepas spp.) ingest microplastic debris in the North Pacific Subtropical Gyre. *PeerJ 1* e184. DOI:10.7717/peerj.184

Goodwin, S. E. and J. Podos. (2013). Shift of song frequencies in response to masking tones. *Animal Behaviour 85* 435–440. DOI:10.1016/j.anbehav.2012.12.003

Goreau, T. J. (2014). Electrical stimulation greatly increases settlement, growth, survival, and stress resistance of marine organisms. *Natural Resources 5* 527–537.

Gorodilov, L. V. and A. P. Sukhotin. (1996). Experimental investigation of craters generated by explosions of underwater surface charges on sand. *Combustion, Explosion, and Shock Waves 32* (3): 344–346.

Goudie, R. I. and I. L. Jones. (2004). Dose-response relationships of harlequin duck behavior to noise from low-level military jet over-flights in central Labrador. *Environmental Conservation 31* (4): 289–298.

Graham, R. T., M. J. Witt, D. W. Castellanos, F. Remolina, S. Maxwell, B. J. Godley, and L. A. Hawkes. (2012). Satellite tracking of manta rays highlights challenges to their conservation. *PLoS ONE 7* (5): e36834.

Greene, G. D., F. R. Engelhardt, and R. J. Paterson. (1985). *Proceedings of the Workshop on Effects of Explosives Use in the Marine Environment*. Aberdeen, Canada: Canada Oil and Gas Lands Administration, Environmental Protection Branch.

Griscom, S. B. and N. S. Fisher. (2004). Bioavailability of Sediment-bound Metals to Marine Bivalve Molluscs: An Overview. *Estuaries 27* (5): 826–838.

Grizzle, R. E., J. R. Adams, and L. J. Walters. (2002). Historical changes in intertidal oyster (*Crassostrea virginica*) reefs in a Florida lagoon potentially related to boating activities. *Journal of Shellfish Research 21* (2): 749–756.

Guerra, A. and A. F. Gonzales. (2006). *Severe injuries in the giant squid, Architeuthis dux, stranded after seismic explorations* [Type]. Presented at the International Workshop: Impacts of seismic survey activities on whales and other marine biota. Dessau, Germany.

Guerra, A., A. F. Gonzalez, F. Rocha, J. Gracia, and M. Vecchione. (2004). Calamares gigantes varados. *Investigacion y Ciencia* 35–37.

Guest, J., M. Heyward, M. Omori, K. Iwao, A. Morse, and C. Boch. (2010). Rearing coral larvae for reef rehabilitation. In A. J. Edwards (Ed.), *Reef Rehabilitation Manual* (pp. 73–98). St. Lucia, Australia: The Coral Reef Targeted Research & Capacity Building for Management Program.

Haase, B. and F. Felix. (1994). A note on the incidental mortality of sperm whales (*Physeter macrocephalus*) in Ecuador. *Report of the International Whaling Commission* (Special Issue 15): 481–483.

Haftorn, S., F. Mehlum, and C. Bech. (1988). Navigation to nest site in the snow petrel (*Pagodroma nivea*). *The Condor 90* (2): 484–486.

Hain, J. H. W., S. L. Ellis, R. D. Kenney, P. J. Clapham, B. K. Gray, M. T. Weinrich, and I. G. Babb. (1995). Apparent bottom feeding by humpback whales on Stellwagen Bank. *Marine Mammal Science 11* (4): 464–479.

Hall, N. M., K. L. E. Berry, L. Rintoul, and M. O. Hoogenboom. (2015). Microplastic ingestion by scleractinian corals. *Marine Biology 162* 725–732. DOI:10.1007/s00227-015-2619-7

Halliday, W. D. (2020). *Literature Review of Ship Strike Risk to Whales*. Toronto, Canada: Wildlife Conservation Society Canada.

Halpern, B. S., K. L. McLeod, A. A. Rosenberg, and L. B. Crowder. (2008). Managing for cumulative impacts in ecosystem-based management through ocean zoning. *Ocean & Coastal Management 51* (3): 203–211. DOI:10.1016/j.ocecoaman.2007.08.002

Halvorsen, K. M. and E. O. Keith. (2008). Immunosuppression cascade in the Florida Manatee (*Trichechus manatus latirostris*). *Aquatic Mammals 34* (4): 412–419. DOI:10.1578/AM.34.4.2008.412

Hamilton, P. K. and S. D. Kraus. (2019). Frequent encounters with the seafloor increase right whales risk of entanglement in fishing groundlines. *Endangered Species Research 39* 235–246.

Hamilton, W. J., III. (1958). Pelagic birds observed on a North Pacific crossing. *The Condor 60* (3): 159–164.

Hanlon, R. T. and J. B. Messenger. (1996). *Cephalopod Behaviour*. Cambridge, United Kingdom: Cambridge University Press.

Hardesty, B. D. and C. Wilcox. (2017). A risk framework for tackling marine debris. *Royal Society of Chemistry 9* 1429–1436. DOI:10.1039/c6ay02934e

Harter, S. L., M. M. Ribera, A. N. Shepard, and J. K. Reed. (2009). Assessment of fish populations and habitat on Oculina Bank, a deep-sea coral marine protected area off eastern Florida. *Fishery Bulletin 107* (2): 195–206.

Hartwell, S. I., C. H. Hocutt, and W. F. van Heukelem. (1991). Swimming response of menhaden (*Brevoortia tyrannus*) to electromagnetic pulses. *Journal of Applied Ichthyology 7* (2): 90–94.

Harwood, L. A., F. McLaughlin, R. M. Allen, J. J. Illasiak, and A. Alikamik. (2005). First-ever marine mammal and bird observations in the deep Canada Basin and Beaufort/Chukchi Seas: expeditions during 2002. *Polar Biology 28* 250–253. DOI:10.1007/s00300-004-0691-4

Hashino, E., M. Sokabe, and K. Miyamoto. (1988). Frequency specific susceptibility to acoustic trauma in the buderigar (*Melopsittacus undulatus*). *The Journal of the Acoustical Society of America 83* (6): 2450–2453.

Haubold, E. M., C. Deutsch, and C. Fonnesbeck. (2006). *Final Biological Status Review of the Florida Manatee (Trichechus manatus latirostris)*. St. Petersburg, FL: Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute.

Haugland, M., J. C. Holst, M. Holm, and L. P. Hansen. (2006). Feeding of Atlantic salmon (*Salmo salar* L.) post-smolts in the Northeast Atlantic. *ICES Journal of Marine Science 63* 1488–1500.

Hawkins, A. D. and A. D. F. Johnstone. (1978). The hearing of the Atlantic salmon, *Salmo salar*. *Journal of Fish Biology 13* 655–673.

Hayes, P. K., N. A. El Semary, and P. Sanchez-Baracaldo. (2007). The taxonomy of cyanobacteria: Molecular insights into a difficult problem. In J. Brodie & J. Lewis (Eds.), *Unravelling the Algae: The Past, Present, and Future of Algal Systematics* (pp. 93–102). Boca Raton, FL: CRC Press.

Hazel, J., I. R. Lawler, and M. Hamann. (2009). Diving at the shallow end: Green turtle behaviour in near-shore foraging habitat. *Journal of Experimental Marine Biology and Ecology 371* (1): 84–92.

Hazel, J., I. R. Lawler, H. Marsh, and S. Robson. (2007). Vessel speed increases collision risk for the green turtle *Chelonia mydas*. *Endangered Species Research 3* 105–113.

Heezen, B. C. (1957). Whales entangled in deep sea cables. *Deep Sea Research 4* (2): 105–115.

Helfman, G. S., B. B. Collette, D. E. Facey, and B. W. Bowen. (2009). *The Diversity of Fishes: Biology, Evolution, and Ecology* (2nd ed.). Malden, MA: Wiley-Blackwell.

Henderson, D., E. C. Bielefeld, K. C. Harris, and B. H. Hu. (2006). The role of oxidative stress in noise-induced hearing loss. *Ear & Hearing 27* 1–19.

Hennessy, M. B., J. P. Heybach, J. Vernikos, and S. Levine. (1979). Plasma corticosterone concentrations sensitively reflect levels of stimulus intensity in the rat. *Physiology and Behavior 22* 821–825.

Henry, A. G., T. V. N. Cole, L. Hall, W. Ledwell, D. Morin, and A. Reid. (2016). *Serious Injury and Mortality Determinations for Baleen Whale Stocks along the Gulf of Mexico, United States East Coast and Atlantic Canadian Provinces, 2010–2014*. Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Henry, P. Y., G. Wey, and G. Balanca. (2011). Rubber band ingestion by a rubbish dump dweller, the white stork (*Ciconia ciconia*). *Waterbirds 34* (4): 504–508.

Hetherington, T. (2008). Comparative anatomy and function of hearing in aquatic amphibians, reptiles, and birds. In J. G. M. Thewissen & S. Nummela (Eds.), *Sensory Evolution on the Threshold* (pp. 182–209). Berkeley, CA: University of California Press.

Hewitt, R. P. (1985). Reaction of dolphins to a survey vessel: Effects on census data. *Fishery Bulletin 83* (2): 187–193.

Heyward, A. J. and A. P. Negri. (2012). Turbulence, cleavage, and the naked embryo: A case for coral clones. *Science 335* 1064. DOI:10.1126/science.1216055

Hillman, M. D., S. M. Karpanty, J. D. Fraser, and A. Derose-Wilson. (2015). Effects of aircraft and recreation on colonial waterbird nesting behavior. *Journal of Wildlife Management 79* (7): 1192–1198. DOI:10.1002/jwmg.925

Hochscheid, S. (2014). Why we mind sea turtles' underwater business: A review on the study of diving behavior. *Journal of Experimental Marine Biology and Ecology 450* 118–136.

Hofmeyr, G. J. G., M. N. Bester, S. P. Kirkman, C. Lydersen, and K. M. Kovacs. (2006). Entanglement of Antarctic fur seals at Bouvetoya, Southern Ocean. *Marine Pollution Bulletin 52* 1077–1080. DOI:10.1016/j.marpolbul.2006.05.003

Hore, P. J. (2012). Are biochemical reactions affected by weak magnetic fields? *Proceedings of the National Academy of Sciences 109* (5): 1357–1358. DOI:10.1073/pnas.1120531109

Horst, T. J. (1977). Use of Leslie Matrix for assessing environmental-impact with an example for a fish population. *Transactions of the American Fisheries Society 106* (3): 253–257.

Horton, T. W., N. Hauser, A. N. Zerbini, M. P. Francis, M. L. Domeier, A. Andriolo, D. P. Costa, P. W. Robinson, C. A. J. Duffy, N. Nasby-Lucas, R. N. Holdaway, and P. J. Clapham. (2017). Route fidelity during marine megafauna migration. *Frontiers in Marine Science 4* 1–21. DOI:10.3389/fmars.2017.00422

Hoss, D. E. and L. R. Settle. (1990). *Ingestion of plastics by teleost fishes* (Proceedings of the Second International Conference on Marine Debris). Honolulu, HI: National Oceanic and Atmospheric Administration.

Houser, D. S., L. A. Dankiewicz-Talmadge, T. K. Stockard, and P. J. Ponganis. (2009). Investigation of the potential for vascular bubble formation in a repetitively diving dolphin. *The Journal of Experimental Biology 213* 52–62. DOI:10.1242/jeb.028365

Hughes, A. R., D. A. Mann, and D. L. Kimbro. (2014). Predatory fish sounds can alter crab foraging behavior and influence bivalve abundance. *Proceedings of the Royal Society Biological Sciences 281* (20140715).

Huveneers, C., P. J. Rogers, J. M. Semmens, C. Beckmann, A. A. Kock, B. Page, and S. D. Goldsworthy. (2013). Effects of an electric field on white sharks: In situ testing of an electric deterrent. *PLoS ONE 8* (5): e62730. DOI:10.1371/journal.pone.0062730

Hyrenbach, K. (2001). Albatross response to survey vessels: Implications for studies of the distribution, abundance, and prey consumption of seabird populations. *Marine Ecology Progress Series 212* 283–295.

Hyrenbach, K. (2006, 2 & 3 October). *Training and Problem-Solving to Address Population Information Needs for Priority Species, Pelagic Species and Other Birds at Sea*. Presented at the Waterbird Monitoring Techniques Workshop, IV North American Ornithological Conference. Veracruz, Mexico.

Hyrenbach, K., M. Hester, S. Lyday, S. Youngren, D. Rapp, M. Dwyer, E. Robinson, P. Hartzell, M. Duhr-Shultz, and M. Fry. (2015, October 26–30). *Overview: Developing Ecosystem Metrics for Plastic Ingestion by Hawaiian Seabirds*. Presented at the World Seabird Conference II. Cape Town, South Africa.

Iversen, M., J. Aars, T. Haug, I. G. Alsos, C. Lydersen, L. Bachman, and K. M. Kovacs. (2013). The diet of polar bears (*Usus maritimus*) from Svalbard, Norway, inferred from scat analysis. *Polar Biology 36* 561–571. DOI:10.1007/s00300-012-1284-2

Jackson, G. D., N. G. Buxton, and M. J. A. George. (2000). Diet of the southern opah, *Lampris immaculatus,* on the Patagonian Shelf; the significance of the squid, *Moroteuthis ingens,* and anthropogenic plastic. *Marine Ecology Progress Series 206* 261–271. DOI:10.3354/meps206261

Jacobsen, J. K., L. Massey, and F. Gulland. (2010). Fatal ingestion of floating net debris by two sperm whales (*Physeter macrocephalus*). *Marine Pollution Bulletin 60* (5): 765–767. DOI:10.1016/j.marpolbul.2010.03.008

James, M. C. and T. B. Herman. (2001). Feeding of *Dermochelys coriacea* on medusae in the northwest Atlantic. *Chelonian Conservation and Biology 4* (1): 202–205.

Jaquet, N. and H. Whitehead. (1996). Scale-dependent correlation of sperm whale distribution with environmental features and productivity in the South Pacific. *Marine Ecology Progress Series 135* 1–9.

Jennings, S. and M. J. Kaiser. (1998). The effects of fishing on marine ecosystems. In J. H. S. Blaxter, A. J. Southward, & P. A. Tyler (Eds.), *Advances in Marine Biology* (Vol. 34, pp. 201–352). Warwick, United Kingdom: Academic Press.

Jensen, A. S. and G. K. Silber. (2004). *Large Whale Ship Strike Database* (NOAA Technical Memorandum NMFS-OPR-25). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Johnson, A. and A. Acevedo-Gutiérrez. (2007). Regulation compliance by vessels and disturbance of harbour seals (*Phoca vitulina*). *Canadian Journal of Zoology 85* 290–294.

Johnson, A., G. Salvador, J. Kenney, J. Robbins, S. Kraus, S. Landry, and P. Clapham. (2005). Fishing gear involved in entanglements of right and humpback whales. *Marine Mammal Science 21* (4): 635–645.

Johnson, R. J., P. H. Cole, and W. W. Stroup. (1985). Starling response to three auditory stimuli. *Journal of Wildlife Management 49* (3): 620–625.

Johnston, D. W., J. Frungillo, A. Smith, K. Moore, B. Sharp, J. Schuh, and A. J. Read. (2015). Trends in stranding and by-catch rates of gray and harbor seals along the northeastern coast of the United States: evidence of divergence in the abundance of two sympatric phocid species? *PLoS ONE 10* (7): e0131660. DOI:10.1371/journal.pone.0131660

Jompa, J., Suharto, E. M. Anpusyahnur, P. N. Dwjja, J. Subagio, I. Alimin, R. Anwar, S. Syamsuddin, T. Heni, U. Radiman, H. Triyono, R. A. Sue, and N. Soeyasa. (2012). Electrically Stimulated Corals in Indonesia Reef Restoration Projects Show Greatly Accelerated Growth Rates. In T. J. Goreau & R. K. T. Trench (Eds.), *Innovative Methods of Marine Ecosystem Restoration* (pp. 47–58). Boca Raton, FL: CRC Press.

Jones, I. L. (2001). Auks. In C. Elphick, J. B. Dunning, Jr., & D. A. Sibley (Eds.), *The Sibley Guide to Bird Life and Behavior* (pp. 309–318). New York, NY: Alfred A. Knopf, Inc.

Jorgensen, R., N. O. Handegard, H. Gjosæter, and A. Slotte. (2004). Possible vessel avoidance behaviour of capelin in a feeding area and on a spawning ground. *Fisheries Research 69* (2): 251–261. DOI:10.1016/j.fishres.2004.04.012

Juhasz, A. L. and R. Naidu. (2007). Explosives: Fate, dynamics, and ecological impact in terrestrial and marine environments. *Reviews of Environmental Contamination and Toxicology 191* 163–215.

Kain, E., J. Lavers, C. Berg, A. Raine, and A. Bond. (2016). Plastic ingestion by Newell's (*Puffinus newelli*) and wedge-tailed shearwaters (*Ardenna pacifica*) in Hawaii. *Environmental Science and Pollution Research* 1–9. DOI:10.1007/s11356-016-7613-1

Kaiser, M. J., J. S. Collie, S. J. Hall, S. Jennings, and I. R. Poiner. (2002). Modification of marine habitats by trawling activities: Prognosis and solutions. *Fish and Fisheries 3* (2): 114–136. DOI:10.1046/j.1467-2979.2002.00079

Kajiura, S. M. and K. N. Holland. (2002). Electroreception in juvenile scalloped hammerhead and sandbar sharks. *The Journal of Experimental Biology 205* 3609–3621.

Kalmijn, A. J. (2000). Detection and processing of electromagnetic and near-field acoustic signals in elasmobranch fishes. *Philosophical Transactions of the Royal Society of London Series B-Biological Sciences 355* (1401): 1135–1141. DOI:10.1098/rstb.2000.0654

Kaplan, M. B. and T. A. Mooney. (2016). Coral reef soundscapes may not be detectable far from the reef. *Scientific Reports 6* (31862): 1–10. DOI:10.1038/srep31862

Kaposi, K. L., B. Mos, B. P. Kelaher, and S. A. Dworjanyn. (2014). Ingestion of microplastic has limited impact on a marine larva. *Environmental Science & Technology 48* (3): 1638–1645. DOI:10.1021/es404295e

Kastelein, R. A., M. Janssen, W. C. Verboom, and D. de Haan. (2005). Receiving beam patterns in the horizontal plane of a harbor porpoise (*Phocoena phocoena*). *The Journal of the Acoustical Society of America 118* (2): 1172–1179. DOI:10.1121/1.1945565

Kastelein, R. A., S. van der Heul, W. C. Verboom, N. Jennings, J. van der Veen, and D. de Haan. (2008). Startle response of captive North Sea fish species to underwater tones between 0.1 and 64 kHz. *Marine Environmental Research 65* (5): 369–377. DOI:10.1016/j.marenvres.2008.01.001

Keevin, T. M. and G. L. Hempen. (1997). *The Environmental Effects of Underwater Explosions with Methods to Mitigate Impacts*. St. Louis, MO: U.S. Army Corps of Engineers.

Keller, A. A., E. L. Fruh, M. M. Johnson, V. Simon, and C. McGourty. (2010). Distribution and abundance of anthropogenic marine debris along the shelf and slope of the U.S. West Coast. *Marine Pollution Bulletin 60* (5): 692–700. DOI:10.1016/j.marpolbul.2009.12.006

Kelley, C., G. Carton, M. Tomlinson, and A. Gleason. (2016). Analysis of towed camera images to determine the effects of disposed mustard-filled bombs on the deep water benthic community off south Oahu. *Deep Sea Research Part II: Topical Studies in Oceanography 128* 34–42. DOI:10.1016/j.dsr2.2015.01.016

Ketten, D. R. (1998). *Marine Mammal Auditory Systems: A Summary of Audiometric and Anatomical Data and Its Implications for Underwater Acoustic Impacts*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Kight, C. R., S. S. Saha, and J. P. Swaddle. (2012). Anthropogenic noise is associated with reductions in the productivity of breeding Eastern Bluebirds (*Sialia sialis*). *Ecological Applications 22* (7): 1989–1996.

Kimber, J. A., D. W. Sims, P. H. Bellamy, and A. B. Gill. (2014). Elasmobranch cognitive ability: using electroreceptive foraging behaviour to demonstrate learning, habituation and memory in a benthic shark. *Animal Cognition 17* (1): 55–65. DOI:10.1007/s10071-013-0637-8

Kirschvink, J. L. (1990). Geomagnetic sensitivity in cetaceans: An update with live stranding records in the United States. In J. A. Thomas & R. A. Kastelein (Eds.), *Sensory Abilities of Cetaceans: Laboratory and Field Evidence* (pp. 639–649). New York, NY: Plenium Press.

Kirschvink, J. L., A. E. Dizon, and J. A. Westphal. (1986). Evidence from strandings for geomagnetic sensitivity in cetaceans. *The Journal of Experimental Biology 120* 1–24.

Kirstein, I. V., S. Kirmizi, A. Wichels, A. Garin-Fernandez, R. Erler, M. Loder, and G. Gerdts. (2016). Dangerous hitchhikers? Evidence for potentially pathogenic *Vibrio spp*. on microplastic particles. *Marine Environmental Research 120* 1–8. DOI:10.1016/j.marenvres.2016.07.004

Klinowska, M. (1985). Cetacean live stranding sites relative to geomagnetic topography. *Aquatic Mammals 1985* (1): 27–32.

Knight, R. L. and S. A. Temple. (1986). Why does intensity of avian nest defense increase during the nesting cycle? *The Auk 103* (2): 318–327.

Knowlton, A. R. and M. W. Brown. (2007). Running the gauntlet: Right whales and vessel strikes. In S. D. Kraus & R. M. Rolland (Eds.), *The Urban Whale: North Atlantic Right Whales at the Crossroads* (pp. 409–435). Cambridge, MA: Harvard University Press.

Knowlton, A. R., J. Robbins, S. Landry, H. A. McKenna, S. D. Kraus, and T. B. Werner. (2016). Effects of fishing rope strength on the severity of large whale entanglements. *Conservation Biology 30* (2): 318–328. DOI:10.1111/cobi.12590

Koehl, M. A. R., J. R. Koseff, J. P. Crimaldi, M. G. McCay, T. Cooper, M. B. Wiley, and P. A. Moore. (2001). Lobster sniffing: Antennule design and hydrodynamic filtering of information in an odor plume. *Science 294* 1948–1951.

Koide, S., J. A. K. Silva, V. Dupra, and M. Edwards. (2016). Bioaccumulation of chemical warfare agents, energetic materials, and metals in deep-sea shrimp from discarded military munitions sites off Pearl Harbor. *Deep Sea Research Part II: Topical Studies in Oceanography 128* 53–62. DOI:10.1016/j.dsr2.2015.03.002

Koubrak, O., D. L. VanderZwaag, and B. Worm. (2021). Saving the North Atlantic right whale in a changing ocean: Gauging scientific and law and policy responses. *Ocean & Coastal Management 200* 105109. DOI:https://doi.org/10.1016/j.ocecoaman.2020.105109

Kovacs, K. M., A. Aguilar, D. Aurioles, V. Burkanov, C. Campagna, N. Gales, T. Gelatt, S. D. Goldsworthy, S. J. Goodman, G. J. G. Hofmeyr, T. Harkonen, L. Lowry, C. Lydersen, J. Schipper, T. Sipila, C. Southwell, S. Stuart, D. Thompson, and F. Trillmich. (2012). Global threats to pinnipeds. *Marine Mammal Science 28* (2): 414–436.

Kowarski, K. A., B. J. Gaudet, A. J. Cole, E. E. Maxner, S. P. Turner, S. B. Martin, H. D. Johnson, and J. E. Moloney. (2020). Near real-time marine mammal monitoring from gliders: Practical challenges, system development, and management implications. *The Journal of the Acoustical Society of America 148* (3): 1215–1230. DOI:10.1121/10.0001811

Kozuck, A. (2003). *Implications of historical changes in fixed fishing gear for large whale entanglements in the northwest Atlantic.* (Master's thesis). Duke University, Chapel Hill, NC. Retrieved from dukespace.lib.duke.edu.

Kremers, D., A. Celerier, B. Schaal, S. Campagna, M. Trabalon, M. Boye, M. Hausberger, and A. Lemasson. (2016a). Sensory Perception in Cetaceans: Part I—Current Knowledge about Dolphin Senses as a Representative Species. *Frontiers in Ecology and Evolution 4* (49): 1–17. DOI:10.3389/fevo.2016.00049

Kremers, D., A. Celerier, B. Schaal, S. Campagna, M. Trabalon, M. Boye, M. Hausberger, and A. Lemasson. (2016b). Sensory Perception in Cetaceans: Part II—Promising Experimental Approaches to Study Chemoreception in Dolphins. *Frontiers in Ecology and Evolution 4* (50): 1–9. DOI:10.3389/fevo.2016.00050

Kremers, D., J. Lopez Marulanda, M. Hausberger, and A. Lemasson. (2014). Behavioural evidence of magnetoreception in dolphins: Detection of experimental magnetic fields. *Die Naturwissenschaften 101* (11): 907–911. DOI:10.1007/s00114-014-1231-x

Kuhn, S., E. L. B. Rebolledo, and J. A. Van Franeker. (2015). Deleterious Effects of Litter on Marine Life. In M. Bergmann, L. Gutow, & M. Klages (Eds.), *Marine Anthropogenic Litter* (pp. 75–116). Den Burg, The Netherlands: IMARES Wageningen UR.

Kühn, S. and J. A. Van Franeker. (2020). Quantitative overview of marine debris ingested by marine megafauna. *Marine Pollution Bulletin 151* 110858.

Kujawa, S. G. and M. C. Liberman. (2009). Adding insult to injury: Cochlear nerve degeneration after "temporary" noise-induced hearing loss. *The Journal of Neuroscience 29* (45): 14077–14085. DOI:10.1523/JNEUROSCI.2845-09.2009

Kunc, H. P., G. N. Lyons, J. D. Sigwart, K. E. McLaughlin, and J. D. R. Houghton. (2014). Anthropogenic noise affects behavior across sensory modalities. *The American Naturalist 184* (4): 1–9.

Kunc, H. P., K. E. McLaughlin, and R. Schmidt. (2016). Aquatic noise pollution: Implications for individuals, populations, and ecosystems. *Proceedings of the Royal Society of London B 283* (20160839): 1–8. DOI:http://dx.doi.org/10.1098/rspb.2016.0839

Kuzhetsov, B. (1999). Vegetative responses of dolphin to changes in permanent magentic field. *Biofizika 44* (3): 496–502.

Lacroix, D. L., R. B. Lanctot, J. A. Reed, and T. L. McDonald. (2003). Effect of underwater seismic surveys on molting male long-tailed ducks in the Beaufort Sea, Alaska. *Canadian Journal of Zoology 81* 1862–1875. DOI:10.1139/Z09-185

Lagardère, J. P. (1982). Effects of noise on growth and reproduction of *Crangon crangon* in rearing tanks. *Marine Biology 71* 177–185.

Laggner, D. (2009). *Blue whale (Baleanoptera musculus) ship strike threat assessment in the Santa Barbara Channel, California.* (Unpublished master's thesis). The Evergreen State College, Olympia, WA. Retrieved from http://archives.evergreen.edu.

Laist, D. W. (1987). Overview of the biological effects of lost and discarded plastic debris in the marine environment. *Marine Pollution Bulletin 18* (6B): 319–326.

Laist, D. W. (1997). Impacts of marine debris: Entanglement of marine life in marine debris including a comprehensive list of species with entanglement and ingestion records. In J. M. Coe & D. B. Rogers (Eds.), *Marine Debris: Sources, Impacts, and Solutions* (pp. 99–140). New York, NY: Springer-Verlag.

Laist, D. W., A. R. Knowlton, J. G. Mead, A. S. Collet, and M. Podesta. (2001). Collisions between ships and whales. *Marine Mammal Science 17* (1): 35–75.

Lambertsen, R. H., K. J. Rasmussen, W. C. Lancaster, and R. J. Hintz. (2005). Functional morphology of the mouth of the bowhead whale and its implications for conservation. *Journal of Mammalogy 86* (2): 342–352.

Lammers, M. O., A. A. Pack, and L. Davis. (2003). *Historical evidence of whale/vessel collisions in Hawaiian waters (1975–Present)*. Honolulu, HI: National Oceanic and Atmospheric Administration Ocean Science Institute.

Larkin, R. P., L. L. Pater, and D. J. Tazlk. (1996). *Effects of Military Noise on Wildlife: A Literature Review. USACERL Technical Report 96/21*. U.S. Army Construction Engineering Research Laboratories, Champaign, IL.

Lavers, J., A. Bond, and I. Hutton. (2014). Plastic ingestion by Flesh-footed Shearwaters (*Puffinus carneipes*): Implications for fledgling body condition and the accumulation of plastic-derived chemicals. *Environmental Pollution 187* 124–129.

Leite, L., D. Campbell, L. Versiani, J. Anchieta, C. C. Nunes, and T. Thiele. (2016). First report of a dead giant squid (*Architeuthis dux*) from an operating seismic vessel. *Marine Biodiversity Records 9* (26): 1–3. DOI:10.1186/s41200-016-0028-3

Levinton, J. S. (2013). Plankton *Marine Biology: Function, Biodiversity, Ecology* (4th ed., pp. 141–162). New York, NY: Oxford University Press.

Li, J., J. Ren, J. Zhang, and S. Liu. (2008). The distribution of dissolved aluminum in the Yellow and East China Seas. *Journal of Ocean University of China 7* (1): 48–54. DOI:10.1007/s11802-008-0048-7

Lillis, A., D. Bohnenstiehl, J. W. Peters, and D. Eggleston. (2016). Variation in habitat soundscape characteristics influences settlement of a reef-building coral. *PeerJ 4:e2557*. DOI:10.7717/peerj.2557

Lin, H. W., A. C. Furman, S. G. Kujawa, and M. C. Liberman. (2011). Primary neural degeneration in the guinea pig cochlea after reversible noise-induced threshold shift. *Journal of the Association for Research in Otolaryngology 12* (5): 605–616. DOI:10.1007/s10162-011-0277-0

Lin, J. (2002). *Alca torda: Animal diversity web*. Retrieved May 9, from http://animaldiversity.ummz.umich.edu/accounts/Alca_torda/.

Lincoln, F. C., S. R. Perterson, and J. L. Zimmerman. (1998). *Migration of Birds* (Migration of Birds Circular 16). Manhattan, KS: U.S. Fish and Wildlife Service.

Lindholm, J., M. Gleason, D. Kline, L. Clary, S. Rienecke, and M. Bell. (2013). *Central Coast Trawl Impact and Recovery Study: 2009–2012 Final Report*. Sacramento, CA: California Ocean Protection Council.

Lohmann, K. J. and C. M. F. Lohmann. (2006). Sea turtles, lobsters, and oceanic magnetic maps. *Marine and Freshwater Behaviour and Physiology 39* (1): 49–64. DOI:10.1080/10236240600563230

Lohmann, K. J., N. D. Pentcheff, G. A. Nevitt, G. D. Stetten, R. K. Zimmer-Faust, H. E. Jarrard, and L. C. Boles. (1995). Magnetic orientation of spiny lobsters in the ocean: Experiments with undersea coil systems. *The Journal of Experimental Biology 198* (10): 2041–2048.

Lotufo, G. R., W. Blackburn, S. J. Marlborough, and J. W. Fleeger. (2010). Toxicity and bioaccumulation of TNT in marine fish in sediment exposures. *Ecotoxicology and Environmental Safety 73* (7): 1720–1727. DOI:10.1016/j.ecoenv.2010.02.009

Lotufo, G. R., G. Rosen, W. Wild, and G. Carton. (2013). *Summary Review of the Aquatic Toxicology of Munitions Constituents*. Washington, DC: U.S. Army Corps of Engineers.

Love, M. S., M. M. Nishimoto, S. Clark, and A. S. Bull. (2016). *Renewable Energy in situ Power Cable Observation*. Camarillo, CA: U.S. Department of the Interior, Bureau of Ocean Energy Management, Pacific OCS Region.

Love, M. S. and A. York. (2005). A comparison of the fish assemblages associated with an oil/gas pipeline and adjacent seafloor in the Santa Barbara Channel, Southern California Bight. *Bulletin of Marine Science 77* (1): 101–117.

Lundquist, C. J., S. F. Thrush, G. Coco, and J. E. Hewitt. (2010). Interactions between disturbance and dispersal reduce persistence thresholds in a benthic community. *Marine Ecology Progress Series 413* 217–228. DOI:10.3354/meps08578

Lusseau, D., D. E. Bain, R. Williams, and J. C. Smith. (2009). Vessel traffic disrupts the foraging behavior of southern resident killer whales, *Orcinus orca*. *Endangered Species Research 6* 211–221. DOI:10.3354/esr00154

MacDonald, J. and C. Mendez. (2005). *Unexploded Ordnance Cleanup Costs: Implications of Alternative Protocols* (MG-244-RC). Santa Monica, CA, Arlington, VA, and Pittsburgh, PA: RAND Corporation.

Macfadyen, G., T. Huntington, and R. Cappell. (2009). *Abandoned, Lost or Otherwise Discarded Fishing Gear*. Rome, Italy: United Nations Environment Programme, Food and Agriculture Organization of the United Nations.

Mach, K. J., B. B. Hale, M. W. Denny, and D. V. Nelson. (2007). Death by small forces: A fracture and fatigue analysis of wave-swept macroalgae. *The Journal of Experimental Biology 210* (13): 2231–2243. DOI:10.1242/jeb.001578

MacLeod, C. D., M. B. Santos, and G. J. Pierce. (2003). Review of data on diets of beaked whales: Evidence of niche separation and geographic segregation. *Journal of the Marine Biological Association of the United Kingdom 83* 651–665.

Macreadie, P. I., A. M. Fowler, and D. J. Booth. (2011). Rigs-to-reefs: Will the deep sea benefit from artificial habitat? *Frontiers in Ecology and the Environment 9* (8): 455–461. DOI:10.1890/100112

Magalhães, S., R. Prieto, M. A. Silva, J. Gonçalves, M. Afonso-Dias, and R. S. Santos. (2002). Short-term reactions of sperm whales (*Physeter macrocephalus*) to whale-watching vessels in the Azores. *Aquatic Mammals 28* (3): 267–274.

Manci, K. M., D. N. Gladwin, R. Villella, and M. G. Cavendish. (1988). *Effects of Aircraft Noise and Sonic Booms on Domestic Animals and Wildlife: A Literature Synthesis* (NERC-88/29). Fort Collins, CO: U.S. Fish and Wildlife Service, National Ecology Research Center.

Manville, A. (2016). *A Briefing Memorandum: What We Know, Can Infer, and Don't Yet Know about Impacts from Thermal and Non-thermal Non-ionizing Radiation to Birds and Other Wildlife—for Public Release*. Washington, DC: U.S. Fish and Wildlife Service.

Marcotte, M. M. and C. G. Lowe. (2008). Behavioral responses of two species of sharks to pulsed, direct current electrical fields: Testing a potential shark deterrent. *Marine Technology Society Journal 42* (2): 53–61.

Martin, J., Q. Sabatier, T. A. Gowan, C. Giraud, E. Gurarie, C. S. Calleson, J. G. Ortega-Ortiz, C. J. Deutsch, A. Rycyk, and S. M. Koslovsky. (2015). A quantitative framework for investigating risk of deadly collisions between marine wildlife and boats. *Methods in Ecology and Evolution 7* (1): 42–50. DOI:10.1111/2041-210X.12447

Mather, J. (2004). Cephalopod Skin Displays: From Concealment to Communication. In D. Kimbrough Oller and Ulrike Griebel (Ed.), *The Evolution of Communication Systems: A Comparative Approach*. Cambridge, MA: The Vienna Series in Theoretical Biology and the Massachusetts Institute of Technology.

McCauley, R. D., R. D. Day, K. M. Swadling, Q. P. Fitzgibbon, R. A. Watson, and J. M. Semmens. (2017). Widely used marine seismic survey air gun operations negatively impact zooplankton. *Nature 1* (0195). DOI:10.1038/s41559-017-0195

McCauley, R. D., J. Fewtrell, A. J. Duncan, C. Jenner, M.-N. Jenner, J. D. Penrose, R. I. T. Prince, A. Adhitya, J. Murdoch, and K. A. McCabe. (2000). *Marine Seismic Surveys: Analysis and Propagation of Air-gun Signals; and Effects of Air-gun Exposure on Humpback Whales, Sea Turtles, Fishes and Squid*. Bentley, Australia: Centre for Marine Science and Technology.

McCauley, S. and K. Bjorndal. (1999). Conservation implications of dietary dilution from debris ingestion: Sublethal effects in post-hatchling loggerhead sea turtles. *Conservation Biology 13* (4): 925–929.

Mead, K. S. and M. W. Denny. (1995). The effects of hydrodynamic shear stress on fertilization and early development of the purple sea urchin *Strongylocentrotus purpuratus*. *The Biological Bulletin 188* 46–56.

Melvin, E. and J. Parrish. (2001). *Seabird Bycatch: Trends, Roadblocks, and Solutions, February 26-27, 1999* [Type]. Presented at the Annual Meeting of the Pacific Seabird Group. Blaine, WA.

Melvin, E. F., K. S. Dietrich, S. Fitzgerald, and T. Cardoso. (2011). Reducing seabird strikes with trawl cables in the pollock catcher-processor fleet in the eastern Bering Sea. *Polar Biology 34* (2): 215–226.

Melvin, E. F., J. K. Parrish, and L. L. Conquest. (1999). Novel tools to reduce seabird bycatch in coastal gillnet fisheries; Nuevas herramientas para reducir la captura accidental de aves marinas con redes agalleras de pesquerías costeras. *Conservation Biology 13* (6): 1386–1397. DOI:10.1046/j.1523-1739.1999.98426

Merkel, F. R. and K. L. Johansen. (2011). Light-induced bird strikes on vessels in Southwest Greenland. *Marine Pollution Bulletin 62* (11): 2330–2336.

Miksis-Olds, J. L., P. L. Donaghay, J. H. Miller, P. L. Tyack, and J. E. Reynolds, III. (2007). Simulated vessel approaches elicit differential responses from manatees. *Marine Mammal Science 23* (3): 629–649. DOI:10.1111/j.1748-7692.2007.00133

Miller, M. H. and C. Klimovich. (2016). *Endangered Species Act Status Review Report: Giant Manta Ray (Manta birostris) and Reef Manta Ray (Manta alfredi).* Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Mintz, J. D. (2012). *Vessel Traffic in the Hawaii-Southern California and Atlantic Fleet Testing and Training Study Areas*. Alexandria, VA: Center for Naval Analyses.

Mintz, J. D. and C. L. Parker. (2006). *Vessel Traffic and Speed Around the U.S. Coasts and Around Hawaii*. Alexandria, VA: Center for Naval Analyses.

Missile Technology Control Regime. (1996). *Missile Technology Control Regime*. Retrieved from http://mtcr.info.

Misund, O. A. (1997). Underwater acoustics in marine fisheries and fisheries research. *Reviews in Fish Biology and Fisheries 7* 1–34.

Mitchelmore, C. L., C. A. Bishop, and T. K. Collier. (2017). Toxicological estimation of mortality of oceanic sea turtles oiled during the Deepwater Horizon oil spill. *Endangered Species Research 33* 39–50.

Monterey Bay Research Institute. (2010). *Periodic Table of Elements in the Ocean*. *The MBARI Chemical Sensor Program.* Retrieved from www.mbari.org/chemsensor/pteo.htm.

Montoto-Martínez, T., J. De la Fuente, R. Puig-Lozano, N. Marques, M. Arbelo, J. J. Hernández-Brito, A. Fernández, and M. D. Gelado-Caballero. (2021). Microplastics, bisphenols, phthalates and pesticides in odontocete species in the Macaronesian Region (Eastern North Atlantic). *Marine Pollution Bulletin 173* 113105. DOI:https://doi.org/10.1016/j.marpolbul.2021.113105

Moore, A. and W. D. Riley. (2009). Magnetic particles associated with the lateral line of the European eel, *Anguilla anguilla*. *Journal of Fish Biology 74* (7): 1629–1634. DOI:10.1111/j.1095-8649.2009.02197

Moore, M. J., T. K. Rowles, D. A. Fauquier, J. D. Baker, I. Biedron, J. W. Durban, P. K. Hamilton, A. G. Henry, A. R. Knowlton, W. A. McLellan, C. A. Miller, R. M. Pace, III, H. M. Pettis, S. Raverty, R. M.

Rolland, R. S. Schick, S. M. Sharp, C. R. Smith, L. Thomas, J. M. van der Hoop, and M. H. Ziccardi. (2021). REVIEW: Assessing North Atlantic right whale health: threats, and development of tools critical for conservation of the species. *Diseases of Aquatic Organisms 143* 205–226.

Moser, M. L. and D. S. Lee. (1992). A fourteen-year survey of plastic ingestion by western north Atlantic seabirds. *Colonial Waterbirds 15* (1): 83–94.

Myrberg, A. A., Jr. (2001). The acoustical biology of elasmobranchs. *Environmental Biology of Fishes 60* 31–45.

National Marine Fisheries Service. (2008a). *Biological Opinion for the 2008 Rim-of-the-Pacific Joint Training Exercises*. Silver Spring, MD: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Endangered Species Division.

National Marine Fisheries Service. (2008b). *Biological Opinion for the 2008 Rim-of-the-Pacific Joint Training Exercises*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Endangered Species Division.

National Marine Fisheries Service. (2013). *Biological Opinion and Conference Opinion on Atlantic Fleet Training and Testing Activities (2013–2018)*. Silver Spring, MD: Office of Protected Resources.

National Marine Fisheries Service. (2015). *Reinitiated Biological Opinion and Conference Report on U.S. Navy Hawaii-Southern California Training and Testing*. Washington, DC: The United States Navy and National Oceanic and Atmospheric Administration's National Marine Fisheries Service, Office of Protected Resources' Permits and Conservation Division.

National Marine Fisheries Service. (2022a). *National Report on Large Whale Entanglements Confirmed in the United States in 2019*. Silver Spring, MD: Office of Protected Resources, Marine Mammal and Sea Turtle Conservation Division, Marine Mammal Health and Stranding Response Program.

National Marine Fisheries Service. (2022b). *National Report on Large Whale Entanglements Confirmed in the United States in 2020*. Silver Spring, MD: Office of Protected Resources, Marine Mammal and Sea Turtle Conservation Division, Marine Mammal Health and Stranding Response Program.

National Marine Fisheries Service. (2023). *2017–2023 North Atlantic Right Whale Unusual Mortality Event*. Retrieved from https://www.fisheries.noaa.gov/national/marine-life-distress/2017-2023-north-atlantic-right-whale-unusual-mortality-event.

National Oceanic and Atmospheric Administration Marine Debris Program. (2014a). *Report on the Entanglement of Marine Species in Marine Debris with an Emphasis on Species in the United States*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Ocean Service.

National Oceanic and Atmospheric Administration Marine Debris Program. (2014b). *Report on the Occurrence of Health Effects of Anthropogenic Debris Ingested by Marine Organisms*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Ocean Service.

National Oceanic and Atmospheric Administration Marine Debris Program. (2014c). *Report on the Occurrence of Health Effects of Anthropogenic Debris Ingested by Marine Organisms*. Silver Spring, MD: National Ocean Service.

National Park Service. (1994). *Report on Effects of Aircraft Overflights on the National Park System* (Report to Congress prepared pursuant to Public Law 100-191, the National Parks Overflights Act of 1987). Washington, DC.

Naughton, M. B., M. D. Romano, and T. S. Zimmerman. (2007). *A Conservation Action Plan for Black-footed Albatross (Phoebastria nigripes) and Laysan Albatross (P. immutabilis) Version 1.0*. Washington, DC: U.S. Fish and Wildlife Service.

Naval Facilities Engineering Systems Command Pacific. (2023). *Literature Review: Impacts of Military Testing and Training on Reef Fish Contaminant Bioaccumulation, Human, and Ecological Impacts in the Mariana Islands*. Joint Base Pearl Harbor-Hickham, HI: Naval Facilities Engineering Systems Command Pacific.

Neckles, H. A., F. T. Short, S. Barker, and B. S. Kopp. (2005). Disturbance of eelgrass, *Zostera marina,* by commercial mussel, *Mytilus edulis,* harvesting in Maine: Dragging impacts and habitat recovery. *Marine Ecology Progress Series 285* 57–73.

Nedelec, S. L., A. N. Radford, S. D. Simpson, B. Nedelec, D. Lecchini, and S. C. Mills. (2014). Anthropogenic noise playback impairs embryonic development and increases mortality in a marine invertebrate. *Scientific Reports 4* (5891): 1–4. DOI:10.1038/srep05891

Negri, A. P., L. D. Smith, N. S. Webster, and A. J. Heyward. (2002). Understanding ship-grounding impacts on a coral reef: Potential effects of anti-foulant paint contamination on coral recruitment. *Marine Pollution Bulletin 44* 111–117. DOI:10.1016/s0025-326x(01)00128-xO'Keefe 198

Neilson, J. L., C. M. Gabriele, A. S. Jensen, K. Jackson, and J. M. Straley. (2012). Summary of Reported Whale-Vessel Collisions in Alaskan Waters. *Journal of Marine Biology 2012* 1–18. DOI:10.1155/2012/106282

Neilson, J. L., J. M. Straley, C. M. Gabriele, and S. Hills. (2009). Non-lethal entanglement of humpback whales (*Megaptera novaeangliae*) in fishing gear in northern Southeast Alaska. *Journal of Biogeography 36* 452–464. DOI:10.1111/j.1365-2699.2007.01820

Nemeth, D. J. and C. H. Hocutt. (1990). Acute effects of electromagnetic pulses (EMP) on fish. *Journal of Applied Ichthyology 6* (1): 59–64.

New, L. F., J. Harwood, L. Thomas, C. Donovan, J. S. Clark, G. Hastie, P. M. Thompson, B. Cheney, L. Scott-Hayward, D. Lusseau, and D. Costa. (2013). Modelling the biological significance of behavioural change in coastal bottlenose dolphins in response to disturbance. *Functional Ecology 27* (2): 314–322. DOI:10.1111/1365-2435.12052

Niemiec, A. J., Y. Raphael, and D. B. Moody. (1994). Return of auditory function following structural regeneration after acoustic trauma: Behavioral measures from quail. *Hearing Research 75* 209–224.

Nipper, M., R. S. Carr, J. M. Biedenbach, R. L. Hooten, and K. Miller. (2002). Toxicological and chemical assessment of ordnance compounds in marine sediments and porewaters. *Marine Pollution Bulletin 44* 789–806.

Noirot, I. C., E. F. Brittan-Powell, and R. J. Dooling. (2011). Masked auditory thresholds in three species of birds, as measured by the auditory brainstem response. *The Journal of the Acoustical Society of America 129* (6): 3445–3448. DOI:10.1121

Normandeau, E., T. Tricas, and A. Gill. (2011). *Effects of EMFs from Undersea Power Cables on Elasmobranchs and Other Marine Species*. Camarillo, CA: U.S. Department of the Interior, Bureau of Ocean Energy Management, Regulation, and Enforcement, Pacific Outer Continental Shelf Region.

Nowacek, D., M. Johnson, and P. Tyack. (2004a). North Atlantic right whales (*Eubalaena glacialis*) ignore ships but respond to alerting stimuli. *Proceedings of the Royal Society of London 271* (B): 227–231. DOI:10.1098/rspb.2003.2570

Nowacek, D., L. H. Thorne, D. Johnston, and P. Tyack. (2007). Responses of cetaceans to anthropogenic noise. *Mammal Review 37* (2): 81–115.

Nowacek, S. M., R. S. Wells, E. C. G. Owen, T. R. Speakman, R. O. Flamm, and D. P. Nowacek. (2004b). Florida manatees, *Trichechus manatus latirostris*, respond to approaching vessels. *Biological Conservation 119* 517–523. DOI:10.1016/j.biocon.2003.11.020

Nozaki, Y. (1997). *Supplementary Materials to "A Fresh Look at Element Distribution in the North Pacific"*. Retrieved February 16, 2011, from http://www.agu.org/pubs/eos-news/supplements/1995-2003/97026e.shtml.

Nybakken, J. W. (1993). *Marine Biology, an Ecological Approach* (3rd ed.). New York, NY: Harper Collins College Publishers.

O'Connell, C. P., D. C. Abel, P. H. Rice, E. M. Stroud, and N. C. Simuro. (2010). Responses of the southern stingray (*Dasyatis americana*) and the nurse shark (*Ginglymostoma cirratum*) to permanent magnets. *Marine and Freshwater Behaviour and Physiology 43* (1): 63–73. DOI:10.1080/10236241003672230

O'Keeffe, D. J. and G. A. Young. (1984). *Handbook on the Environmental Effects of Underwater Explosions*. Silver Spring, MD: U.S. Navy, Naval Surface Weapons Center (Code R14).

O'Shea, T., C. A. Beck, R. K. Bonde, H. I. Kochman, and D. K. Odell. (1985). An analysis of manatee mortality patterns in Florida, 1976–81. *The Journal of Wildlife Management 49* (1): 1–11.

Ocean Conservancy. (2010). *Trash Travels: From Our Hands to the Sea, Around the Globe, and Through Time* (International Coastal Cleanup Report). Washington, DC: Ocean Conservancy.

Ohman, M. C., P. Sigray, and H. Westerberg. (2007). Offshore windmills and the effects of electromagnetic fields on fish. *Ambio 36* (8): 630–633. DOI:10.1579/0044-7447(2007)36[630:OWATEO]2.0.CO;2

Olson, J. K., D. M. Lambourn, J. L. Huggins, S. Raverty, A. A. Scott, and J. K. Gaydos. (2021). Trends in Propeller Strike-Induced Mortality in Harbor Seals (*Phoca vitulina*) of the Salish Sea. *Journal of Wildlife Diseases 57* (3): 689-693. DOI:10.7589/jwd-d-20-00221

Pace, D. S., A. Miragliuolo, and B. Mussi. (2008). Behaviour of a social unit of sperm whales (*Physeter macrocephalus*) entangled in a driftnet off Capo Palinuro (Southern Tyrrhenian Sea, Italy). *Journal of Cetacean Research and Management 10* (2): 131–135.

Pacific Shark Research Center. (2017). *Basking Sharks*. Retrieved May 30, 2017, from https://psrc.mlml.calstate.edu/current-research/citizen-science/basking-shark/about-basking-sharks/.

Page, B., J. McKenzie, R. McIntosh, A. Baylis, A. Morrissey, N. Calvert, T. Haase, M. Berros, D. Dowie, P. D. Shaughnessy, and S. D. Goldsworthy. (2004). Entanglement of Australian sea lions and New Zealand fur seals in lost fishing gear and other marine debris before and after government and industry attempts to reduce the problem. *Marine Pollution Bulletin 49* (1–2): 33–42. DOI:10.1016/j.marpolbul.2004.01.006

Pait, A. S., A. L. Mason, D. R. Whitall, J. D. Christensen, and S. I. Hartwell. (2010). Chapter 5: Assessment of Chemical Contaminants in Sediments and Corals in Vieques. In L. J. Bauer & M. S. Kendall

(Eds.), *An Ecological Characterization of the Marine Resources of Vieques, Puerto Rico* (pp. 101–150). Silver Spring, MD: National Oceanic and Atmospheric Administration.

Palka, D. L. and P. S. Hammond. (2001). Accounting for responsive movement in line transect estimates of abundance. *Canadian Journal of Fisheries and Aquatic Sciences 58* 777–787. DOI:10.1139/cjfas-58-4-777

Parks, S. E. and D. Wiley. (2009). *Fine-scale Focal Dtag Behavioral Study of Diel Trends in Activity Budgets and Sound Production of Endangered Baleen Whales in the Gulf of Maine* (Marine Mammals and Biology Annual Reports: FY09). Arlington, VA: Office of Naval Research.

Parry, G. D. and A. Gason. (2006). The effect of seismic surveys on catch rates of rock lobsters in western Victoria, Australia. *Fisheries Research 79* 272–284. DOI:10.1016/j.fishres.2006.03.023

Partecke, J., I. Schwabl, and E. Gwinner. (2006). Stress and the city: Urbanization and its effects on the stress physiology in European blackbirds. *Ecology 87* (8): 1945–1952.

Patricelli, G. L. and J. L. Blickley. (2006). Avian communication in urban noise: Causes and consequences of vocal adjustment. *The Auk 123* (3): 639–649.

Pavlostathis, S. G. and G. H. Jackson. (2002). Biotransformation of 2, 4, 6-trinitrotoluene in a continuous-flow *Anabaena* sp. system. *Water Research 36* 1699–1706.

Payne, J. F., C. A. Andrews, L. L. Fancey, A. L. Cook, and J. R. Christian. (2007). *Pilot Study on the Effects of Seismic Air Gun Noise on Lobster (Homarus americanus)*. St. John's, Canada: Environmental Studies Research Funds Report.

Pearson, W. H., J. R. Skalski, S. D. Sulkin, and C. I. Malme. (1994). Effects of seismic energy releases on the survival and development of zoeal larvae of dungeness crab (*Cancer magister*). *Marine Environmental Research 38* 93–113.

Peng, C., X. Zhao, S. Liu, W. Shi, Y. Han, C. Guo, J. Jiang, H. Wan, T. Shen, and G. Liu. (2016). Effects of anthropogenic sound on digging behavior, metabolism, $Ca^{2+}/Mg^{2+}$ ATPase activity, and metabolism-related gene expression of the bivalve *Sinonovacula constricta*. *Scientific Reports 6* (24266): 12. DOI:10.1038/srep24266

Pennington, J. C. and J. M. Brannon. (2002). Environmental fate of explosives. *Thermochimica Acta 384* (1–2): 163–172.

Perkol-Finkel, S., N. Shashar, and Y. Benayahu. (2006). Can artificial reefs mimic natural reef communities? The roles of structural features and age. *Marine Environmental Research 61* 121–135. DOI:10.1016/j.marenvres.2005.08.001

Phillips, R. C. and E. G. Meñez. (1988). Seagrasses. *Smithsonian Contributions to the Marine Sciences 34* 104.

Pierce, K., R. Harris, L. Larned, and M. Pokras. (2004). Obstruction and starvation associated with plastic ingestion in a northern gannet *Morus bassanus* and a greater shearwater *Puffinus gravis*. *Marine Ornithology 32* 187–189.

Pine, M. K., A. J. Jeffs, and C. A. Radford. (2016). Effects of Underwater Turbine Noise on Crab Larval Metamorphosis. In A. Popper & A. Hawkins (Eds.), *The Effects of Noise on Aquatic Life II. Advances in Experimental Medicine and Biology* (Vol. 875, pp. 847–852). New York, NY: Springer.

Pinzone, M., E. S. Nordøy, G. Eppe, C. Malherbe, K. Das, and F. Collard. (2021). First record of plastic debris in the stomach of a hooded seal pup from the Greenland Sea. *Marine Pollution Bulletin 167* 112350. DOI:https://doi.org/10.1016/j.marpolbul.2021.112350

Pirotta, E., N. D. Merchant, P. M. Thompson, T. R. Barton, and D. Lusseau. (2015). Quantifying the effect of boat disturbance on bottlenose dolphin foraging activity. *Biological Conservation 181* 82–89. DOI:10.1016/j.biocon.2014.11.003

Plumpton, D. (2006). *Review of Studies Related to Aircraft Noise Disturbance of Waterfowl: A Technical Report in Support of the Supplemental Environmental Impact Statement for Introduction of F/A-18 E/F (Super Hornet) Aircraft to the East Coast of the United States*. Norfolk, VA: U.S. Department of the Navy.

Polese, G., C. Bertapelle, and A. Cosmo. (2015). Role of olfaction in *Octopus vulgaris* reproduction. *General and Comparative Endocrinology 210* 55–62. DOI:10.1016/j.ygcen.2014.10.006

Poot, H., B. J. Ens, H. de Vries, M. A. H. Donners, M. R. Wernand, and J. M. Marquenie. (2008). Green light for nocturnally migrating birds. *Ecology and Society 13* (2): 47. DOI:47

Popper, A. N., A. D. Hawkins, R. R. Fay, D. A. Mann, S. M. Bartol, T. J. Carlson, S. Coombs, W. T. Ellison, R. L. Gentry, M. B. Halvorsen, S. Løkkeborg, P. H. Rogers, B. L. Southall, D. G. Zeddies, and W. N. Tavolga. (2014). *ASA S3/SC1.4 TR-2014 Sound Exposure Guidelines for Fishes and Sea Turtles: A Technical Report prepared by ANSI-Accredited Standards Committee S3/SC1 and registered with ANSI*. Acoustical Society of America Press, New York, NY.

Popper, A. N., M. Salmon, and K. W. Horch. (2001). Acoustic detection and communication by decapod crustaceans. *Journal of Comparative Physiology A 187* 83–89.

Posey, M. and T. Alphin. (2002). Resilience and Stability in an Offshore Benthic Community: Responses to Sediment Borrow Activities and Hurricane Disturbance. *Journal of Coastal Research 18* (4): 685–697.

Possatto, F. E., M. Barletta, M. F. Costa, J. A. I. do Sul, and D. V. Dantas. (2011). Plastic debris ingestion by marine catfish: An unexpected fisheries impact. *Marine Pollution Bulletin 62* (5): 1098–1102.

Powell, S., P. D. Franzmann, R. Cord-Ruwisch, and S. Toze. (1998). Degradation of 2-nitrodiphenylamine, a component of Otto Fuel II, by *Clostridium* spp. *Anaerobe 4* 95–102.

Precht, W. F., R. B. Aronson, and D. W. Swanson. (2001). Improving scientific decision-making in the restoration of ship-grounding sites on coral reefs. *Bulletin of Marine Science 69* (2): 1001–1012.

Price, C. B., J. M. Brannon, and S. L. Yost. (1998). *Transformation of RDX and HMX Under Controlled Eh/pH Conditions*. Washington, DC: U.S. Army Corps of Engineers, Waterways Experiment Station.

Provencher, J., A. Bond, A. Hedd, W. Montevecchi, S. Muzaffar, S. Courchesne, H. Gilchrist, S. Jamieson, F. Merkel, K. Falk, J. Durinck, and M. Mallory. (2014). Prevalence of marine debris in marine birds from the North Atlantic. *Marine Pollution Bulletin 84* 411–417.

Pytte, C. L., K. M. Rusch, and M. S. Ficken. (2003). Regulation of vocal amplitude by the blue-throated hummingbird, *Lampornis clemenciae*. *Animal Behaviour 66* 703–710. DOI:10.1006/anbe.2003.2257

Quattrini, A. M., M. S. Nizinski, J. D. Chaytor, A. W. Demopoulos, E. B. Roark, S. C. France, J. A. Moore, T. Heyl, P. J. Auster, B. Kinlan, C. Ruppel, K. P. Elliott, B. R. Kennedy, E. Lobecker, A. Skarke, and T. M. Shank. (2015). Exploration of the canyon-incised continental margin of the Northeastern

United States reveals dynamic habitats and diverse communities. *PLoS ONE 10* (10): 1–32. DOI:10.1371/journal.pone.0139904

Radford, C. A., A. G. Jeffs, and J. C. Montgomery. (2007). Directional swimming behavior by five species of crab postlarvae in response to reef sound. *Bulletin of Marine Science 2* (80): 369–378.

Ramirez-Macias, D., M. Meekan, R. De La Parra-Venegas, F. Remolina-Suarez, M. Trigo-Mendoza, and R. Vazquez-Juarez. (2012). Patterns in composition, abundance and scarring of whale sharks *Rhincodon typus* near Holbox Island, Mexico. *Journal of Fish Biology 80* 1401–1416.

Ravera, S., C. Falugi, D. Calzia, I. M. Pepe, I. Panfoli, and A. Morelli. (2006). First cell cycles of sea urchin Paracentrotus lividus are dramatically impaired by exposure to extremely low-frequency electromagnetic field. *Biology of Reproduction 75* (6): 948–953. DOI:10.1095/biolreprod.106.051227

Reed, J., J. Sincock, and J. Hailman. (1985). Light Attraction in Endangered Procellariiform Birds: Reduction by Shielding Upward Radiation. *The Auk 102* 377–383.

Reeder, D. M. and K. M. Kramer. (2005). Stress in free-ranging mammals: Integrating physiology, ecology, and natural history. *Journal of Mammalogy 86* (2): 225–235.

Reinhall, P. G. and P. H. Dahl. (2011). Underwater Mach wave radiation from impact pile driving: Theory and observation. *The Journal of the Acoustical Society of America 130* (3): 1209–1216. DOI:10.1121/1.3614540

Renaud, M. L. and J. A. Carpenter. (1994). Movements and submergence patterns of loggerhead turtles (*Caretta caretta*) in the Gulf of Mexico determined through satellite telemetry. *Bulletin of Marine Science 55* (1): 1–15.

Richardson, W. J., C. R. Greene, Jr., C. I. Malme, and D. H. Thomson. (1995). *Marine Mammals and Noise*. San Diego, CA: Academic Press.

Richter, C., S. Dawson, and E. Slooten. (2006). Impacts of commercial whale watching on male sperm whales at Kaikoura, New Zealand. *Marine Mammal Science 22* (1): 46–63. DOI:10.1111/j.1748-7692.2006.00005

Richter, C., S. M. Dawson, and E. Slooten. (2003). Sperm whale watching off Kaikoura, New Zealand: Effects of current activities on surfacing and vocalisation patterns. *Science for Conservation 219* 78.

Rieth, T. M., T. L. Hunt, C. Lipo, and J. M. Wilmshurst. (2011). The 13th century Polynesian colonization of Hawaii Island. *Journal of Archaeological Science 28* 2740–2749. DOI:10.1016/j.jas.2011.06.017

Rigg, D. P., S. C. Peverell, M. Hearndon, and J. E. Seymour. (2009). Do elasmobranch reactions to magnetic fields in water show promise for bycatch mitigation? *Marine and Freshwater Research 60* (9): 942–948. DOI:10.1071/mf08180

Rios, L. M., C. Moore, and P. R. Jones. (2007). Persistent organic pollutants carried by synthetic polymers in the ocean environment. *Marine Pollution Bulletin 54* 1230–1237. DOI:10.1016/j.marpolbul.2007.03.022

Ritter, F. (2012). *Collisions of Sailing Vessels with Cetaceans Worldwide: First Insights into a Seemingly Growing Problem* (SC/61/BC 1). Berlin, Germany: Mammals Encounters Education Research e.V.

Robbins, J. (2009). *Scar-Based Inference into Gulf of Maine Humpback Whale Entanglement: 2003–2006* (Northeast Fisheries Science Center, National Marine Fisheries Service). Provincetown, MA: Center for Coastal Studies.

Robbins, J. (2010). *A Review of the Frequency and Impact of Entanglement on Gulf of Maine Humpback Whales* (IWC/A10/E3). Washington, DC: International Whaling Commission.

Roberts, L., S. Cheesman, T. Breithaupt, and M. Elliott. (2015). Sensitivity of the mussel *Mytilus edulis* to substrate-borne vibration in relation to anthropogenically generated noise. *Marine Ecology Progress Series 538* 185–195. DOI:10.3354/meps11468

Roberts, L., S. Cheesman, M. Elliott, and T. Breithaupt. (2016a). Sensitivity of *Pagurus bernhardus* (L.) to substrate-borne vibration and anthropogenic noise. *Journal of Experimental Marine Biology and Ecology 474* 185–194.

Roberts, L., H. R. Harding, I. Voellmy, R. Bruintjes, S. D. Simpson, A. N. Radford, T. Breithaupt, and M. Elliott. (2016b). *Exposure of benthic invertebrates to sediment vibration: From laboratory experiments to outdoor simulated pile-driving* [Type]. Presented at the Proceedings of Meetings on Acoustics. Dublin, Ireland.

Rochman, C. M., E. Hoh, T. Kurobe, and S. J. Teh. (2013). Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress. *Scientific Reports 3* 3263. DOI:10.1038/srep03263

Rochman, C. M., A. Tahir, S. L. Williams, D. V. Baxa, R. Lam, J. T. Miller, F. Teh, S. Werorilangi, and S. J. Teh. (2015). Anthropogenic debris in seafood: Plastic debris and fibers from textiles in fish and bivalves sold for human consumption. *Nature 5* 14340–14350.

Rommel, S., A. M. Costidis, T. D. Pitchford, J. D. Lightsey, R. H. Snyder, and E. M. Haubold. (2007). Forensic methods for characterizing watercraft from watercraft-induced wounds on the Florida manatee (*Trichechus manatus latirostris*). *Marine Mammal Science 23* (1): 110–132. DOI:10.1111/j.1748-7692.2006.00095

Ronconi, R. (2001). *Cepphus grylle, black guillemot*. *Animal Diversity Web.* Retrieved from http://animaldiversity.ummz.umich.edu/site/accounts/information/Cepphus_grylle.html.

Rooper, C. N., M. E. Wilkins, C. S. Rose, and C. Coon. (2011). Modeling the impacts of bottom trawling and the subsequent recovery rates of sponges and corals in the Aleutian Islands, Alaska. *Continental Shelf Research 31* 1827–1834. DOI:10.1016/j.csr.2011.08.003

Rosen, G. and G. R. Lotufo. (2005). Toxicity and fate of two munitions constituents in spiked sediment exposures with the marine amphipod *Eohaustorius estuarius*. *Environmental Toxicology and Chemistry 24* (11): 2887–2897.

Rosen, G. and G. R. Lotufo. (2010). Fate and effects of composition B in multispecies marine exposures. *Environmental Toxicology and Chemistry 9999* (12): 1–8. DOI:10.1002/etc.153

Ross, S. W., M. Rhode, S. T. Viada, and R. Mather. (2016). Fish species associated with shipwreck and natural hard-bottom habitats from the middle to outer continental shelf of the Middle Atlantic Bight near Norfolk Canyon. *Fishery Bulletin 114* (1): 45–57. DOI:10.7755/FB.114.1.4

Rostad, A., S. Kaartvedt, T. A. Klevjer, and W. Melle. (2006). Fish are attracted to vessels. *ICES Journal of Marine Science 63* (8): 1431–1437. DOI:10.1016/j.icejms.2006.03.026

Rowat, D., M. G. Meekan, U. Engelhardt, B. Pardigon, and M. Vely. (2007). Aggregations of juvenile whale sharks (*Rhincodon typus*) in the Gulf of Tadjoura, Djibouti. *Environmental Biology of Fishes 80* (4): 465–472. DOI:10.1007/s10641-006-9148-7

Rubel, E. W., S. A. Furrer, and J. S. Stone. (2013). A brief history of hair cell regeneration research and speculations on the future. *Hearing Research 297* 42–51. DOI:10.1016/j.heares.2012.12.014

Runge, M. C., C. A. Sanders-Reed, C. A. Langtimm, and C. J. Fonnesbeck. (2007). *A Quantitative Threats Analysis for the Florida Manatee (Trichechus manatus latirostris)*. Laurel, MD: U.S. Geological Survey, Patuxent Wildlife Research Center.

Russel, W. A., Jr., N. D. Lewis, and B. T. Brown. (1996). The impact of impulsive noise on bald eagles at Aberdeen Proving Ground, Maryland. *The Journal of the Acoustical Society of America 99* (4): 2576–2603.

Ryals, B. M., R. J. Dooling, E. Westbrook, M. L. Dent, A. MacKenzie, and O. N. Larsen. (1999). Avian species differences in susceptibility to noise exposure. *Hearing Research 131* 71–88.

Ryan, P. (1987). The Effects of Ingested Plastic on Seabirds: Correlations between Plastic Load and Body Condition. *Environmental Pollution 46* 119–125.

Rycyk, A. M., C. J. Deutsch, M. E. Barlas, S. K. Hardy, K. Frisch, E. H. Leone, and D. P. Nowacek. (2018). Manatee behavioral response to boats. *Marine Mammal Science 34* (4): 924–962. DOI:10.1111/mms.12491

Sabates, A., M. P. Olivar, J. Salat, I. Palomera, and F. Alemany. (2007). Physical and biological processes controlling the distribution of fish larvae in the NW Mediterranean. *Progress in Oceanography 74* (2–3): 355–376. DOI:10.1016/j.pocean.2007.04.017

Sargent, F. J., T. J. Leary, D. W. Crewz, and C. R. Kruer. (1995). *Scarring of Florida's Seagrasses: Assessment and Management Options*. St. Petersburg, FL: Florida Department of Environmental Protection.

Sasso, C. R. and W. N. Witzell. (2006). Diving behaviour of an immature Kemp's ridley turtle (*Lepidochelys kempii*) from Gullivan Bay, Ten Thousand Islands, southwest Florida. *Journal of the Marine Biological Association of the United Kingdom 86* 919–925.

Saunders, J. C. and R. Dooling. (1974). Noise-induced threshold shift in the parakeet (*Melopsittacus undulatus*). *Proceedings of the National Academy of Sciences 71* (5): 1962–1965.

Savoca, M. (2016, November 10). *Plastic Garbage Chemical Attracts Hungry Seabirds*. Retrieved April 12, 2017, from https://www.scientificamerican.com/article/plastic-garbage-chemical-attracts-hungry-seabirds/.

Savoca, M., M. Wohlfeil, S. Ebeler, and G. Nevitt. (2016). Marine plastic debris emits a keystone infochemical for olfactory foraging seabirds. *Science Advances 2* (e1600395): 1–9.

Scheuhammer, A. (1987). The chronic toxicity of aluminium, cadmium, mercury, and lead in birds: A review. *Environmental Review 46* 263–295.

Schiff, S. C. (1977). *Effects of Aluminized Fiberglass on Representative Chesapeake Bay Marine Organisms*. Washington, DC: U.S. Department of the Navy, Naval Research Laboratory.

Schoeman, D. S., A. McLachlan, and J. E. Dugan. (2000). Lessons from a disturbance experiment in the intertidal zone of an exposed sandy beach. *Estuarine, Coastal and Shelf Science 50* (6): 869–884. DOI:10.1006/ecss.2000.0612

Schoeman, R. P., C. Patterson-Abrolat, and S. Plön. (2020). A global review of vessel collisions with marine animals. *Frontiers in Marine Science 7*. DOI:10.3389/fmars.2020.00292

Schuyler, Q., B. D. Hardesty, C. Wilcox, and K. Townsend. (2014). Global analysis of anthropogenic debris ingestion by sea turtles. *Conservation Biology 28* (1): 129–139.

Schuyler, Q. A., C. Wilcox, K. A. Townsend, K. R. Wedemeyer-Strombel, G. Balazs, E. Sebille, and B. D. Hardesty. (2016). Risk analysis reveals global hotspots for marine debris ingestion by sea turtles. *Global Change Biology 22* (2): 567–576.

Schwartz, A. L. (1985). The behavior of fishes in their acoustic environment. *Environmental Biology of Fishes 13* (1): 3–15.

Setala, O., V. Fleming-Lehtinen, and M. Lehtiniemi. (2014). Ingestion and transfer of microplastics in the planktonic food web. *Environmental Pollution 185* 77–83.

Setala, O., J. Norkko, and M. Lehtiniemi. (2016). Feeding type affects microplastic ingestion in a coastal invertebrate community. *Marine Pollution Bulletin 102* 95–101.

Shckorbatov, Y., I. Rudneva, V. Pasiuga, V. Grabina, N. Kolchigin, D. Ivanchenko, O. Kazanskiy, V. Shayda, and O. Dumin. (2010). Electromagnetic field effects on *Artemia* hatching and chromatin state. *Central European Journal of Biology 5* (6): 785–790. DOI:10.2478/s11535-010-0063-8

Shorr, J., J. Cervino, C. Lin, R. Weeks, and T. J. Goreau. (2012). Electrical Stimulation Increases Oyster Growth and Survival in Restoration Projects. In T. J. Goreau & R. K. Trench (Eds.), *Innovative Methods of Marine Ecosystem Restoration* (pp. 151–160). Boca Raton, FL: CRC Press.

Sies, H. (1997). Physiological Society Symposium: Impaired endothelial and smooth muscle cell function in oxidative stress-oxidative stress: Oxidants and antioxidants. *Experimental Physiology 82* 291–295.

Sievert, P. R. and L. Sileo. (1993). The effects of ingested plastic on growth and survival of albatross chicks. In K. Vermeer, K. T. Briggs, K. H. Morgan, & D. Siegel-Causey (Eds.), *The Status, Ecology, and Conservation of Marine Birds of the North Pacific* (pp. 212–217). Ottawa, Canada: Canadian Wildlife Service Special Publication.

Sijimol, M. R. and M. Mohan. (2014). Environmental impacts of perchlorate with special reference to fireworks - A review. *Environmental Monitoring and Assessment 186* 7203–7210. DOI:10.1007/s10661-014-3921-4

Silber, G., J. Slutsky, and S. Bettridge. (2010). Hydrodynamics of a ship/whale collision. *Journal of Experimental Marine Biology and Ecology 391* 10–19. DOI:10.1016/j.jembe.2010.05.013

Simpson, S. D., A. N. Radford, E. J. Tickle, M. G. Meekan, and A. G. Jeffs. (2011). Adaptive avoidance of reef noise. *PLoS ONE 6* (2): 1–5. DOI:10.1371/journal.pone.0016625

Singh, R., P. Soni, P. Kumar, S. Purohit, and A. Singh. (2009). Biodegradation of high explosive production effluent containing RDX and HMX by denitrifying bacteria. *World Journal of Microbiology and Biotechnology 25* 269–275.

Slabbekoorn, H. and A. den Boer-Visser. (2006). Cities change the songs of birds. *Current Biology 16* (23): 2326–2331.

Slabbekoorn, H. and E. A. Ripmeester. (2007). Birdsong and anthropogenic noise: Implications and applications for conservation. *Molecular Ecology 17* (1): 72–83.

Smith, P. N. (2002). Perchlorate in the Environment: Ecological Considerations. Texas Tech University, Lubbock, TX: U.S. Environmental Protection Agency, National Center for Environmental Assessment.

Smith, S. H. and D. E. Marx, Jr. (2016). De-facto marine protection from a Navy bombing range: Farallon de Medinilla, Mariana Archipelago, 1997 to 2012. *Marine Pollution Bulletin 102* (1): 187–198. DOI:10.1016/j.marpolbul.2015.07.023

Solan, M., C. Hauton, J. A. Godbold, C. L. Wood, T. G. Leighton, and P. White. (2016). Anthropogenic sources of underwater sound can modify how sediment-dwelling invertebrates mediate ecosystem properties. *Scientific Reports 6* (20540): 1–9. DOI:10.1038/srep20540

Sole, M., M. Lenoir, M. Durfort, M. Lopez-Bejar, A. Lombarte, and M. Andre. (2013). Ultrastructural damage of *Loligo vulgaris* and *Illex coindetii* statocysts after low frequency sound exposure. *PLoS ONE 8* (10). DOI:10.1371/journal.pone.0078825

Sole, M., M. Lenoir, J. M. Fontuno, M. Durfort, M. van der Schaar, and M. André. (2016). Evidence of Cnidarians sensitivity to sound after exposure to low frequency noise underwater sources. *Scientific Reports 6* (37979): 1–18. DOI:DOI: 10.1038/srep37979

Sole, M., P. Sigray, M. Lenoir, M. Van der Schaar, E. Lalander, and M. André. (2017). Offshore exposure experiments on cuttlefish indicate received sound pressure and particle motion levels associated with acoustic trauma. *Scientific Reports 7* (45899): 1–13.

Southall, B., A. Bowles, W. Ellison, J. Finneran, R. Gentry, C. Greene, D. Kastak, D. Ketten, J. Miller, P. Nachtigall, W. Richardson, J. Thomas, and P. Tyack. (2007). Marine mammal noise exposure criteria: Initial scientific recommendations. *Aquatic Mammals 33* (4): 122.

Southall, B. L., S. L. DeRuiter, A. Friedlaender, A. K. Stimpert, J. A. Goldbogen, E. Hazen, C. Casey, S. Fregosi, D. E. Cade, A. N. Allen, C. M. Harris, G. Schorr, D. Moretti, S. Guan, and J. Calambokidis. (2019). Behavioral responses of individual blue whales (*Balaenoptera musculus*) to mid-frequency military sonar. *Journal of Experimental Biology 222* (Pt 5). DOI:10.1242/jeb.190637

Spalding, M., M. Taylor, C. Ravilious, F. Short, and E. Green. (2003). Global overview: The distribution and status of seagrasses. In E. P. Green & F. T. Short (Eds.), *World Atlas of Seagrasses* (pp. 5–26). Berkeley, CA: University of California Press.

Spargo, B. J. (1999). *Environmental Effects of RF Chaff: A Select Panel Report to the Undersecretary of Defense for Environmental Security*. Washington, DC: U.S. Department of the Navy, Naval Research Laboratory.

Spargo, B. J. (2007). Personal Communication with Mark Collins Regarding Chaff End Cap and Piston Bouyancy.

Speed, C. W., M. G. Meekan, D. Rowat, S. J. Pierce, A. D. Marshall, and C. J. A. Bradshaw. (2008). Scarring patterns and relative mortality rates of Indian Ocean whale sharks. *Journal of Fish Biology 72* (6): 1488–1503. DOI:10.1111/j.1095-8649.2008.01810

Spiga, I. (2016). Acoustic response to playback of pile-driving sounds by snapping shrimp. In A. Popper & A. Hawkins (Eds.), *The Effects of Noise on Aquatic Life II, Advances in Experimental Medicine and Biology* (Vol. 875, pp. 1081–1088). New York, NY: Springer.

Srinivasan, A. and T. Viraraghavan. (2009). Perchlorate: Health effects and technologies for its removal from water resources. *International Journal of Environmental Research and Public Health 6* (4): 1418–1442. DOI:10.3390/ijerph6041418

St. Aubin, D. and L. A. Dierauf. (2001). Stress and Marine Mammals. In L. A. Dierauf & F. M. D. Gulland (Eds.), *Marine Mammal Medicine* (2nd ed., pp. 253–269). Boca Raton, FL: CRC Press.

Stalmaster, M. V. and J. L. Kaiser. (1997). Flushing responses of wintering bald eagles to military activity. *The Journal of Wildlife Management 61* (4): 1307–1313.

Stanley, J. A., C. A. Radford, and A. G. Jeffs. (2010). Induction of settlement in crab megalopae by ambient underwater reef sound. *Behavioral Ecology 21* (3): 113–120. DOI:10.1093/beheco/arp159

Steimle, F. W. and C. Zetlin. (2000). Reef Habitats in the Middle Atlantic Bight: Abundance, Distribution, Associated Biological Communities, and Fishery Resource Use. *Marine Fisheries Review 62* (2): 24–42.

Stepanuk, J. E., E. I. Heywood, J. F. Lopez, R. A. DiGiovanni Jr, and L. H. Thorne. (2021). Age-specific behavior and habitat use in humpback whales: Implications for vessel strike. *Marine Ecology Progress Series 663* 209–222.

Stevens, J. D. (2007). Whale shark (*Rhincodon typus*) biology and ecology: A review of the primary literature. *Fisheries Research 84* (1): 4–9. DOI:10.1016/j.fishres.2006.11.008

Stuhmiller, J. H., Y. Y. Phillips, and D. R. Richmong. (1991). The Physics and Mechanisms of Primary Blast Injury. In D. P. J. R. Zatchuck, R. F. Bellamy, and C. M. Quick (Ed.), *Textbook of Military Medicine. Part I. Warfare, Weapons, and the Casualty* (Vol. 5, pp. 241–270). Washington, DC: TMMM Publications.

Swisdak, M. M., Jr. and P. E. Montanaro. (1992). *Airblast and Fragmentation Hazards Produced by Underwater Explosions*. Silver Spring, MD: Naval Surface Warfare Center.

Swope, B. and J. McDonald. (2013). *Copper-Based Torpedo Guidance Wire: Applications and Environmental Considerations*. San Diego, CA: Space and Naval Warfare Systems Command Center Pacific.

Szesciorka, A. R., A. N. Allen, J. Calambokidis, J. Fahlbusch, M. F. McKenna, and B. Southall. (2019). A case study of a near vessel strike of a Blue Whale: Perceptual cues and fine-scale aspects of behavioral avoidance. *Frontiers in Marine Science 6*.

The Shark Trust. (2017). *Basking Shark Project: Human Impacts*. Retrieved May 30th, 2017, from http://www.sharktrust.org/en/basking_shark_human_impacts.

Titmus, A. J. and K. D. Hyrenbach. (2011). Habitat associations of floating debris and marine birds in the North East Pacific Ocean at coarse and meso spatial scales. *Marine Pollution Bulletin 62* (11): 2496–2506.

Touyz, R. M. (2004). Reactive oxygen species, vascular oxidative stress, and redox signaling in hypertension: What is the clinical significance? *Hypertension 44* 248–252. DOI:10.1161/01.HYP.0000138070.47616.9d

U.S. Air Force. (1997). *Environmental Effects of Self-Protection Chaff and Flares - Final Report*. Langley Air Force Base, VA: U.S. Air Force, Headquarters Air Combat Command.

U.S. Army Corps of Engineers. (2001). *Environmental Effects of Beach Nourishment Projects*.

U.S. Army Corps of Engineers. (2012). *New York and New Jersey Harbor Deepening Project: 2012 Benthic Recovery Report*. New York, NY: New York District.

U.S. Department of the Air Force. (1994). *Technical Reports on Chaff and Flares-Technical Report No. 5: Laboratory Analysis of Chaff and Flare Materials*. Langley Air Force Base, VA: U.S. Air Force Headquarters Air Combat Command.

U.S. Department of the Air Force. (1997). *Environmental Effects of Self-Protection Chaff and Flares*. Langley Air Force Base, VA: U.S. Air Force, Headquarters Air Combat Command.

U.S. Department of the Navy. (1975). *Explosion Effects and Properties Part I – Explosion Effects in Air*. Silver Spring, MD: White Oak Laboratory, Naval Surface Weapons Center.

U.S. Department of the Navy. (1993). *Report on Continuing Action: Standard Range Sonobuoy Quality Assurance Program, San Clemente Island, California*. San Diego, CA: Naval Facilities Engineering Command, Pacific.

U.S. Department of the Navy. (1996). *Environmental Assessment of the Use of Selected Navy Test Sites for Development Tests and Fleet Training Exercises of the MK-46 and MK-50 Torpedoes*. Pearl Harbor, HI: United States Command Pacific Fleet.

U.S. Department of the Navy. (1999). *Environmental Effects of RF Chaff: A Select Panel Report to the Undersecretary of Defense for Environmental Security*. Washington, DC: U.S. Department of the Navy, Naval Research Laboratory.

U.S. Department of the Navy. (2001a). *Airborne Mine Neutralization System (AMNS) Inert Target Tests: Environmental Assessment and Overseas Environmental Assessment*. Panama City, FL: Coastal Systems Station.

U.S. Department of the Navy. (2001b). *Final Environmental Impact Statement (EIS)/Overseas Environmental Impact Statement (OEIS) for the Shock Trial of the USS WINSTON S. CHURCHILL (DDG 81)*. Washington, DC: U.S. Department of the Navy.

U.S. Department of the Navy. (2001c). *Overseas Environmental Assessment (OEA) for Cape Cod TORPEDO EXERCISE (TORPEX) in Fall 2001*. Arlington, VA: Undersea Weapons Program Office.

U.S. Department of the Navy. (2002). *San Clemente Island Integrated Natural Resources Management Plan*. San Diego, CA: U.S. Department of the Navy Southwest Division.

U.S. Department of the Navy. (2006a). *2006 Supplement to the 2002 Rim of the Pacific (RIMPAC) Programmatic Environmental Assessment (PEA)*. Washington, DC: U.S. Department of the Navy.

U.S. Department of the Navy. (2006b). *Final Environmental Assessment, San Clemente Island Wastewater Treatment Plant: Increase in Maximum Allowable Discharge Volume*. San Diego, CA: Naval Facilities Engineering Command Southwest.

U.S. Department of the Navy. (2008a). *Final Atlantic Fleet Active Sonar Training Environmental Impact Statement/Overseas Environmental Impact Statement* (Chapter 4: Environmental Consequences). Norfolk, VA: Naval Facilities Engineering Command, Atlantic.

U.S. Department of the Navy. (2008b). *Southern California Range Complex Environmental Impact Statement/Overseas Environmental Impact Statement*. San Diego, CA: Naval Facilities Engineering Command Southwest.

U.S. Department of the Navy. (2009a). *Environmental Assessment for Construction & Operation of Electromagnetic Railgun Research, Development, Test, and Evaluation Facility MILCON P-306*. Dahlgren, VA: Naval Surface Warfare Center, Dahlgren Laboratory.

U.S. Department of the Navy. (2009b). *Swimmer Interdiction Security System (SISS) Final Environmental Impact Statement*. San Diego, CA: Space and Naval Warfare Systems Center.

U.S. Department of the Navy. (2010a). *Jacksonville Range Complex Operating Area Undersea Warfare Training Range Bottom Mapping and Habitat Characterization, Florida. Final Cruise Report*. Jacksonville, FL: Naval Facilities Engineering Command Atlantic.

U.S. Department of the Navy. (2010b). *Laser System Usage in the Marine Environment: Applications and Environmental Considerations*. San Diego, CA: Space and Naval Warfare Systems Command Center Pacific.

U.S. Department of the Navy. (2010c). *Water Range Assessment for the Jacksonville Range Complex*. Norfolk, VA: Parsons.

U.S. Department of the Navy. (2010d). *Water Range Assessment for the VACAPES Range Complex (Final Report)*. Norfolk, VA: Naval Facilities Engineering Command, Atlantic Division.

U.S. Department of the Navy. (2011). *CC Range Bottom Mapping and Habitat Characterization, Florida. Final Cruise Report*. Norfolk, VA: Naval Facilities Engineering Command Atlantic.

U.S. Department of the Navy. (2012a). *Ecosystem Technical Report for the Atlantic Fleet Training and Testing (AFTT) Draft Environmental Impact Statement*. Arlington, VA: Naval Facilities Engineering Command, Atlantic Division. Retrieved from http://www.warcosts.net/wp-content/uploads/us-navy/317BA_1_2012_AFTT_U.S._Navy_Atlantic_Fleet_Training_Testing_Ecosystem_Final_Technical_Report_April_13_2012.pdf.

U.S. Department of the Navy. (2012b). *Ecosystem Technical Report for the Atlantic Fleet Training and Testing (AFTT) Environmental Impact Statement*. Arlington, VA: Naval Facilities Engineering Command, Atlantic Division.

U.S. Department of the Navy. (2013a). *Comprehensive Exercise and Marine Species Monitoring Report for the U.S. Navy's Atlantic Fleet Active Sonar Training (AFAST) and Virginia Capes, Cherry Point, Jacksonville, and Gulf of Mexico Range Complexes 2009–2012*. Norfolk, VA: United States Fleet Forces Command.

U.S. Department of the Navy. (2013b). *Ecosystem Technical Report for the Hawaii-Southern California Training and Testing (HSTT) Environmental Impact Statement*. Pearl Harbor, HI: Naval Facilities Engineering Command, Pacific.

U.S. Department of the Navy. (2013c). *Military Expended Material Shrimp Fisheries Study in the U.S. South Atlantic and Eastern Gulf of Mexico*. Newport, RI: Naval Undersea Warfare Center Division.

U.S. Department of the Navy. (2013d). *San Diego Bay Integrated Natural Resources Management Plan*. San Diego, CA: Unified Port of San Diego.

U.S. Department of the Navy. (2013e). *Water Range Sustainability Environmental Program Assessment: Potomac River Test Range*. Dahlgren, VA: Naval Surface Warfare Center.

U.S. Department of the Navy. (2013f). *Water Range Sustainability Environmental Program Assessment: Potomac River Test Range*. Dahlgren, VA: Naval Surface Warfare Center.

U.S. Department of the Navy. (2017a). *Criteria and Thresholds for U.S. Navy Acoustic and Explosive Effects Analysis (Phase III)*. San Diego, CA: Space and Naval Warfare System Command, Pacific.

U.S. Department of the Navy. (2017b). *U.S. Navy Marine Species Density Database Phase III for the Hawaii-Southern California Training and Testing Study Area* (Naval Facilities Engineering Command Pacific Technical Report). Pearl Harbor, HI: Naval Facilities Engineering Command Pacific.

U.S. Department of the Navy. (2023). *Draft Environmental Assessment for Eelgrass Habitat Expansion in San Diego Bay, California*. San Diego, CA: Naval Facilities Engineering Systems Command Southwest.

U.S. Environmental Protection Agency. (2012). *EPA Federal Facilities Forum Fact Sheet*. Washington, DC: Solid Waste and Emergency Response.

U.S. Environmental Protection Agency. (2014a). *2014 Letter: Marine Protection, Research, and Sanctuaries Act (MPRSA) General Permit for the Sinking Exercise (SINKEX) Program; Evaluation, Determination, and Agreement*. Washington, DC: U.S. Department of the Navy.

U.S. Environmental Protection Agency. (2014b). *Technical Fact Sheet: Perchlorate*. Washington, DC: Office of Solid Waste and Emergency Response.

U.S. Fish and Wildlife Service. (2014). *West Indian Manatee (Trichechus manatus) Florida Stock (Florida subspecies, Trichechus manatus latirostris)*. Jacksonville, FL: U.S. Fish and Wildlife Service.

University of Hawaii. (2010). *Hawaii Undersea Military Munitions Assessment, Final Investigation Report HI-05, South of Pearl Harbor, Oahu, HI*. Honolulu, HI: University of Hawaii at Manoa.

van der Hoop, J., P. Corkeron, and M. Moore. (2017). Entanglement is a costly life-history stage in large whales. *Ecology and Evolution 7* (1): 92–106. DOI:https://doi.org/10.1002/ece3.2615

Van der Hoop, J. M., M. J. Moore, S. G. Barco, T. V. Cole, P. Y. Daoust, A. G. Henry, D. F. McAlpine, W. A. McLellan, T. Wimmer, and A. R. Solow. (2013). Assessment of management to mitigate anthropogenic effects on large whales. *Conservation Biology: The Journal of the Society for Conservation Biology 27* (1): 121–133. DOI:10.1111/j.1523-1739.2012.01934

Van der Hoop, J. M., A. S. M. Vanderlaan, and C. T. Taggart. (2012). Absolute probability estimates of lethal vessel strikes to North Atlantic right whales in Roseway Basin, Scotian Shelf. *Ecological Applications 22* (7): 2021–2033.

van Franeker, J. and K. Law. (2015). Seabirds, gyres and global trends in plastic pollution. *Environmental Pollution 203* 89–96. DOI:10.1016/j.envpol.2015.02.034

Van Waerebeek, K., A. N. Baker, F. Felix, J. Gedamke, M. Iñiguez, G. P. Sanino, E. Secchi, D. Sutaria, A. van Helden, and Y. Wang. (2007). Vessel collisions with small cetaceans worldwide and with large whales in the southern hemisphere, an initial assessment. *Latin American Journal of Aquatic Mammals 6* (1): 43–69.

Vanderlaan, A. S. M., J. J. Corbett, S. L. Green, J. A. Callahan, C. Wang, R. D. Kenney, C. T. Taggart, and J. Firestone. (2009). Probability and mitigation of vessel encounters with North Atlantic right whales. *Endangered Species Research 6* (3): 273–285. DOI:10.3354/esr00176

Vanderlaan, M. S. A. and T. C. Taggart. (2007). Vessel collisions with whales: The probability of lethal injury based on vessel speed. *Marine Mammal Science 23* (1): 144–156. DOI:10.1111/j.1748-7692.2006.00098

Vazzana, M., M. Celi, G. Maricchiolo, L. Genovese, V. Corrias, E. M. Quinci, G. de Vincenzi, V. Maccarrone, G. Cammilleri, S. Mazzola, G. Buscaino, and F. Filiciotto. (2016). Are mussels able to distinguish underwater sounds? Assessment of the reactions of *Mytilus galloprovincialis* after exposure to lab-generated acoustic signals. *Comparative Biochemistry and Physiology, Part A 201* 61–70. DOI:10.1016/j.cbpa.2016.06.029

Velando, A. and I. Munilla. (2011). Disturbance to a foraging seabird by sea-based tourism: Implications for reserve management in marine protected areas. *Biological Conservation 144* 1167–1174. DOI:10.1016/j.biocon.2011.01.004

Vermeij, M. J. A., K. L. Marhaver, C. M. Huijbers, I. Nagelkerken, and S. D. Simpson. (2010). Coral larvae move toward reef sounds. *PLoS ONE 5* (5): e10660. DOI:10.1371/journal.pone.0010660

Visser, I. N. and D. Fertl. (2000). Stranding, resighting, and boat strike of a killer whale (*Orcinus orca*) off New Zealand. *Aquatic Mammals 26.3* 232–240.

Votier, S. C., K. Archibald, G. Morgan, and L. Morgan. (2011). The use of plastic debris as nesting material by a colonial seabird and associated entanglement mortality. *Marine Pollution Bulletin 62* (1): 168–172. DOI:https://doi.org/10.1016/j.marpolbul.2010.11.009

Wale, M. A., S. D. Simpson, and A. N. Radford. (2013a). Noise negatively affects foraging and antipredator behaviour in shore crabs. *Animal Behaviour 86* 111–118.

Wale, M. A., S. D. Simpson, and A. N. Radford. (2013b). Size-dependent physiological responses of shore crabs to single and repeated playback of ship noise. *Biology Letters 9* (20121194).

Walker, M. M., J. L. Kirschvink, G. Ahmed, and A. E. Diction. (1992). Evidence that fin whales respond to the geomagnetic field during migration. *The Journal of Experimental Biology 171* 67–78.

Walker, S. W., C. L. Osburn, T. J. Boyd, L. J. Hamdan, R. B. Coffin, M. T. Montgomery, J. P. Smith, Q. X. Li, C. Hennessee, F. Monteil, and J. Hawari. (2006). *Mineralization of 2, 4, 6-Trinitrotoluene (TNT) in Coastal Waters and Sediments*. Washington, DC: U.S. Department of the Navy, Naval Research Laboratory.

Walker, W. A. and J. M. Coe. (1990). *Survey of Marine Debris Ingestion by Odontocete Cetaceans* (Proceedings of the Second International Conference on Marine Debris, 2–7 April 1989. Honolulu, Hawaii). Washington, DC: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center.

Wang, W. X. and P. S. Rainbow. (2008). Comparative approaches to understand metal bioaccumulation in aquatic animals. *Comparative Biochemistry and Physiology, Part C 148* (4): 315–323. DOI:10.1016/j.cbpc.2008.04.003

Wardle, C. S. (1986). Fish behaviour and fishing gear. In T. J. Pitcher (Ed.), *The Behavior of Teleost Fishes* (pp. 463–495). Baltimore, MD: The Johns Hopkins University Press.

Ware, C., D. N. Wiley, A. S. Friedlaender, M. Weinrich, E. L. Hazen, A. Bocconcelli, S. E. Parks, A. K. Stimpert, M. A. Thompson, and K. Abernathy. (2014). Bottom side-roll feeding by humpback whales (*Megaptera novaeangliae*) in the southern Gulf of Maine, U.S.A. *Marine Mammal Science 30* (2): 494–511. DOI:10.1111/mms.12053

Waters, D. F., C. P. Cadou, and W. E. Eagle. (2013). Quantifying unmanned undersea vehicle range improvement enabled by aluminum–water power system. *Journal of Propulsion and Power 29* (3): 675–685. DOI:10.2514/1.b34701

Watkins, W. A. (1986). Whale reactions to human activities in Cape Cod waters. *Marine Mammal Science 2* (4): 251–262.

Watkins, W. A., M. A. Daher, N. A. DiMarzio, A. Samuels, D. Wartzok, K. M. Fristrup, D. P. Gannon, P. W. Howey, and R. R. Maiefski. (1999). Sperm whale surface activity from tracking by radio and satellite tags. *Marine Mammal Science 15* (4): 1158–1180.

Wedemeyer, G. A., B. A. Barton, and D. J. McLeay. (1990). Stress and acclimation. In C. B. Schreck & P. B. Moyle (Eds.), *Methods for Fish Biology* (pp. 451–489). Bethesda, MD: American Fisheries Society.

Wei, C. L., G. T. Rowe, C. C. Nunnally, and M. K. Wicksten. (2012). Anthropogenic "Litter" and macrophyte detritus in the deep Northern Gulf of Mexico. *Marine Pollution Bulletin 64* (5): 966–973. DOI:10.1016/j.marpolbul.2012.02.015

Wells, R. S. and M. D. Scott. (1997). Seasonal incidence of boat strikes on bottlenose dolphins near Sarasota, Florida. *Marine Mammal Science 13* (3): 475–480.

Whitehead, H. (2003). *Sperm Whales Social Evolution in the Ocean*. Chicago, IL: University of Chicago Press.

Wilber, D., D. Clarke, G. Ray, and R. Dolah. (2009). *Lessons Learned from Biological Monitoring of Beach Nourishment Projects*. Tempe, AZ: Western Dredging Association.

Wilcox, C., E. Van Sebille, and B. Hardesty. (2015). Threat of plastic pollution to seabirds is global, pervasive, and increasing. *PNAS 112* (38): 11899–11904.

Wiley, D. N., C. A. Mayo, E. M. Maloney, and M. J. Moore. (2016). Vessel strike mitigation lessons from direct observations involving two collisions between noncommercial vessels and North Atlantic right whales (*Eubalaena glacialis*). *Marine Mammal Science 32* (4): 1501–1509. DOI:10.111/mms.12326

Williams, R., E. Ashe, and P. D. O'Hara. (2011). Marine mammals and debris in coastal waters of British Columbia, Canada. *Marine Pollution Bulletin 62* 1303–1316. DOI:10.1016/j.marpolbul.2011.02.029

Wilson, C. L., D. P. Arfsten, R. L. Carpenter, W. K. Alexander, and K. R. Still. (2002). Effect of Navy chaff release on aluminum levels in an area of the Chesapeake Bay. *Ecotoxicology Environmental Safety 52* (2): 137–142. DOI:10.1006/eesa.2002.2178

Wiltschko, R., S. Denzau, D. Gehring, P. Thalau, and W. Wiltschko. (2011). Magnetic orientation of migratory robins, *Erithacus rubecula*, under long-wavelength light. *The Journal of Experimental Biology 214* (18): 3096–3101. DOI:10.1242/jeb.059212

Wiltschko, W. and R. Wiltschko. (2005). Magnetic orientation and magnetoreception in birds and other animals. *Journal of Comparative Physiology A 191* (8): 675–693. DOI:10.1007/s00359-005-0627-7

Woodruff, D. L., V. I. Cullinan, A. E. Copping, and K. E. Marshall. (2013). *Effects of Electromagnetic Fields on Fish and Invertebrates* (Task 2.1.3: Effects of Aquatic Organisms, Fiscal Year 2012 Progress Report). Washington, DC: U.S. Department of Energy.

Wright, D. G. (1982). *A Discussion Paper on the Effects of Explosives on Fish and Marine Mammals in the Waters of the Northwest Territories* (Canadian Technical Report of Fisheries and Aquatic Sciences). Winnipeg, Canada: Western Region Department of Fisheries and Oceans.

Wright, S. L., D. Rowe, R. C. Thompson, and T. S. Galloway. (2013). Microplastic ingestion decreases energy reserves in marine worms. *Current Biology 23* (23): R1031–R1033. DOI:10.1016/j.cub.2013.10.068

Würsig, B. and W. J. Richardson. (2009). Noise, effects of. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 765–773). Cambridge, MA: Academic Press.

Yamashita, R., H. Takada, M. A. Fukuwaka, and Y. Watanuki. (2011). Physical and chemical effects of ingested plastic debris on short-tailed shearwaters, *Puffinus tenuirostris*, in the north Pacific Ocean. *Marine Pollution Bulletin 62* (12): 2845–2849.

Yelverton, J. T. and D. R. Richmond. (1981). *Underwater Explosion Damage Risk Criteria for Fish, Birds, and Mammals* [Type]. Presented at the 102nd Meeting of the Acoustical Society of America. Miami Beach, FL.

Young, G. A. (1991). *Concise Methods for Predicting the Effects of Underwater Explosions on Marine Life*. Silver Spring, MD: Naval Surface Warfare Center.

Zabawa, C. and C. Ostrom. (1980). *Final Report on the Role of Boat Wakes in Shore Erosion in Anne Arundel County, Maryland*. Annapolis, MD: Maryland Department of Natural Resources.

Zaitsev, Y. P. (1971). *Marine Neustonology*. Jerusalem, Israel: Israel Program for Scientific Translations.

Zellar, R., A. Pulkkinen, K. Moore, D. Reeb, E. Karakoylu, and O. Uritskaya. (2017). *Statistical Assessment of Cetacean Stranding Events in Cape Cod (Massachusetts, USA) Area OS21A-1345*. Greenbelt, MD: National Aeronautics and Space Administration.

Zhao, J. S., C. W. Greer, S. Thiboutot, G. Ampleman, and J. Hawari. (2004a). Biodegradation of the nitramine explosives hexahydro-1,3,5-triazine and octahydro-1,3,5,7-tetranitro-1,3,5,7-tetrazocine in cold marine sediment under anaerobic and oligotrophic conditions. *Canadian Journal of Microbiology 50* 91–96. DOI:10.1139/W03-112

Zhao, J. S., J. Spain, S. Thiboutot, G. Ampleman, C. Greer, and J. Hawari. (2004b). Phylogeny of cyclic nitramine-degrading psychrophilic bacteria in marine sediment and their potential role in the natural attenuation of explosives. *FEMS Microbiology Ecology 49* (3): 349–357. DOI:10.1016/j.femsec.2004.04.008

Zhao, S., C. Feng, W. Quan, X. Chen, J. Niu, and Z. Shen. (2012). Role of living environments in the accumulation characteristics of heavy metals in fishes and crabs in the Yangtze River Estuary, China. *Marine Pollution Bulletin 64* (6): 1163–1171. DOI:10.1016/j.marpolbul.2012.03.023

Zoeger, J., J. R. Dunn, and M. Fuller. (1981). Magnetic material in the head of the common Pacific dolphin. *Science 213* (4510): 892–894.