Appendix C Biological Resources Supplemental Information

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APPENDIX C Biological Resources Supplemental Information

C.1 Sediments and Water Quality

C.1.1 Introduction

C.1.1.1 Regulatory Environment

C.1.1.1.1 State Standards and Guidelines

State-level standards for sediments and water quality standards and guidelines begins with each state establishing a use for the water, which is referred to as its “designated” use. Examples of such uses of marine waters include fishing, shellfish harvesting, and recreation. For this appendix, a water body is considered “impaired” if any one of its designated uses is not met. The designated uses are declared under the Clean Water Act (CWA) by the states, because they are directed by the law, or allow the United States Environmental Protection Agency (USEPA) to designate the uses for them. Designating impaired waters comes under section 401 of the CWA and is delegated to the state by the USEPA. Applicable state standards and guidelines specific to each stressor are detailed in their respective subsections.

C.1.1.1.2 Federal Standards and Guidelines

Federal jurisdiction regarding sediments and water quality extends to 200 nautical miles (NM) along the Hawaiian Islands and the Pacific coast of the United States (U.S.). These standards and guidelines are mainly the responsibility of the USEPA, specifically ocean discharge provisions of the CWA (33 United States Code [U.S.C.] section 1343). The USEPA sets the water quality standards that include designated uses, criteria for pollutants that would protect those designated uses, establishes antidegradation policy and methods, and then implements the standards through permitting and other regulatory processes. Ocean disposal regulation is one of those implementation tools. The states or tribes may assume responsibility for implementing the water quality standards and they may establish more rigorous standards if that are science based, but the states must at minimum meet the USEPAs standards (with special process being available in the case where variances are needed). Ocean disposal may not result in: (1) unreasonable degradation on human health; (2) unacceptable negative effects on the marine ecosystem; (3) unacceptable negative persistent or permanent effects due to the particular volumes or concentrations of the dumped materials; and (4) unacceptable negative effects on the ocean for other uses as a result of direct environmental impact (40 Code of Federal Regulations [CFR] section 125.122). Proposed training and testing activities also occur beyond 200 NM. Even though CWA regulations may not apply, pertinent water quality standards are used as accepted scientific standards to assess potential impacts on sediments and water quality from the Proposed Action.

The International Convention for the Prevention of Pollution from Ships (Convention) addresses pollution generated by normal vessel operations. The Convention is incorporated into U.S. law as 33 U.S.C. sections 1901–1915. The Convention includes six annexes: Annex I, oil discharge; Annex II, hazardous liquid control; Annex III, hazardous material transport; Annex IV, sewage discharge; Annex V, plastic and garbage disposal; and Annex VI, air pollution. The U.S. Department of the Navy (Navy) is required to comply with the Convention; however, the U.S. is not a party to Annex IV. The discharge of sewage by military vessels is regulated by Section 312(d) of the CWA. The Convention contains handling requirements and specifies where materials can be discharged at sea, but it does not contain standards related to sediments or water quality.

The National Defense Authorization Act (NDAA) of 1996 amended section 312 of the CWA, directing the USEPA and the Department of Defense (DoD) to jointly establish the Uniform National Discharge Standards for discharges (other than sewage) incidental to the normal operation of military vessels. The Uniform National Discharge Standards program establishes national discharge standards for military vessels in U.S. coastal and inland waters extending seaward to 12 NM. Twenty-five types of discharges were identified as requiring some form of pollution control (e.g., a device or policy) to reduce or eliminate the potential for impacts. The discharges addressed in the program include, ballast water, deck runoff, and seawater used for cooling equipment. For a complete list of discharges refer to 40 CFR part 1700.4.

These national discharge standards reduce the environmental impacts associated with vessel discharges, stimulate the development of improved pollution control devices aboard vessels, and advance the development of environmentally sound military vessels.

C.1.1.2 Sediments

C.1.1.2.1 Characteristics of Sediment

Sediments consist of solid fragments of organic and inorganic matter forming the bottom, or substrate, of bodies of water. Sediments in the marine environment (e.g., in ocean basins) are either terrigenous, meaning that they originate from land, or are biogenic (i.e., formed from the remains of marine organisms). Terrigenous sediments come from the weathering of rock and other land-based substrates and are transported by water, wind, and ice (glaciers) to the seafloor. Biogenic sediments are produced in the oceans by the skeletal remains of single-celled benthic and planktonic organisms (e.g., foraminiferans and diatoms). When an organism dies, its remains are deposited on the seafloor. The remains are composed primarily of either calcium carbonate (e.g., a shell) or silica, and mixed with clays, form either a calcareous or siliceous ooze (Chester, 2003). Sediments in the Atlantic Ocean are predominantly composed of calcareous oozes and the Pacific Ocean has more siliceous oozes (Kennett, 1982). In addition to composition, sediments are also classified by size. Blott and Pye (2012) reviewed commonly used historical classification systems and offered a refined system that is adopted for describing sediments in this section. Sediments are grouped into five size classes: boulders, gravel, sand, silt, and clay. Sands range in size from 0.063 millimeter (mm) (very fine sands) to 2 mm (very coarse sands) (Figure C-1). For comparison, the thickness of a nickel is approximately 2 mm. Sediment types smaller than sands are silts (0.0020.063 mm in diameter) and clays (particles less than 0.002 mm in diameter). Sediments larger than sands are various types of gravel ranging in size from 2 mm (granules) to 64 mm (cobbles). Sediments greater than 64 mm in diameter are defined as boulders and range up to 2,048 mm (Blott & Pye, 2012; U.S. Department of Agriculture, 1993). Fine-grained silts and clays are often found mixed together in areas beyond the continental slope, such as on abyssal plains, and are referred to generally as mud (Kennett, 1982). Sediments in nearshore waters and on the continental shelf contain more sands that are primarily terrigenous, and sediments farther from shore in deep ocean basins are primarily biogenic. As organic and inorganic particles move downward through the water column and ultimately to the seafloor, many substances, including contaminants, that adhere to the particles and that are otherwise scarce in the water column become concentrated in bottom sediments (Chapman et al., 2003; Kszos et al., 2003).

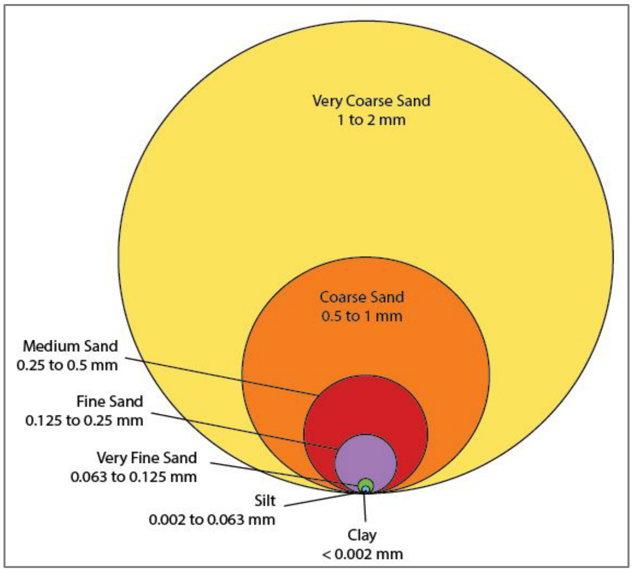

Figure C-1: Sediment Particle Size Comparison

C.1.1.2.2 Factors Affecting Marine Sediment Quality

The quality of sediments is influenced by their physical, chemical, and biological components; by where they are deposited; by the properties of seawater; and by other inputs and sources of contamination. Sediments tend to be dynamic, where factors affecting marine sediments often interact and influence each other. These factors are summarized below.

C.1.1.2.2.1 Physical Characteristics and Processes

At any given site, the texture and composition of sediments are important physical factors that influence the types of substances that are retained in the sediments, and subsequent biological and chemical processes. For example, clay-sized and smaller sediments and similarly sized organic particles tend to bind potential sediment contaminants and potentially limit their movement in the environment (U.S. Environmental Protection Agency, 2009). Conversely, fine-grained sediments are easily disturbed by currents and bottom-dwelling organisms, dredging, storms, and bottom trawling (Eggleton & Thomas, 2004; Hedges & Oades, 1997). Disturbance is also possible in deeper areas, where currents are minimal (Carmody et al., 1973), from mass wasting events such as underwater slides and debris flows (Coleman & Prior, 1988). If re-suspended, fine-grained sediments (and any substances bound to them) can be transported long distances.

C.1.1.2.2.2 Chemical Characteristics and Processes

The concentration of oxygen in sediments strongly influences sediment quality through its effect on the binding of materials to sediment particles. At the sediment surface, the level of oxygen is usually the same as that of the overlying water. Deeper sediment layers, however, are often low in oxygen (i.e., hypoxic) or have no oxygen (i.e., anoxic), and have a low oxidation-reduction potential, which predicts the stability of various compounds that regulate nutrient and metal availability in sediments. Certain substances combine in oxygen-rich environments and become less available for other chemical or biological reactions.

C.1.1.2.2.3 Biological Characteristics and Processes

Organic matter in sediment provides food for resident microbes. The metabolism of these microbes can change the chemical environment in sediments and thereby increase or decrease the mobility of various substances and influence the ability of sediments to retain and transform those substances (Mitsch et al., 2009a; U.S. Environmental Protection Agency, 2008a). Bottom-dwelling animals often rework sediments in the process of feeding or burrowing. In this way, marine organisms influence the structure, texture, and composition of sediments, as well as, the horizontal and vertical distribution of substances in the sediment (Boudreau, 1998). Moving substances out of or into low or no-oxygen zones in the sediment may alter the form and availability of various substances. The metabolic processes of bacteria also influence sediment components directly. For example, sediment microbes may convert mercury to methyl mercury, increasing its toxicity (Mitchell & Gilmour, 2008). However, it is more common that biological processes break down contaminants and reduce toxicity in sediments (White et al., 1997).

C.1.1.2.2.4 Location

The quality of coastal and marine sediments is influenced substantially by inputs from adjacent watersheds (Turner & Rabalais, 2003). Proximity to watersheds with large cities or intensively farmed lands often increases the amount of both inorganic and organic contaminants that find their way into coastal and marine sediments. A wide variety of metals and organic substances, such as polycyclic aromatic hydrocarbons, polychlorinated biphenyls (PCBs), and pesticides—often referred to collectively as “persistent organic pollutants”—are discharged into coastal waters by urban, agricultural, and industrial point and non-point sources in the watershed (U.S. Environmental Protection Agency, 2008a). Location on the ocean floor also influences the distribution and concentration of various elements through local geology and volcanic activity (Demina & Galkin, 2009), as well as through landslides and debris flow events (Coleman & Prior, 1988).

C.1.1.2.2.5 Other Contributions to Sediments

While the greatest mass of sediments is carried into marine systems by rivers (U.S. Environmental Protection Agency, 2008a), wind and rain also deposit materials in coastal waters and contribute to the mass and quality of sediments. For example, approximately 80 percent of the mercury released by human activities comes from coal combustion, mining and smelting, and solid waste incineration (Agency for Toxic Substances and Disease Registry, 1999). These activities are generally considered to be the major sources of mercury in marine systems (Fitzgerald et al., 2007). Atmospheric deposition of lead is similar in that human activity is a major source of lead in sediments (Wu & Boyle, 1997).

C.1.1.3 Water Quality

The discussion of water quality begins with an overview of the characteristics of marine waters, including pH (a measure of acidity), temperature, oxygen, nutrients (e.g., chlorophyll), salinity, and dissolved elements. The discussion then considers how those characteristics of marine waters are influenced by physical, chemical, and biological processes.

C.1.1.3.1 Characteristics of Marine Waters

The composition of water in the marine environment is determined by complex interactions among physical, chemical, and biological processes. Physical processes include region-wide currents and tidal flows, seasonal weather patterns and temperature, sediment characteristics, and unique local conditions, such as the volume of fresh water delivered by large rivers. Chemical processes involve salinity, pH, dissolved minerals and gases, particulates, nutrients, and pollutants. Biological processes involve the influence of living things on the physical and chemical environment. The two dominant biological processes in the ocean are photosynthesis and respiration, particularly by microorganisms. These processes involve the uptake, conversion, and excretion of waste products during growth, reproduction, and decomposition (Mann & Lazier, 1996).

C.1.1.3.2 Influence of Marine Properties and Processes on Seawater Characteristics

Ocean currents and tides mix and redistribute seawater. In doing so, they alter surface water temperatures, transport and deposit sediment, and concentrate and dilute substances that are dissolved and suspended in the water. These processes operate to varying degrees from nearshore areas to the abyssal plain. Salinity also affects the density of seawater and, therefore, its movement relative to the sea surface (Libes, 2009). Upwelling brings cold, nutrient-rich waters from deeper areas, increasing the productivity of local surface waters (Mann & Lazier, 1996). Storms and hurricanes also cause strong mixing of marine waters (Li et al., 2008).

Temperature and pH influence the behavior of trace metals in seawater, such as the extent to which they dissolve in water (i.e., the metal’s solubility) or their tendency to adsorb to organic and inorganic particles. However, the degree of influence differs widely among metals (Byrne, 1996). The concentration of a given element may change with position in the water column. For example, some metals (e.g., cadmium) are present at low concentrations in surface waters and at higher concentrations at depth (Bruland, 1992), while others decline quickly with increasing depth below the surface (e.g., zinc and iron) (Morel & Price, 2003; Nozaki, 1997). On the other hand, dissolved aluminum concentrations are highest at the surface, lowest at mid-depths, and increase again at depths below about 1,000 meters (m) (Li et al., 2008).

Substances, such as nitrogen, carbon, silicon, and trace metals are extracted from the water by biological processes. Others, like oxygen and carbon dioxide (CO_2), are produced by biological processes. Metabolic waste products add organic compounds to the water, and may also absorb trace metals, removing those metals from the water column. Those organic compounds may then be consumed by biological organisms, or they may aggregate with other particles and sink (Mann & Lazier, 1996; Wallace et al., 1977).

Runoff from coastal watersheds influences local and regional coastal water conditions, especially large rivers. Influences include increased sediments and pollutants, and decreased salinity (Rabalais et al., 2002; Turner & Rabalais, 2003; Wiseman & Garvine, 1995). Coastal bays and large estuaries serve to filter river outflows and reduce total discharge of runoff to the ocean (Edwards et al., 2006; Mitsch et al., 2009a). Depending on their structure and components, estuaries can directly or indirectly affect coastal water quality by recycling various compounds (e.g., excess nutrients), sequestering elements in more inert forms (e.g., trace metals), or altering them, such as the conversion of mercury to methyl mercury (Mitchell & Gilmour, 2008; Mitsch & Gosselink, 2007).

C.1.1.3.3 Coastal Water Quality

Most water quality problems in coastal waters of the U.S. are from degraded water clarity or increased concentrations of phosphates or chlorophyll-*a* (U.S. Environmental Protection Agency, 2012). Water quality indicators measured are dissolved inorganic nitrogen, dissolved inorganic phosphorus, water clarity or turbidity, dissolved oxygen (DO), and chlorophyll-*a*. Chlorophyll-*a* is an indicator of microscopic algae (phytoplankton) abundance used to judge nutrient availability (i.e., phosphates and nitrates). Excess phytoplankton blooms can decrease water clarity and, when phytoplankton die off following blooms, lower concentrations of DO. Most sources of these impacts arise from on-shore point and non-point sources of pollution. Point sources are direct water discharges from a single source, such as industrial or sewage treatment plants, while non-point sources are the result of many diffuse sources, such as runoff caused by rainfall.

C.1.2 Methods

The following four stressors may impact sediments or water quality: (1) explosives and explosives byproducts, (2) metals, (3) chemicals other than explosives, and (4) other materials (e.g., plastics). The term “stressor” is used because the military expended materials in these four categories may affect sediments or water quality by altering their physical or chemical characteristics. The potential impacts of these stressors are evaluated based on the extent to which the release of these materials could directly or indirectly impact sediments or water quality such that existing laws or standards would be violated or recommended guidelines would be exceeded. The differences between standards and guidelines are described below.

- **Standards** are established by law or through government regulations that have the force of law. Standards may be numerical or narrative. Numerical standards set allowable concentrations of specific pollutants (e.g., micrograms per liter [μg/L]) or levels of other parameters (e.g., pH) to protect the water’s designated uses. Narrative standards describe water conditions that are not acceptable.
- **Guidelines** are non-regulatory, and generally do not have the force of law. They reflect an agency’s preference or suggest conditions that should prevail. Guidelines are often used to assess the condition of a resource to guide subsequent steps, such as the disposal of dredged materials. Terms such as screening criteria, effect levels, and recommendations are also used.

C.1.2.1 Intensity and Duration of Impact

The intensity or severity of impacts is defined as follows (listed by increasing level of impact):

- Chemical, physical, or biological changes in sediment or water quality would not be detectable as a result of the use of military materials.
- Chemical, physical, or biological changes in sediments or water quality would be measurable, but total concentrations would not violate applicable standards, regulations, and guidelines. Sediment and water quality would be equivalent to existing conditions, and designated uses of the water body or substrate would not change.
- Chemical, physical, or biological changes in sediments or water quality would be measurable and readily apparent but total concentrations would not violate applicable standards, regulations, and guidelines. Sediment or water quality would be altered compared to the historical baseline or desired conditions, and designated uses of the water body or substrate would be changed. Mitigation would be necessary and would likely be successful.
- Chemical, physical, or biological changes in sediment or water quality would be readily measurable, and some standards, regulations, and guidelines would be periodically approached, equaled, or exceeded as measured by total concentrations. Sediment or water quality would be frequently altered from the historical baseline or desired conditions, and designated uses of the water body or substrate would be changed. Mitigation measures would be necessary to limit or reduce impacts on sediment or water quality, although the efficacy of those measures would not be assured.

Duration is characterized as either short-term or long-term. Short-term is defined as days or months. Long-term is defined as months or years, depending on the type of activity or the materials involved.

C.1.2.2 Measurement and Prediction

Many of the conditions discussed above often influence each other, so measuring and characterizing various substances in the marine environment is often difficult (Byrne, 1996; Ho et al., 2007). For instance, sediment contaminants may change over time. Valette-Silver (1993) reviewed several studies that demonstrated the gradual increase in a variety of contaminants in coastal sediments that began as early as the 1800s, continued into the 1900s, peaked between the 1940s and 1970s, and declined thereafter (e.g., lead, dioxin, PCBs). After their initial deposition, normal physical, chemical, and biological processes can re-suspend, transport, and redeposit sediments and associated substances in areas far removed from the source (Hameedi et al., 2002; U.S. Environmental Protection Agency, 2012). The conditions noted above further complicate predictions of the impact of various substances on the marine environment.

C.1.2.3 Sources of Information

Relevant literature was systematically reviewed to complete this analysis of sediments and water quality. The review included journals, technical reports published by government agencies, work conducted by private businesses and consulting firms, U.S. DoD reports, operational manuals, natural resource management plans, and current and prior environmental documents for facilities and activities in the Study Area.

Because of the proximity of inshore and nearshore areas to humans, information on the condition of sediments and water quality in those areas tends to be relatively readily available. However, much less is known about deep ocean sediments and open ocean water quality. Since sediments and water quality in inshore and nearshore areas tend to be affected by various human social and economic activities, two general assumptions are used in this discussion: (1) sediments and water quality generally improve as distance from the shore increases; and (2) sediments and water quality generally improve as depth increases.

C.1.3 General Background

C.1.3.1 Sediments

C.1.3.1.1 Sediment Quality in the Nearshore and Offshore Regions of the Hawaiian Islands

The 2012 National Coastal Condition Report IV is the most recent, comprehensive sediment quality analysis of nearshore and offshore Hawaiian Islands. The two sites receiving a poor rating were in Waimea Bay, Kauai where chromium concentrations exceeded the level where adverse effects are likely to occur in 50 percent of samples. Sampling sites in Pearl Harbor, Keehi Lagoon on Oahu, Hilo Bay on Hawaii, and other harbor areas exceeded effects levels for individual metals. For total organic carbon, 12 percent of coastal waters were rated poor and 18 percent were rated fair. Some of the same areas with relatively high concentrations of contaminants in sediments also had higher concentrations of total organic carbon, including Keehi Lagoon and Hilo Bay. Suburban development east of Honolulu contributed to higher levels of total organic carbon in adjacent coastal waters. Higher levels of total organic carbon in sediments can be an indicator of higher concentrations of chemical pollutants and poor sediment quality (U.S. Environmental Protection Agency, 2012).

Some metals naturally occur at elevated concentrations in the volcanic soils of Hawaii. Natural concentrations of copper, zinc, nickel, and chromium are high compared to soils in the mainland U.S.

Pearl Harbor receives a substantial amount of metal contamination because it serves as a natural trap for sediment particles (U.S. Environmental Protection Agency, 2008b). See Table C-1 for sediment quality criteria and index by site and by region.

Table C-1: Sediment Quality Criteria and Index, United States West Coast and Hawaiian Islands

Parameter	Site Criteria			Regional Criteria		
	Good	Fair	Poor	Good	Fair	Poor
Sediment Toxicity	Amphipod[1] survival rate ≥ 80%	n/a	Amphipod[1] survival rate < 80%	< 5% of coastal area in poor condition	n/a	≥ 5% of coastal area in poor condition
Sediment Contaminants	No ERM[2] concentration exceeded, and < 5 ERL[3] concentrations exceeded	No ERM[2] concentration exceeded and ≥ 5 ERL[3] concentrations exceeded	An ERM[2] concentration exceeded for one or more contaminants	< 5% of coastal area in poor condition	5–15% of coastal area in poor condition	> 15% of coastal area in poor condition
Excess Sediment TOC	TOC concentration < 2%	TOC concentration 2% to 5%	TOC concentration > 5%	< 20% of coastal area in poor condition	20–30% of coastal area in poor condition	> 30% of coastal area in poor condition
Sediment Quality Index	No poor ratings, sediment contaminants criteria are rated "good"	No poor ratings, sediment contaminants criteria are rated "fair"	One or more individual criteria rated poor	< 5% of coastal area in poor condition, and > 50% in good condition	5–15% of coastal area in poor condition, and > 50% in combined fair and poor condition	> 15% of coastal area in poor condition

Source: (U.S. Environmental Protection Agency, 2009, 2012); State of California (2009)

[1] Amphipods are small animals found in a wide variety of aquatic habitats. Because they are so widely distributed, they are often used as an indicator of toxicity in sediments and water bodies.

[2] ERM (effects range-median) is the level measured in the sediment below which adverse biological effects were measured 50 percent of the time.

[3] ERL (effects range-low) is the level measured in the sediment below which adverse biological effects were measured 10 percent of the time Long et al. (1995).

Notes: % = percent. ≥ = equal to or greater than, < = less than, > = greater than, n/a = not applicable, TOC = total organic carbon

Anthropogenic activities within and around Pearl Harbor, including Navy activities and private industrial, commercial, and agricultural activities, contribute pollutants through point and non-point sources. These activities release numerous pollutants into Pearl Harbor, where sediments can act as a sink or repository for chemicals. The Navy conducted a Remedial Investigation/Feasibility Study of the sediments in Pearl Harbor from March to June 2009. The results of the Remedial Investigation indicated that eight metals (antimony, cadmium, copper, lead, mercury, selenium, silver, and zinc), total high molecular weight polycyclic aromatic hydrocarbons, total PCBs, and two chlorinated pesticides (dieldrin and total endosulfan) exceeded the project screening criteria (Table C-2) (U.S. Department of the Navy, 2010a). Surface weighted-average concentrations in sediment were below project screening criteria in Middle Loch and West Loch and above project screening criteria in Southeast Loch, Bishop Point, northwest shoreline of Ford Island, Aiea Bay, shoreline of Oscar 1 and 2, and off the Waiau Power Plant (U.S. Department of the Navy, 2010a).

In 2012, the Space and Naval Warfare Systems Command conducted field measurements on the resuspension of sediments from propeller wash in DoD harbors, including Pearl Harbor (Wang et al., 2014a). Background concentration of contaminants were measured in sediments prior to conducting the study (Table C-2). In an earlier study, (Wang et al., 2009), estimated that transiting Navy (and other military) vessels in Pearl Harbor resuspended 54 tons of sediments per day, which amounts to more than 10 percent of the sediment load from the entire Pearl Harbor watershed. Wang et al. (2014a) measured sediment resuspension and associated metal contaminants from a tugboat propeller wash at two piers, Bravo Pier and Oscar Pier in Pearl Harbor; measurements included the concentrations of the metals chromium, nickel, copper, zinc, arsenic, silver, cadmium, and lead, in the sediment plume. The concentrations of chromium and nickel were the only ones consistently above USEPA water quality criteria following the resuspension events (Wang et al., 2014a). Between 65 and 90 percent of metal concentrations, depending on the metal, were in the dissolved phase, rather the particle-bound phase, contributing to greater dispersion within the harbor. The data were used as input and validation of a fate and transport model, which predicted that resuspended metals can be transported and dispersed far from the piers, and, over several days, throughout much of the harbor, potentially resulting in recontamination of remediated areas as well as increased contaminant concentrations in more remote areas of the harbor (Wang et al., 2014a).

The Hawaii Undersea Military Munitions Assessment is a comprehensive effort to characterize the potential impacts of chemical and conventional munitions disposed of at sea in a deepwater environment (Edwards et al., 2016). The program collected data in a location south of Pearl Harbor, Oahu between Barber's Point and Diamond Head from 2007 to 2012 with the goals of defining the bounds of the disposal site, characterizing the state of the munitions found on the seafloor, and assessing the potential impacts that degrading munitions may have on the benthic environment. Researchers mapped the disposal site using high-resolution acoustic imaging, took thousands of digital photos and recorded hundreds of hours of video, and collected physical samples within two meters of munitions to assess sediment contamination. Concentrations of metals detected in sediments at the disposal sites were similar to samples taken from nearby (within 50 m) control sites (Briggs et al., 2016). The chemical warfare agent sulfur mustard and its degradation products were detected as a thin dust-like coating on bottom sediments near chemical munitions. There appeared to be no vertical mixing with adjacent sediments, and the combination of the chemical's low water solubility, the formation of a protective coating by the products of hydrolysis, and near freezing temperatures at the site (greater than 250 m depth), likely resulted in the chemical's persistence as a thin coating (Briggs et al., 2016). There were very few detections of energetic compounds (e.g., explosive materials) at the disposal sites, leading researchers to conclude that the compounds remain contained within the munitions casing or were widely dispersed or degraded before samples were taken.

Following these studies and reports, in September of 2018, the Final Record of Decision Pearl Harbor Sediment for Joint Base Pearl Harbor-Hickan (JBPHH) Oahu, Hawaii was released for the remediation of the Pearl Harbor Sediment Site 19 (Naval Facilities Engineering Command Pacific, 2018). Selected remediation activities selected for the site include focused dredging, institutional controls, long-term monitoring, inspections, and five-year reviews. At the time of publication of this report, remedial action was ongoing at the site.

In 2019, the Naval Facilities Engineering Command Pacific published the Basis of Design, Field Investigation Report for Pearl Harbor sediment. This report supports the development of remediation activities by detailing condition of focus areas within the harbor. Sediments were found to have harmful concentrations of chemicals of concern in six locations within the harbor: Southeast Loch, Oscar 1 and 2 Piers Shoreline, Off Ford Island Landfill and Camel Refurbishing Area, Bishop Point, Off Waiau Power Plant, and Aiea Bay. Chemicals of concern identified in these locations include metals and PCBs. For these locations, remedial alternatives may include focused dredging, enhanced natural recovery, in-situ treatment with activated carbon amendment, and monitored natural recovery (AECOM Technical Services, 2019).

Table C-2: Sediment Screening Criteria and Background Concentrations for Pearl Harbor Sediment Remedial Investigation

Parameter		Pier in Pearl Harbor	Background Concentrations (ppm)	Sediment Screening Criteria (mg/kg [ppm], dry weight)
Metals	Antimony	Bravo	Not Measured	8.4
		Oscar		
	Arsenic	Bravo	13.0	27.5
		Oscar	10.5	
	Cadmium	Bravo	0.82	3.2
		Oscar	0.41	
	Chromium	Bravo	86.4	277
		Oscar	51.8	
	Copper	Bravo	97.8	214
		Oscar	49.0	
	Lead	Bravo	53.0	119
		Oscar	41.8	
	Mercury	Bravo	Not Measured	0.71
		Oscar		
	Nickel	Bravo	54.0	660
		Oscar	33.2	
	Selenium	Bravo	Not Measured	3.8
		Oscar		
	Silver	Bravo	0.67	1.8
		Oscar	0.32	
	Zinc	Bravo	290	330
		Oscar	225	
Organic Compounds	HMW-PAHs	N/A	Not Measured	35,253
	Total PCBs	Bravo/Oscar	ND	92 (> 2 m water depth) 29 (< 2 m water depth)

Parameter		Pier in Pearl Harbor	Background Concentrations (ppm)	Sediment Screening Criteria (mg/kg [ppm], dry weight)
Pesticides	Total DDT	Bravo/Oscar	Not Measured	106.6
	Dieldrin	Bravo/Oscar	ND	14.4
	Total BHC	Bravo/Oscar	Not Measured	1,215
	Total Chlordane	Bravo/Oscar	Not Measured	174
	Heptachlor Epoxide	Bravo/Oscar	Not Measured	174
	Total Endosulfan	Bravo/Oscar	ND	1.09
Dioxins	2,3,7,8-TCDD	Bravo/Oscar	Not Measured	0.36

Sources: U.S. Department of the Navy (2010a), Wang et al. (2014a)
Notes: mg = milligram, kg = kilogram, ppm = parts per million, HMW-PAH = high molecular weight-polyaromatic hydrocarbons, PCBs = polychlorinated biphenyls, DDT = dichlorodiphenyltrichloroethane, BHC = benzene hexachloride, TCDD = tetrachlorodibenzo-p-dioxin, < = less than, > = greater than, N/A = not applicable, ND = Not Detected

C.1.3.1.2 Sediment Quality in the Nearshore and Offshore Regions of the California Study Area

Within the West Coast Region, only two sites, both in the Channel Islands, received a poor rating for total organic carbon. Although these sites are located adjacent to the Study Area neither fall within the Study Area boundaries.

In a report on the *Southern California Bight 2018 Regional Monitoring Program*, the Southern California Coastal Water Research Project found that 79 percent of sediments in the Southern California Bight have minimal or low contamination, and less than 1 percent have high contamination, the worst category (Du et al., 2018). The Study Area overlaps with approximately the southern half of the Bight, from just north of Santa Catalina Island to the U.S.-Mexico border. Higher levels of sediment contamination occurred generally in nearshore embayments rather than in offshore sediments on the continental shelf and slope, and the distribution of contaminants was dependent on the location of the source of the contaminant. For example, concentrations of dichlorodiphenyltrichloroethane (DDT) are higher in sediments off Los Angeles due to long-term discharges from the Los Angeles sanitation district ocean outfall, whereas copper concentrations are higher in sediments in San Diego Bay, which is home to several large marinas, due to the use of anti-fouling paints on recreational and commercial vessels (Du et al., 2018).

Overall, trends for the entire Bight have been stable since 2003, but the sediment condition for some habitats within the Bight has changed. For example, the spatial extent of sediments with acceptable chemistry in ports, bays, and marinas steadily improved from 40 percent in 1998 to 72 percent in 2013, then decreased 53 percent in 2018. The extent of acceptable sediment chemistry in estuaries and continental shelf sediments remained generally unchanged from 2003 to 2018 (Du et al., 2018).

In 2018, for the first time, the Southern California Coastal Water Research Project surveyed for fipronil, a new pesticide product. Overall, fipronil pesticides were not widely detected (Du et al., 2018).

C.1.3.1.2.1 Sediment Quality off San Clemente Island and the Silver Strand Training Complex

Sediment quality in the waters surrounding San Clemente Island (SCI) was tested in 2006 (U.S. Department of the Navy, 2006a); concentrations for all contaminants were well below USEPA sediment quality guidelines (Effects Range Median values) (Table C-3). The 10-day solid-phase amphipod bioassay tests of the sediments also indicated high survival and no substantial toxicity. The results indicate that ocean bottom sediment quality is good off SCI, including areas were training and testing activities occur.

An Area of Special Biological Concern has been designated by the California State Water Resources Control Board to include nearshore waters around SCI out to 1 NM from shore or to the 300 feet (ft.) isobath, whichever is greater, along the island's 58-mile (mi.) coastline. The designation prohibits all waste discharges, both point and non-point, with the exception of a 1,000 ft. radius area at Wilson Cove where the wastewater treatment plant is located (U.S. Department of the Navy, 2013d). A 2011 survey of intertidal habitat and associated biological communities noted no substantial differences between species richness at a discharge site and a reference site, supporting the 2006 data indicating that low contaminant levels good sediment quality (U.S. Department of the Navy, 2013d).

Table C-3: Contaminant Concentrations in Bottom Sediments Offshore of San Clemente Island

Constituent	Sediment Concentration at SCI Reference Sampling Site (ppm)	USEPA Sediment Quality Guidelines (ERM Values) (ppm)
Arsenic	2.87	70
Cadmium	0.11	9.6
Chromium	8.56	370
Copper	7.48	270
Lead	2.19	218
Mercury	0.275	0.71
Nickel	4.6	51.6
Selenium	0.56	n/a
Silver	0.09	3.7
Zinc	19.2	410
Polychlorinated biphenyls	ND (< 0.005)	180
Phenols	ND (< 0.1)	n/a
Dioxins (TEQ)	0.0–0.028	n/a

Sources: (National Oceanic and Atmospheric Administration, 1999; U.S. Department of the Navy, 2006a)
Notes: ppm = parts per million, ERM = Effects Range Median, ND = not detectable concentration, n/a = not available, TEQ = toxicity equivalency factor, SCI = San Clemente Island, USEPA = United States (U.S.) Environmental Protection Agency, < = less than

Pacific Ocean sediments offshore of Silver Strand have above-average levels of organic loading and concentrations of some metals (aluminum, arsenic, chromium, copper, iron, manganese, and zinc), but these substances are not present at concentrations that pose a risk to public health or the environment. Traces of synthetic organic contaminants (e.g., chemicals released from the burning of coal) are occasionally detected in sediments, but have been well below a threshold of concern (U.S. Army Corps of Engineers, 2002, 2012). Concentrations of contaminants and particulate organic matter are highly variable due to changes in the outflow from the Tijuana River, which can increase substantially following heavy rainfall events (Svejkovsky et al., 2010). Sediment sampling in San Diego Bay near Silver Strand Training Complex-North indicates that—while concentrations of some contaminants are elevated above background levels—no contaminants were present at concentrations which would adversely affect marine organisms (U.S. Department of the Navy, 2013d).

C.1.3.1.2.2 Sediment Quality in San Diego

While multiple sources of pollution contribute to contaminants in the bay, including recreational, commercial, and Navy vessels urban runoff is the largest source of pollutants in the bay, contributing more heavy metals than all other sources combined. Despite reductions in the production and use of polybrominated diphenyl ether flame retardants, some of the highest concentrations of the contaminant in the Southern California Bight were reported in San Diego Bay (Dodder et al., 2016). In the past, sources of sediment contamination other than urban runoff in San Diego Bay have included sewage, industrial wastes, discharges from ships, and accidental spills of contaminants (e.g., oil or fuel). Progress has been made to eliminate or reduce the likelihood of these sources of pollutants entering the bay; however, many residual contaminants remain imbedded in bay sediments (Thompson et al., 2009; Wang et al., 2014a; Wang et al., 2000). Current sources of pollutants (other than urban runoff) include resuspension of sediments, industries surrounding and using the bay, Navy installations and activities in the bay, underwater hull cleaning, and vessel anti-fouling paints (Wang et al., 2006; Wang et al., 2000).

Known contaminants found in sediments in San Diego Bay include arsenic, copper, chromium, lead, cadmium, selenium, mercury, tin, manganese, silver, zinc, polycyclic aromatic hydrocarbons, petroleum hydrocarbons, PCBs, chlordane, dieldrin, and DDT (Dodder et al., 2016; Neira et al., 2009; U.S. Department of the Navy, 2000). Sediment sampling in the 1990s revealed that sediment quality indicators were exceeded at all San Diego Bay sampling stations and the number of exceedances was high at most stations (U.S. Department of the Navy, 2013d). Chlordane, polycyclic aromatic hydrocarbons, and PCBs were the pollutants most often found at elevated concentrations. Copper, lead, mercury and zinc were often found at elevated levels in Naval Shipyard areas, although the data indicate the probability of metal toxicity was low in those areas (U.S. Department of the Navy, 2013d).

Copper concentrations in marinas in San Diego Bay have frequently exceeded water quality standards (Biggs & D’Anna, 2012). Increasing copper concentrations in sediments at Shelter Island marina, a small, manmade basin with only one opening to the bay, coincided spatially with a higher concentration of boats in the marina (Neira et al., 2009). A second study measured copper concentrations before and after boat slips were occupied at the Pier 32 Marina near the middle of San Diego Bay and adjacent to the Sweetwater National Wildlife Refuge (Biggs & D’Anna, 2012). This study provided further confirmation that elevated concentrations of copper in water and sediments are primarily due to copper leaching from boat paints used on recreational and commercial vessels (Biggs & D’Anna, 2012). A follow-on study in the Shelter Island marina by Neira et al. (2009) showed that the elevated copper levels in sediments had widespread impacts on the benthic faunal community in the marina. While the proposed Navy training and testing activities would not use either marina, the studies indicate that elevated copper concentrations in sediments continues to be a concern and is likely to occur in other locations within the bay.

The San Diego Regional Water Quality Control Board assessed sediment contamination data from 161 sampling stations across San Diego Bay to determine the effects of sediment contamination on benthic macrofauna at-large and previously identified sediment clean-up sites (Thompson et al., 2012). The concentrations of 10 contaminants, 5 metals (Cadmium, copper, lead, mercury and zinc) and 5 organic compounds (total chlordanes, DDT, PCBs, high molecular weight polycyclic aromatic hydrocarbons, and low molecular weight polycyclic aromatic hydrocarbons), were analyzed and ranked on a scale of 1 to 5; sediments receiving a score of 1 were un-impacted and sediments receiving a score of 5 were clearly impacted. The impact score rated the likelihood that the level of contamination would impact benthic macrofauna. Thompson et al. (2012) cites several studies that show sediment toxicity and the probability of associated impacts on biological resources are better represented by indicators that represent mixtures of contaminants rather than concentrations of individual contaminants (see, for example, (Carr et al., 1996; Thompson et al., 2009). One such indicator is the mean Effects Range Median Quotient, which Thompson et al. (2012) uses to evaluate sediment quality in the bay.

With the exception of polycyclic aromatic hydrocarbons, the mean concentrations of contaminants in impacted sediments were between the Effects Range Low and Effects Range Median concentrations (Table C-4). The mean concentrations of chlordanes, copper, DDT, and mercury in sediment samples characterized as “un-impacted” exceeded the Effects Range Low value, suggesting some tolerance by biota. Biological impacts correlated more closely with high concentrations of mixtures of contaminants rather than individual contaminants, leading Thompson et al. (2012) to recommend using an indicator such as the mean Effects Range Median quotient as an indicator of sediment quality rather than basing an impacts assessment on the concentrations of individual contaminants in sediments. Based on USEPA guidelines for sediment chemistry using the mean Effects Range Median quotient, the sediment quality in San Diego Bay would be considered fair (i.e., mERMq is between 0.1 and 0.5) (U.S. Environmental Protection Agency, 2016).

Table C-4: Mean Concentration of Contaminants in San Diego Bay for Un-Impacted and Impacted Sediments with Comparison to Effects Ranges

Contaminant	Units	N	Mean Concentration of Un-impacted Samples (Score 1 to 2)	N	Mean Concentration of Impacted Samples (Score 3 to 5)	ERL	ERM
Cadmium	µg/g	72	0.209	89	0.342	1.2	9.6
Chlordane	ng/g	57	1.393	71	4.995	0.5	6
Copper	µg /g	72	77.302	89	153.159	34	270
DDTs	ng/g	57	2.362	72	7.302	1.58	46.1
HPAH	ng/g	72	407.126	89	1234.824	1,700	9,600
Lead	µg /g	72	30.528	89	62.71	46.8	218
LPAH	ng/g	72	99.0169	89	202.978	552	3,160
Mercury	µg /g	72	0.314	89	0.489	0.15	0.71
PCBs	ng/g	72	19.72	89	58.68	22.7	180
Zinc	µg /g	72	136.38	89	256.448	150	410
mERMq	ng/g	72	0.166	89	0.332	NA	NA

Source: Thompson et al. (2012)

Notes: DDT = dichlorodiphenyltrichloroethane, HPAH = high molecular weight polycyclic aromatic hydrocarbons, LPAH = Low PAH, PCB = polychlorinated biphenyls, ERM = Effects Range Median, mERMq = mean ERM quotient, ERL = Effects Range Low, µg /g = micrograms per gram, ng/g = nanograms per gram, N = Number of Samples, NA = Not Applicable

Wang et al. (2000; 2014b), use field measurements and a fate and transport model to estimate that docking Navy vessels in San Diego Bay resuspends approximately 26 tons of sediments per day. Wang et al. (2014a) measured sediment resuspension and associated metal contaminants from a tugboat propeller wash at Pier 4–5 in San Diego Bay; measurements included the concentrations of the metals chromium, copper, silver, cadmium, and nickel in the sediment plume. Only copper concentrations exceeded USEPA water quality criteria. However, all metal concentrations, with the exception of cadmium, were increased above ambient levels following resuspension events. Transport of sediments and dissolved or particle-bound metals as a results of propeller wash can potentially result in recontamination of remediated areas as well as increased contaminant concentrations in areas far from piers where docking occurs (Wang et al., 2014a).

C.1.3.1.3 Marine Debris in Nearshore and Offshore Areas off the Hawaiian Islands

A comprehensive review of anthropogenic marine debris, particularly plastics, and their worldwide distribution highlights the growing concern over global environmental impacts and the need for continued scientific research and improved waste disposal management practices (Bergmann et al., 2015). Marine debris in the North Pacific Ocean has been well documented in numerous publications since the early 1970s when Venrick et al. (1973) estimated that there were approximately 4.2 pieces of debris/square kilometers (km^2) , most of which were made from plastic, northeast of Hawaii in an area now known as the "North Pacific Garbage Patch" (Bergmann et al., 2015; Venrick et al., 1973). Nearly 40 years later, Titmus and Hyrenback (2011) recorded a density of 459 pieces/km^2 in the same region with over 95 percent of the debris composed of plastic. Analysis of 11 years of data from plankton net tows in the eastern North and South Pacific have allowed researchers to better define the scale of plastic distribution and density (Law et al., 2014). The accumulation of plastic and other debris is largely driven by surface ocean circulation patterns. Large-scale ocean surface currents driven by winds and geostrophic circulation converge in the subtropical North Pacific and result in an accumulation zone for plastic (i.e., the Garbage Patch). The accumulation zone occurs between latitude 25 to 41°N and longitude 130 to 180°W, which is north and primarily east of the Hawaiian Islands (19° 43' N, 155° 05' W). The median concentration of plastics within the accumulation zone was 33,090 pieces/km^2; outside of the zone the median concentration was 0 pieces/km^2. Plastic was collected on some tows outside of the zone. If considering only those tows and not the tows during which no plastic was collected, the median concentration outside of the zone was 1,485 pieces/km^2, approximately 22 times less than within the accumulation zone (Law et al., 2014). Nearly half of all net tows within the accumulation zone collected over 50,000 pieces/km^2, with the area of highest concentrations located between latitude 30 to 35°N and longitude 135 to 140°W, which is farther to the northeast from Hawaii.

Because of their buoyancy, many types of plastic float, and may be transported thousands of mi. in the ocean (U.S. Commission on Ocean Policy, 2004). Although plastics are highly resistant to degradation, plastics exposed to ultraviolet radiation from the sun gradually break down into smaller particles through a process called photo oxidation (Law et al., 2010). However, once plastic sinks below the photic zone, degradation rates become very slow, and once plastic reaches the seafloor degradation rates are further reduced. Microbial degradation of plastics in the marine environment does occur, but has a negligible impact on the amount of plastic that persists in the environment, because the process is slow and often occurs in low-oxygen environments on the seafloor (Andrady, 2015). Plastics can take hundreds of years to degrade and some plastics may never fully degrade and would persist in the environment indefinitely (Bergmuller et al., 2007).

The Hawaii Undersea Military Munitions Assessment documents various types of chemical, explosive, and non-explosive munitions and other military expended materials (MEM) located on the seafloor in a munitions disposal site south of Pearl Harbor, Oahu (Briggs et al., 2016; Koide et al., 2016).

In 2018 the Pacific Islands Fisheries Science Center evaluated the marine debris in the Northwestern Hawaiian Islands. In the greater Northern Pacific Grye it is estimated that approximately 52 tons of fishing gear is accumulated in the shallow waters. The greatest marine debris include derelict fishing gear, plastic, and other marine debris (Pacific Islands Fisheries Science Center, 2018).

C.1.3.1.4 Marine Debris in the Nearshore and Offshore Areas of the California Study Area

The Southern California 2018 Regional Monitoring Program conducted a comprehensive regional assessment of trash and marine debris in streams and nearshore waters of the Southern California bight (McLaughlin et al., 2022). The study found that 75 percent of stream kilometers and 30 percent of offshore areas has trash present during sampling. Between 1994 and 2018, marine debris is estimated to have increased from 4 percent to 17 percent.

The study evaluated macro-marine debris (particles 5 mm or less in diameter) imbedded in seafloor sediments. The study analyzed 138 benthic trawl samples and found that one-third of the seafloor in the SOCAL contained anthropogenic macro-debris with plastics being the most widespread type of debris. The most common debris consisted of wrappers, paper/cardboard, plastic, bags, and foam. Changes in the amount of marine debris did not vary significantly from the 2013 report (Moore et al., 2016). However, there was a decrease in plastic bags in the marine environment, likely due California's ban of single use plastic bags in 2016.

Watters et al. (2010) conducted a visual survey of the seafloor that included a portion of the Navy's Southern California (SOCAL) Range Complex as part of a 15-year quantitative assessment of marine debris on the seafloor off the California coast. Watters et al. (2010) found that plastics were the most abundant material and, along with recreational monofilament fishing line, dominated the debris encountered on the seafloor. The visual survey of the seafloor by Watters et al. (2010) encountered only a single object that was potentially "military" in origin (it appeared to be a shell casing). Navy vessels have a zero-plastic discharge policy and return all plastic waste to appropriate disposal or recycling sites on shore.

In a study of marine debris along the U.S. West Coast, Keller et al. (2010) characterized the composition and abundance of man-made marine debris at 1,347 randomly selected stations during Groundfish Bottom Trawl Surveys that took place in 2007 and 2008. The sample sites included some locations within the California Study Area. A subset of the sites sampled included historically used post-World War II dump sites. Recovered items identifying the sites as post-World War II-era dump sites included equipment described as "helmets," "gas masks," "uniforms," and other miscellaneous and diverse items such as "plastic," "file cabinets," and "buckets." Since approximately the 1970s, items such as these are no longer disposed of at sea. The items listed here are not military expended material and would not be expended during training and testing activities in the Hawaii-California Training and Testing (HCTT) Study Area. For this reason, the characterization of "military debris" in Keller et al. (2010) has little if any relevance to the Proposed Action or to present-day standard Navy conduct that (among other procedures) restricts the discharge of plastic at sea.

Overall, fourteen ocean disposal sites were utilized in Southern California waters between 1930 and 1970. Disposal sites are in deep water and includes waste from drilling and military activities as well as other chemicals and garbage. Since 2021, the USEPA and collaborating agencies are currently developing plans to evaluate threats to the environment and human health. Ocean Disposal Site #2 is located halfway between Palos Verdes Peninsula and Santa Catalina Island at a depth of 3,200 ft. USEPA identified that water activities near the surface would likely not be impacted by the disposal site (U.S. Environmental Protection Agency, 2024) .

C.1.3.1.5 Climate Change and Sediment Quality

Aspects of climate change that influence sediments include increasing ocean acidity (pH), increasing sea surface water temperatures, and increasing storm activity. Breitbarth et al. (2010) referred to seawater temperature and pH as “master variables for chemical and biological processes,” and noted that effects of changes on trace metal biogeochemistry “may be multifaceted and complex.” Under more acidic conditions, metals tend to dissociate from particles to which they are bound in sediments, become more soluble, and potentially more available.

The effects of climate change over the next century will impact water and sediment quality within coastal protected areas within the study area in a variety of ways. Most notably will be the effects of sea level rise and increased tidal surges on natural resources and shore infrastructure, and a diminution of freshwater inputs (U.S. Department of the Navy, 2013d). However, more frequent or intense storms due to climate change can have effects on nearshore water due to freshwater input (De Carlo et al., 2007). Marginal bay habitats without protective buffers are most at risk, especially those that require special salinity conditions, intermittent inundation, or light penetration. Changes in water temperature affect mud temperature (Stillman & Paganini, 2015 {Stillman, 2015 #13757) and influence nutrient processing.

As noted in the beginning of this section, tropical storms can have significant impacts on the resuspension and distribution of bottom sediment (Wren & Leonard, 2005). However, no consensus appears to exist on whether climate change will generate more tropical storms or whether those storms will be more intense. If storm frequency and intensity increase, the additional disturbance of sediment may impact water quality in nearshore and coastal areas.

C.1.3.2 Water Quality

C.1.3.2.1 Water Quality in the Nearshore and Offshore Waters of the Hawaiian Islands

Pearl Harbor is on Hawaii’s CWA Section 303(d) list of impaired waters. The Pearl Harbor Water Quality Limited Segment includes the entire harbor and the mouths of perennial streams discharging into the harbor. Beneficial uses of Pearl Harbor include bait fish and shellfish propagation in West and East Lochs, shipping navigation and industrial water in East Loch, and water fowl habitat in Middle and West Lochs (Hawaii State Department of Health, 2000).

Contaminants are introduced into Pearl Harbor via point source and non-point source discharges. Surface runoff from urban, industrial, and agricultural activities carries variable levels of herbicides, pesticides, and other contaminants, in addition to natural loads of sediment, dissolved metals, and other soluble constituents (Agency for Toxic Substances and Disease Registry, 2005). Water quality criteria that are frequently violated in Pearl Harbor include maximum nitrogen, phosphorous, fecal coliform, and chlorophyll-*a* concentrations, and turbidity and temperature limits (Hawaii State Department of Health, 2000). The Hawaii State Department of Health assessment of water quality in 160 marine water bodies included an evaluation of water quality in Pearl Harbor (State of Hawaii Department of Health, 2014). Waters in Pearl Harbor were in non-attainment for total phosphorous, total nitrogen, and chlorophyll-*a*, but were in attainment for turbidity. The presence of contaminants including PCBs, pesticides, and lead continue to restrict the consumption of fish and shellfish caught in Pearl Harbor. Table C-5 and Table C-6 provide the water quality criteria and an associated index for the U.S. West Coast and the Hawaiian Islands, respectively.

Table C-5: Water Quality Criteria and Index, United States West Coast

Criterion	Site Criteria			Regional Criteria		
	Good	**Fair**	**Poor**	**Good**	**Fair**	**Poor**
Dissolved Inorganic Nitrogen	< 0.5 mg/L	0.5–1.0 mg/L	> 1.0 mg/L	Less than 10% of the coastal area is in poor condition, and more than 50% of the coastal area is in good condition.	10–25% of the coastal area is in poor condition, or more than 50% of the coastal area is in combined poor and fair condition.	More than 25% of the coastal area is in poor condition.
Dissolved Inorganic Phosphorus	< 0.01 mg/L	0.01–0.1 mg/L	> 0.1 mg/L			
Water Clarity	Sites with naturally high turbidity: > 10% light at 1 meter Sites with normal turbidity: > 20% light at 1 meter Sites that support submerged aquatic vegetation: > 40% light at 1 meter	Sites with naturally high turbidity: 5–10% light at 1 meter Sites with normal turbidity: 10–20% light at 1 meter Sites that support submerged aquatic vegetation: 20–40% light at 1 meter	Sites with naturally high turbidity: < 5% light at 1 meter Sites with normal turbidity: < 10% light at 1 meter Sites that support submerged aquatic vegetation: < 20% light at 1 meter			
Dissolved Oxygen	> 5.0 mg/L	2.0–5.0 mg/L	< 2.0 mg/L	Less than 5% of the coastal area is in poor condition and more than 50% of the coastal area is in good condition.	5–15% of the coastal area is in poor condition, or more than 50% of the coastal area is in combined poor and fair condition.	More than 15% of the coastal area is in poor condition.
Chlorophyll-*a*	< 5 µg/L	5–20 µg/L	> 20 µg/L	Less than 10% of the coastal area is in poor condition, and more than 50% of the coastal area is in good condition.	10–20% of the coastal area is in poor condition, or more than 50% of the coastal area is in combined poor and fair condition.	More than 20% of the coastal area is in poor condition.
Water Quality Index	A maximum of one indicator is rated fair, and no indicators are rated poor.	One of the indicators is rated poor, or two or more indicators are rated fair.	Two or more of the five indicators are rated poor.			

Source: U.S. Environmental Protection Agency (2009)
Notes: < = less than, > = greater than, mg/L = milligram per liter, µg/L = microgram per liter

Table C-6: Water Quality Criteria and Index, Hawaiian Islands

Criterion	Site Criteria			Regional Criteria		
	Good	Fair	Poor	Good	Fair	Poor
Dissolved Inorganic Nitrogen	< 0.05 mg/L	0.05–0.1 mg/L	> 0.1 mg/L	Less than 10% of the coastal area is in poor condition, and more than 50% of the coastal area is in good condition.	10–25% of the coastal area is in poor condition, or more than 50% of the coastal area is in combined poor and fair condition.	More than 25% of the coastal area is in poor condition.
Dissolved Inorganic Phosphorus	< 0.005 mg/L	0.005–0.01 mg/L	> 0.01 mg/L			
Water Clarity	Sites with naturally high turbidity: > 10% light at 1 meter Sites with normal turbidity: > 20% light at 1 meter Sites that support submerged aquatic vegetation: > 40% light at 1 meter	Sites with naturally high turbidity: 5–10% light at 1 meter Sites with normal turbidity: 10–20% light at 1 meter Sites that support submerged aquatic vegetation: 20–40% light at 1 meter	Sites with naturally high turbidity: < 5% light at 1 meter Sites with normal turbidity: < 10% light at 1 meter Sites that support submerged aquatic vegetation: < 20% light at 1 meter			
Dissolved Oxygen	> 5.0 mg/L	2.0–5.0 mg/L	< 2.0 mg/L	Less than 5% of the coastal area is in poor condition and more than 50% of the coastal area is in good condition.	5%-15% of the coastal area is in poor condition, or more than 50% of the coastal area is in combined poor and fair condition.	More than 15% of the coastal area is in poor condition.
Chlorophyll-*a*	< 0.5 µg/L	0.5–1.0 µg/L	> 1.0 µg/L	Less than 10% of the coastal area is in poor condition, and more than 50% of the coastal area is in good condition.	10%-20% of the coastal area is in poor condition, or more than 50% of the coastal area is in combined poor and fair condition.	More than 20% of the coastal area is in poor condition.
Water Quality Index	A maximum of one indicator is rated fair, and no indicators are rated poor.	One of the indicators is rated poor, or two or more indicators are rated fair.	Two or more of the five indicators are rated poor.			

Source: U.S. Environmental Protection Agency (2009)

Notes: < = less than, > = greater than, mg/L= milligram per liter, µg/L = microgram per liter

C.1.3.2.2 Water Quality in the Nearshore and Offshore Waters of the California Study Area

Water quality in the nearshore waters of SCI, which are affected by baseline at-sea and ashore training and testing activities, has been tested (U.S. Department of the Navy, 2006a), and was reassessed for the 2010 SCI Range Condition Assessment (U.S. Department of the Navy, 2010b). Surface water and nearshore sediments just beyond the surf zone were sampled for metals, cyanide, chlorine, ammonia, phenols, pesticides, gross alpha/beta, perchlorate, and dioxin. Both acute and chronic toxicity tests were conducted with test organisms that included algae (kelp), an invertebrate (mussel), and a fish (topsmelt larvae). Two sample locations were in close proximity to the island's land-based shore bombardment area located on the southern end of the island. Samples from these two locations would be the most likely to show signs of munitions constituents entering the marine environment. Results indicated that most chemicals were not detected in receiving water and sediment samples, and in the cases where chemicals were detected and criteria were available, results fell well below all chemistry-related numerical objectives. The assessment concluded that only traces of the explosive compound HMX were detected in surface water—no other munitions constituents were detected, and HMX was detected only in a duplicate sample at five orders of magnitude below the threshold requiring that some level of action be taken. Therefore, although it appears that some munitions constituents may be migrating into the Pacific Ocean, they are doing so at concentrations well below levels of concern and well below concentrations that would be detectable in ocean water. Furthermore, no statistically significant toxicity was observed in topsmelt, giant kelp, or bivalve bioassays. These data suggest that Navy discharges from SCI do not compromise protection of ocean waters for beneficial uses around SCI.

Based on *California Ocean Plan* objectives for protection of aquatic life, concentrations of potential water pollutants are low, and have no substantial effects on marine water quality in a portion of the SOCAL Study Area where training and testing activities are most concentrated (U.S. Department of the Navy, 2013a).

Major contaminants found in San Diego Bay include chlorinated hydrocarbons, PCBs, toxic components of petroleum hydrocarbons, polycyclic aromatic hydrocarbons, heavy metals, and organotins such as tributyltin (U.S. Department of the Navy, 2013d). The sources of these compounds include effluents from non-point-source storm drain runoff (municipal and industrial); contaminants from vessel maintenance; antifouling paints (military, commercial, and private vessels); marina discharges; and residues of prior industrial discharges. These contaminants have generally been incorporated into bottom sediments in the bay, and are periodically re-suspended in the water column when bottom sediments are disturbed by natural or human activities.

Water quality in north and central San Diego Bay is affected primarily by tidal flushing and currents. Water quality also is influenced locally by freshwater inflows. The watershed that contributes to San Diego Bay has a number of Total Maximum Daily Loads established for Chollas Creek and Shelter Island for criteria such as diazinon, dissolved metals, and indicator bacteria (U.S. Department of the Navy, 2013d). Gross water quality characteristics (e.g., salinity, temperature, and DO) form a gradient within San Diego Bay. Waters in northern San Diego Bay are similar to ocean conditions; waters in southern San Diego Bay are strongly affected by shallow depths, fresh water inflows, and solar insolation; waters in central San Diego Bay are intermediate in character.

Beach closures for Silver Strand Shoreline and Coronado are common from frequent water quality monitoring and high levels of harmful bacteria. Since December 8, 2021, the Tijuana Slough Shoreline has been closed due bacterial levels that exceed health standards.

C.1.3.2.3 Marine Debris and Water Quality

The National Marine Debris Monitoring Program developed three categories of marine debris for its study of the extent of man-made materials in the oceans: land-based, ocean-based, and general (i.e., origin unspecified) (Sheavly, 2007). Land-based debris may blow in on the wind, be washed in with storm water, arise from recreational use of coastal areas, and be generated by extreme weather such as tsunamis. Ocean-based sources of marine debris include commercial shipping and fishing, private boating, offshore mining and extraction, and legal and illegal dumping at sea. Ocean current patterns, weather and tides, and proximity to urban centers, industrial and recreational areas, shipping lanes, and fishing grounds influence the types and amount of debris found (U.S. Environmental Protection Agency, 2010). These materials are concentrated at the near-surface and in the water column.

According to the U.S. Environmental Protection Agency (2010), land-based sources account for about half of marine debris, and ocean/waterway-based sources contribute another 18 percent. Bergmuller et al. (2007) confirm that the majority of marine debris originates from land. Land-based debris included items like syringes, condoms, metal beverage cans, motor oil containers, balloons, six-pack rings, straws, tampon applicators, and cotton swabs. Ocean-based debris included gloves, plastic sheets, light bulbs and tubes, oil and gas containers, pipe-thread protectors, nets, traps and pots, fishing line, light sticks, rope, salt bags, fish baskets, cruise line logo items, and floats and buoys. Plastics, generally referring to petroleum-based, manmade materials, make up the vast majority of marine debris (Bergmuller et al., 2007; Law & Thompson, 2014).

Within the HCTT Study Area, Currie et al. (2017b) conducted surveys for marine mammals and floating marine debris in the waters around the island of Lanai and waters between Lanai and the islands of Maui and Kahoolawe from April 2013 to April 2016. The survey encountered, collected, and categorized 1,027 pieces of marine debris. Items categorized as “plastic” were the predominant type of debris encountered and accounted for 86 percent of total debris. Plastics consisted mainly of plastic bottles, tubs, baskets, foamed polystyrene disposable plates, cups, fragments, plastic bags, and other soft plastic films. A smaller portion of the plastic (13 percent; 11 percent of the total amount of material) were fishing-related and included items such as buoys, netting, rope, and fishing lines. Milled lumber and rubber accounted for 10 percent of total debris, and the remaining 4 percent consisted of metal, glass, and clothing/fabric.

Microscopic plastic fragments enter the marine environment from use as scrubbers in hand cleaning and other cosmetic products, abrasive beads for cleaning ships, and deterioration of macroscopic plastics (Teuten et al., 2007). Microplastic beads commonly used in cosmetic products such as facial scrubs and other exfoliants are not broken down in wastewater treatment facilities and are largely not filtered out of the waste stream before they are flushed into the marine environment (Chang, 2015; Napper et al., 2015). These microbeads are found worldwide in marine sediments, persist in the marine environment, and accumulate up the food chain (Cole & Galloway, 2015). On December 18, 2015, the *Microbead-Free Waters Act of 2015* (21 U.S.C 301) was passed prohibiting the manufacturing, packing, and distribution of rinse-off cosmetics containing plastic microbeads.

Plastics may serve as vehicles for transport of various pollutants, whether by binding them from seawater or from the constituents of the plastics themselves. Mato et al. (2001) noted that polypropylene resin pellets (precursors to certain manufactured plastics) collected from sites in Japan contained PCBs, dichlorodiphenyldichloroethylene (a breakdown product of DDT), and the persistent organic pollutant nonylphenol (a precursor to certain detergents). PCBs and DDT were adsorbed from seawater and accumulated on the surface of plastics. The original source of nonylphenol was less clear;

it may have originated from the pellets themselves or may have been adsorbed from the seawater. Microbeads have also been shown to adsorb hydrophobic chemical contaminants, such as DDT, from seawater, allowing for the accumulation and transport of these often toxic chemicals to widely dispersed areas of the oceans. While the impacts on the marine ecosystem are largely unknown, some examples illustrating potential widespread impacts have been discussed. For example, it has been suggested that white and blue microplastic beads, common in many exfoliants, resemble plankton and may be mistakenly ingested by plankton-feeding fishes, which rely on visual cues to find prey (Napper et al., 2015; Wright et al., 2013). The long-term effects on the environment from the proliferation of microbeads and other micro plastics are still being researched. Since there is no way of effectively removing micro plastics from the marine environment, and given that plastics are highly resistant to degradation, it is likely that the quantity of micro plastics in the marine environment will only continue to increase, and therefore the likelihood of environmental impacts can only increase (Napper et al., 2015). The only way to reduce long-term impacts is to reduce or eliminate the use of micro plastics, a course of action that is gaining recognition (Chang, 2015).

Because of their buoyancy, many types of plastic items float and may travel thousands of miles in the ocean (U.S. Commission on Ocean Policy, 2004). Exceptions include heavy nets and ropes. Although plastics are resistant to degradation, they do gradually break down into smaller particles due to sunlight and mechanical wear (Law et al., 2010). A study by Teuten et al. (2007) indicated that the water-borne phenanthrene (a type of polycyclic aromatic hydrocarbon) adhered preferentially to small pieces of plastic ingested by a bottom-dwelling marine lugworm and incorporated into its tissue. Marine microbes and fungi are known to degrade biologically produced polyesters, such as polyhydroxyalkanoates, a bacterial carbon and energy source (Doi et al., 1992). Marine microbes also degrade other synthetic polymers, although at slower rates (Shah et al., 2008).

Annex V of the International Convention for the Prevention of Pollution from Ships prohibits the discharge of plastic waste from vessels at sea, and the U.S. Act to Prevent Pollution from Ships brought U.S. public vessels in alignment with the international convention. The NDAA of 1996 specifically directed the Navy to install plastic waste processors aboard the surface fleet. The Navy's plastics waste processors compress and melt shipboard-generated plastic waste into dense, sanitary disks of compressed plastics that can be stored over long at-sea deployments. The plastic wastes items include lightly contaminated food containers as well as clean plastics and other materials that may be combined with, or contain, plastic components that cannot be processed in the normal solid waste stream. The plastic waste disks are offloaded for proper disposal once a ship comes into port. The plastic compression technology enables Navy ships to operate at sea over long time periods without discharging plastics into the oceans.

C.1.3.2.4 Climate Change and Water Quality

According to the U.S. Global Change Research Program, the rise in ocean temperature over the last century will continue into the reasonably foreseeable future, with continued and perhaps increasing impacts on ocean circulation, marine chemistry, and marine ecosystems. Because the ocean currently absorbs about a quarter of human-produced CO_2 emissions, increasing CO_2 absorption will increase acidification of ocean waters. This in turn will alter the distribution, abundance, and productivity of many marine species (Melillo et al., 2014).

Key findings of the 2014 National Climate Assessment that may pertain to waters surrounding the Hawaiian Islands include:

- Warmer oceans are leading to increased coral bleaching events and disease outbreaks in coral reefs, as well as changed distribution patterns of tuna fisheries. Ocean acidification will reduce coral growth and health. Warming and acidification, combined with existing stresses, will strongly affect coral reef fish communities.
- Saltwater intrusion associated with sea level rise will reduce the quantity and quality of freshwater in coastal aquifers, especially on low islands.
- Rising sea levels, coupled with high water levels caused by storms, will incrementally increase coastal flooding and erosion, damaging coastal ecosystems, infrastructure, and agriculture, and indirectly affecting tourism.

Key findings of the 2014 National Climate Assessment that may pertain to waters off California include:

- With the decreases in snowpack and streamflow expected to continually decline, freshwater inputs into California's coastal estuaries will decrease, with subsequent losses of ecosystem services that estuaries provide (e.g., nutrient cycling, filtration).
- Sea level rise is projected to increase, resulting in major damage as wind-driven waves ride upon higher seas and reach farther inland.

The Paris Agreement builds upon the Convention and – for the first time – brings all nations into a common cause to undertake ambitious efforts to combat climate change and adapt to its effects, with enhanced support to assist developing countries to do so. As such, it charts a new course in the global climate effort.

At the 2015 Paris Climate Conference, 195 parties to the United Nations Framework Convention on Climate Change adopted the first-ever universal, global climate agreement, referred to as the Paris Agreement in which all countries voluntarily set and committed to individual carbon reduction goals. The Agreement marks the latest step in the evolution of the United Nations climate change initiative and builds on the work undertaken under the Convention over the past several decades.

The Paris Agreement seeks to accelerate and intensify the actions and investment needed for sustaining low carbon emissions into the future. Its central aim is to strengthen the global response to the threat of climate change and greenhouse gas emissions by limiting a global temperature rise over this century to no more than 2 degrees Celsius (°C)above pre-industrial levels. The Paris Agreement also includes a commitment to pursue efforts to limit the temperature increase even further to 1.5 °C.

The U.S. signed the Paris Agreement on April 22, 2016, and on September 3, 2016, the U.S. accepted ratification of the Agreement. The U.S. withdrew from the Paris Agreement in 2020 but reentered under the Biden Administration in January of 2021.

C.2 Vegetation

C.2.1 General Background

C.2.1.1 Taxonomic Overview

Vegetation includes diverse taxonomic/ecological groups of marine algae throughout the Study Area, as well as flowering plants in the coastal and inland waters. Vegetation can be divided into eight groups that encompass taxonomic categories, distributions, and ecological relationships. These groups include blue-green algae (phylum Cyanobacteria), dinoflagellates (phylum Dinophyta), green algae (phylum Chlorophyta), coccolithophores (phylum Haptophyta), diatoms (phylum Ochrephyta), brown algae (phylum Phaeophyta), red algae (phylum Rhodophyta), and vascular plants (phylums Tracheophyta and Spermatophyte). Furthermore, the analysis considers the distribution of vegetation based on oceanic features and vertical distribution. Open-ocean oceanographic features of the Study Area include the North Pacific Subtropical Gyre and the North Pacific Transition Zone. Additionally, vertical distribution within the water column or the bottom substrate is considered.

C.2.1.2 General Threats

Human-made stressors that act on marine vegetation include excessive nutrient input (such as fertilizers), siltation (the addition of fine particles to the ocean), pollution (oil, sewage, trash) (Mearns et al., 2011), climate change (Arnold et al., 2012; Doney et al., 2012; Martinez et al., 2012; Olsen et al., 2012), fishing practices (Mitsch et al., 2009b; Steneck et al., 2002), shading from structures (National Marine Fisheries Service, 2002), harvesting (Wilson, 2002), habitat degradation from construction and dredging, and introduced or invasive species (Hemminga & Duarte, 2000; Spalding et al., 2003). The seagrass, cordgrass, and mangrove taxonomic group is often more sensitive to stressors than the algal taxonomic groups. The great diversity of algae makes generalization difficult, but overall, algae are resilient and colonize disturbed environments created by stressors (Levinton, 2009b).

Marine algae and vascular plants are important ecologically and economically, providing an important source of food, essential ecosystem services (e.g., coastal protection, nutrient recycling, food for other animals, and habitat formation), and income from tourism and commercial fisheries (Spalding et al., 2001).

C.2.1.2.1 Development and Human Activities

Green seaweed is harvested for human consumption in Hawaii’s coastal waters. Common species harvested include *Ulva fasciata, Enteromorpha prolifera,* and *Codium edule* (Preskitt, 2002a, 2010). Edible brown seaweeds that are collected in Hawaii include *Sargassum echinocarpum* and *Dictyopteris plagiograma* (Preskitt, 2002b). The State of Hawaii Department of Land and Natural Resources regulates the collection of seaweeds.

Although historically important, large-scale harvesting of kelp beds no longer occurs along the California coast. Small-scale commercial operations, however, continue to harvest kelp, primarily for abalone feed (Wilson, 2002). The California Department of Fish and Game, which issues exclusive leases to harvest designated beds for up to 20 years, manages kelp harvesting. Although they are not limited in the amount, California regulations prohibit commercial harvesters from cutting attached *Macrocystis pyrifera* and *Nereocystis luetkeana* (giant and bull) kelp from deeper than 4 ft. (1.2 m) below the water’s surface (14 California Code of Regulations 165[c][2]), which protects the reproductive structures at the kelp’s base and allows vegetative re-growth (Wilson, 2002).

C.2.1.2.2 Water Quality Degradation

Water quality in the Study Area may be impacted by the introduction of harmful contaminants from diverse sources unrelated to either action alternative. Common ocean pollutants include toxic compounds such as metals, pesticides, herbicides, and other organic chemicals, excess nutrients from fertilizers and sewage, detergents, oil, plastics, and other solids. Coastal pollution and agricultural runoff may cause toxic red tide events in the Study Area (Hayes et al., 2007). Coastal development and pollution, particularly storm water runoff and point source discharges, affect water quality of bays and coastal areas throughout the world. Depending on the proximity to and nature of the discharge, sediment and water quality may be degraded, which in turn can impact marine vegetation communities.

Erosion and sedimentation may also affect sediment and water quality of coastal areas during storm runoff from urban streets into rivers and streams.

Oil in runoff from land-based sources, natural seeps, and accidental spills (such as offshore drilling and oil tanker leaks) are some of the major sources of oil pollution in the marine environment (Levinton, 2009b). The type and amount of oil spilled, weather conditions, season, location, oceanographic conditions, and the method used to remove the oil (containment or chemical dispersants) are some of the factors that determine the severity of the impacts. Sensitivity to oil varies among species and within species (Hayes et al., 1992). The tolerance to oil pollutants varies among the types of marine vegetation, but their exposure to sources of oil pollutants makes them all vulnerable.

Oil pollution, as well as chemical dispersants used in response to oil spills, can impact seagrasses directly by smothering the individuals, or indirectly by lowering their ability to combat disease and other stressors (U.S. National Response Team, 2010). Seagrasses that are totally submerged are less susceptible to oil spills since they largely escape direct contact with the pollutant. Depending on various factors, oil spills can result in a range of effects from no impact to long-lasting impacts, such as decreases in eelgrass density (Kenworthy et al., 1993; Peterson, 2001). Algae are relatively resilient to oil spills, while mangroves are highly sensitive to oil exposure. Contact with oil can cause death, leaf loss, and failure to germinate (Hoff et al., 2002). Salt marshes can also be severely impacted by oil spills, with long-term effects (Culbertson et al., 2008).

C.2.1.2.3 Climate Change

The impacts of anthropogenically induced climate change on the marine environments include rising sea levels, ocean acidification, increased sea temperature, and an increase in severe weather events. All of these changes may have impacts on vegetation in the Study Area.

Rising sea levels will alter the amount of sunlight reaching various areas, which may decrease the photosynthetic capabilities of vegetation in those areas. However, the fast growth and resilient nature of vegetation may enable most species to adapt to these changes (Harley et al., 2006). Increased sea temperature may lead to several impacts that could affect vegetation. Warmer waters may lead to a greater stratification in the water column, which may support harmful algal blooms (Lehmköster, 2015). The stratification may also inhibit upwelling, as seen during El Niño events, which would prevent nutrients from circulating to the surface (Lehmköster, 2015). Additionally, increased sea temperatures may lead to changes in the composition of vegetation communities (Schiel et al., 2004). These changes in community composition could impact biological interactions, including the mutualism between reef-building corals and algae (Doney et al., 2012). These indirect and direct impacts of climate change that decrease coral reef habitat may enable vegetation to overtake areas that were previously biogenic reef habitat (Hughes et al., 2007; Pandolfi et al., 2005). Increases in severe weather events may lead to increased erosion and sedimentation in the marine environments and higher energy wave action that could increase impacts on vegetation by physical disturbance, such as marine vascular plants becoming unrooted.

Vegetation is susceptible to water quality changes from erosion and disturbances from storm events. Increased storm events are expected to impact species diversity in kelp ecosystems (Byrnes et al., 2011). The impacts of ocean acidification on vegetation are poorly understood (Harley et al., 2006). Ocean acidification may impact the ecological function of coralline algae by decreasing habitat-forming capabilities (Ragazzola et al., 2016).

C.2.1.2.4 Diseases and Parasites

Marine algae and vascular plants may be susceptible to disease caused by other marine organisms, which may impact individuals or populations. In particular, eelgrass is vulnerable to a wasting disease caused by a marine pathogen that has caused devastating population loss in the past (Ralph & Short, 2002). Certain species of microscopic algae (e.g., dinoflagellates and diatoms) can form algal blooms, which can pose serious threats to human health and wildlife species. Harmful algal blooms can deplete oxygen within the water column and block sunlight that other organisms need to live, and some algae within algal blooms release toxins that are dangerous to human and ecological health (Center for Disease Control and Prevention, 2004). These algal blooms have a negative economic impact of hundreds of millions of dollars annually worldwide (National Centers for Coastal Ocean Science, 2010). Additional information on harmful algal blooms can be accessed on the Centers for Disease Control and the National Oceanic and Atmospheric Administration (NOAA) websites.

C.2.1.2.5 Invasive Species

Invasive vegetation species are present throughout the Study Area. The red mangrove (*Rhizophora mangle*) is an invasive species in Hawaii and various resource agencies and organizations (e.g., Hawaii Department of Land and Natural Resources, Pacific Cooperative Studies Unit, Malama O Puna) have eradication programs targeting the red mangrove and other mangrove infestations (Figure C-2). First introduced primarily to stabilize coastal flats in the early 1900s (Allen, 1998), the red mangrove is native to Florida and the Caribbean. Since the introduction of this species, mangroves have invaded intertidal areas formerly devoid of trees. In 2013 and 2014, the Navy completed several mangrove removal actions in Pearl Harbor (Figure C-2), which enhanced native sedge growth among other environmental benefits (U.S. Department of the Navy, 2014).

Invasive marine green algal species are found in coastal waters of the Study Area. The invasive green algae, *Avrainvillea amadelpha,* has been recorded in the main Hawaiian Islands (Preskitt, 2010). Invasive green algae represent a serious threat to coral reefs, and may displace, outcompete, or hybridize with non-invasive native green algae species, resulting in the loss of native biodiversity or alteration of ecosystem processes. Representative non-native invasive species of red algae in the Hawaii Study Area include *Acanthophora spicifera*, *Gracilaria salicornia*, *Hypnea musciformis*, *Kappaphycus alvarezii*, and *Gracilaria tikvahiae* (Smith et al., 2002).

Caulerpa taxifolia and *Codium fragile tomentosoide* are invasive green algal species found in the California Study Area (Dobroski et al., 2015; Gagnon et al., 2015). In addition, *Sargassum muticum* (Japanese wireweed) and *Sargassum horneri* (devil weed) are invasive brown algal species found within the California Study Area (Dobroski et al., 2015; Marks et al., 2015). *Undaria pinnatifida* (or wakame), which is an edible seaweed native to Japan, is an invasive species that is also found along the California coast (Dobroski et al., 2015; Global Invasive Species Database, 2005). Devil weed and wakame are found in San Diego County and have exhibited characteristics of successful invaders such as establishing in new areas, spreading locally, and persisting through multiple generations. They primarily occur in harbors but have also been found in open coast sites. This rapid and uncontrolled spread has ecological and economic consequences that will require further research (Kaplanis et al., 2016).

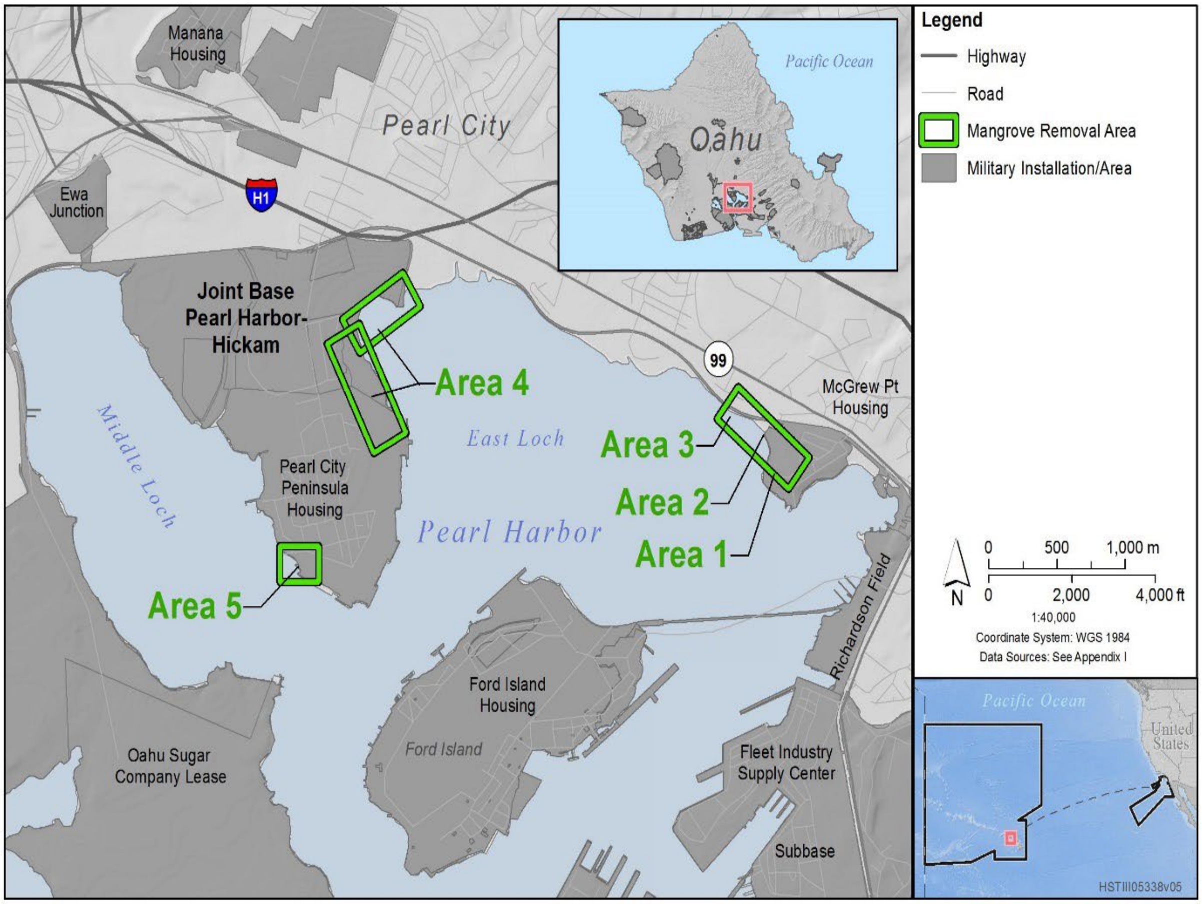

Figure C-2: Areas Subject to Mangrove Removal in Pearl Harbor

The DoD has implemented projects to control invasive microalgae at critical control points (specific areas where spread and transport of invasive species are likely to occur). For example, in 2011, an experimental macroalgae cleanup occurred in an infested area of Mokapu Peninsula, at the sea plane ramps. Lessons learned from this experiment were discussed with Sikes Act partners and provided the basis for tackling more ambitious projects in the future. A slow and steady phased approach is often the most successful in making progress with controlling invasive species, based on the experiences of the Marine Corps Base at Mokapu Peninsula (Marine Corps Base Hawaii, 2011).

C.2.1.2.6 Marine Debris

Marine debris (especially plastics) is a threat to many marine ecosystems, particularly in coastal waters adjacent to urban development. Microplastics (generally considered to be particles less than 5 mm in size), which may consist of degraded fragments of larger plastic items or intentionally manufactured items (e.g., microbeads), are of concern because of their durability, long lifespan, and potential to enter marine food webs (Setala et al., 2016). Marine debris may injure marine vegetation if it is large and is pulled around by tidal influences and currents (Gregory, 2009). Refer to Section C.1.3.1.3 (Marine Debris in the Nearshore and Offshore Areas off the Hawaiian Islands) and Section C.1.3.1.4 (Marine Debris in the Nearshore and Offshore Areas of the California Study Area) for a more detailed discussion of marine debris and the associated effects on water quality.

C.2.2 Species Not Listed under the Endangered Species Act

For the following discussion, vegetation has been divided into eight major phyla that have distinct morphological, biochemical, physiological, and life history traits that reflect their evolutionary history and influence their distributions and ecological relationships.

C.2.2.1 Blue-Green Algae (Phylum Cyanobacteria)

Blue-green algae are single-celled, photosynthetic bacteria that inhabit the photic zone and seafloors of the world’s oceans (Roskov et al., 2015). Blue-green algae are key primary producers in the marine environment and provide valuable ecosystem services such as producing oxygen and nitrogen. The blue-green algae, *Prochlorococcus* species, is responsible for a large portion of the global oxygen production by photosynthetic organisms. Other species of blue-green algae have specialized cells that convert nitrogen gas into a form that can be used by other marine plants and animals (nitrogen fixation) (Hayes et al., 2007). In the nutrient-poor waters of coral reef ecosystems within the Hawaii Study Area, blue-green algae are an important source of food for marine species. Diverse grazers, particularly large grazers such as sea urchin and fish, as well as mesoherbivores (e.g., small fish and crabs) and microherbivores (e.g., amphipods, gastropods, and polychaetes) are known to feed on blue-green algae and may influence algal community structures. Physical and biological disturbances to algae may, ultimately, shift the algal community structure to more disturbance-tolerant forms of algae (e.g., turfs and crusts) (Cheroske et al., 2000).

C.2.2.2 Dinoflagellates (Phylum Dinophyta)

Dinoflagellates are single-celled organisms with two flagella (whip-like structures used for locomotion) in the phylum Dinophyta (Roskov et al., 2015). Dinoflagellates are predominantly marine algae, with an estimated 1,200 species living in surface waters of the ocean worldwide (Castro & Huber, 2007). Most dinoflagellates can use the sun’s energy to produce food through photosynthesis and can ingest small food particles. Photosynthetic dinoflagellates are important primary producers in coastal waters (Waggoner & Speer, 1998). Organisms such as zooplankton (microscopic animals that drift passively in the water column) feed on dinoflagellates.

Dinoflagellates are also valuable for their close relationship with some invertebrates, most notably reef-building corals. Some species of dinoflagellates (zooxanthellae) live inside corals. This mutually beneficial relationship provides shelter and food (in the form of coral waste products) for the dinoflagellates; in turn, the corals receive essential nutrients produced by dinoflagellates (Spalding et al., 2007). Dinoflagellates cause some types of harmful algal blooms, which result from sudden increases in nutrients (e.g., fertilizers) from land into the ocean or changes in temperature and sunlight (Levinton, 2009b).

C.2.2.3 Green Algae (Phylum Chlorophyta)

Green algae are single-celled organisms in the phylum Chlorophyta that may form large colonies of individual cells (Roskov et al., 2015). Green algae may be found in the water column and benthic habitats. Only 10 percent of the estimated 7,000 species of green algae are found living in the marine environment (Castro & Huber, 2000c). These species are important primary producers that play a key role at the base of the marine food web. Green algae are found in areas with a wide range of salinity, such as bays and estuaries, and are eaten by various organisms, including zooplankton and snails.

C.2.2.4 Coccolithophores (Phylum Haptophyta)

Coccolithophores are single-celled phytoplankton that are especially abundant in tropical oceans but also bloom seasonally at higher latitudes. Up to 200 species have been described in the scientific record, 30–40 of which are common in the sedimentary record (Giraudeau & Beaufort, 2007). Coccolithophores are found in the water column as free-floating phytoplankton. They are nearly spherical and covered with plates made of calcite (calcium carbonate), which account for approximately one-third of calcium carbonate production in the entire ocean. They are an often-abundant component of the phytoplankton and account for a large fraction of primary production and carbon sequestration in the ocean. Blooms produce a strong bluish-white reflection that may cover thousands of square miles (Levinton, 2013).

C.2.2.5 Diatoms (Phylum Ochrophyta)

Diatoms are single-celled organisms with cell walls made of silicon dioxide. Two major groups of diatoms are generally recognized, centric diatoms and pinnate diatoms. Centric diatoms exhibit radial symmetry (symmetry about a point), while the pinnate diatoms are bilaterally symmetrical (symmetry about a line). Diatoms are found in the water column and benthic habitats in coastal areas. Diatoms such as *Coscinodiscus* species commonly occur throughout the Study Area. Some strains of another genus of diatoms, Pseudonitzschia, produce a toxic compound called domoic acid. Humans, marine mammals, and seabirds become sick or die when they eat organisms that feed on Pseudonitzschia strains that produce the toxic compound. The California Study Area, off the coasts of Los Angeles and Orange Counties, had some of the highest concentrations of the toxic compound ever recorded in U.S. waters (Schnetzer et al., 2007). *Pseudo-nitzschia* blooms in the Southern California Bight during 2003 and 2004 were linked to stranding over 1,400 marine mammals (Schnetzer et al., 2007). Pollutants carried from land to the ocean by rainwater (Kudela & Cochlan, 2000), and decreases in the movement of cool, nutrient-rich waters by the wind are believed to be the main causes of these harmful algal blooms in the California Study Area (Kudela et al., 2004).

C.2.2.6 Brown Algae (Phylum Phaeophyta)

Brown and golden-brown algae are large multi-celled marine species with structures varying from fine filaments to thick leathery forms (Castro & Huber, 2000c). Most species are attached to the seafloor in coastal waters (such as kelp), although a species with both attached and free-floating forms (*Sargassum muticum* [invasive]) occurs within the California Study Area.

C.2.2.6.1 Kelp

Kelp is a general term that refers to brown algae of the order Laminariales. Kelp plants are made of three parts: the leaf-like blade(s), the stipe (a stem-like structure), and the holdfast (a root-like structure that anchors the plant to the bottom). The following five species of canopy-forming kelp occur in the coastal waters of the California coast: giant kelp, bull kelp, elk horn kelp (*Pelagophycus porra*), feather boa kelp (*Egregia menziesii*), and chain bladder kelp (*Stephanocystis osmundacea*). The dominant kelp in the California Study Area is giant kelp. Since the first statewide survey in 1967, the total area of kelp canopies has generally declined; the greatest decline occurred along the mainland coast of southern California (Wilson, 2002; Young et al., 2016b). The canopy coverage of kelp beds varies under changing oceanographic conditions, and is also influenced by the level of harvesting, invasive species, coastal pollution, and development (Wilson, 2002).

Kelp is the most conspicuous brown algae occurring extensively along the coast in the California Study Area. The giant kelp can live up to eight years and can reach lengths of 197 ft. (60 m). The leaf-like fronds can grow up to 23.6 inches (in.) (60 centimeters [cm]) per day (Leet et al., 2001). Bull kelp

(*Nereocystis luetkeana*) growth can exceed 3.9 in. (10 cm) per day. Bull kelp attaches to rocky substrates and can grow up to 164 ft. (50 m) in length in nearshore areas. In turbid waters, the offshore edge of kelp beds occurs at depths of 49–59 ft. (15–18 m), which can extend to a depth of 98.4 ft. (30 m) in the clear waters around the Channel Islands off the coast of southern California (Wilson, 2002). The kelp beds along the California coast and off the Channel Islands are the most extensive and elaborate submarine forests in the world (Rodriguez et al., 2001). El Niño events tend to have a direct influence on the region and have the potential to affect kelp populations, especially when these events are major (Grove et al., 2002).

C.2.2.6.2 Sargassum

Sargassum is a genus of brown algae that generally inhabits shallow waters and coral reefs within the Study Area. *Sargassum echinocarpum* (Limu kala) is a native species of Hawaii and is usually found within tide pools and on reef flats. Meanwhile, *Sargassum agardhianum* is native to California.

Two introduced species of *Sargassum* also inhabit the California Study Area—*Sargassum muticum* and *Sargassum horneri*. The brown alga *Sargassum muticum* was introduced from the Sea of Japan and now occupies portions of the California coast (Dobroski et al., 2015; Monterey Bay Aquarium Research Institute, 2009). *Sargassum horneri* is native to western Japan and Korea. Since *Sargassum horneri* was first discovered in Long Beach Harbor in 2003, the species continues to increase its spatial extent and can now be found near harbors and anchorages from Santa Barbara, California, to Isla Natividad in Baja California (Mexico) (Marks et al., 2015). Specifically, *Sargassum horneri* has been found in the Study Area, in places like San Diego and the Channel Islands (U.S. Department of the Navy, 2013d). Both *Sargassum muticum* and *Sargassum horneri* are present in the Study Area.

C.2.2.7 Red Algae (Phylum Rhodophyta)

Red algae are predominately marine, with approximately 4,000 species worldwide (Castro & Huber, 2000c). Red algal species exist in a range of forms, including single and multicellular forms (Roskov et al., 2015), from fine filaments to thick calcium carbonate crusts. Within the Study Area, they occur in the water column and bottom habitats of coastal waters, primarily in reef environments and intertidal zones of Hawaii and California. Common native species in Hawaii include *Laurencia* species, *Gracilaria coronopifolia*, *Hypnea cervicornis*, and *Gracilaria parvispora*. Many red algae species contribute to reef formation by hardening the reef (by producing calcium carbonate) and by cementing coral fragments (Veron, 2000), and are food for various sea urchins, fishes, and chitons. In California waters, common species include *Endocladia muricata*, *Mastocarpus papillatus*, and species of *Mazaella*.

C.2.2.8 Seagrasses, Cordgrasses, and Mangroves (Phylum Spermatophyta)

Seagrasses, cordgrasses, and mangroves are flowering marine plants in the phylum Spermatophyta (Roskov et al., 2015). These marine flowering plants create important habitat and are a food source for many marine species. These marine vascular plants are found only in coastal waters, attached to the bottom.

C.2.2.8.1 Seagrasses

Seagrasses are unique among flowering plants because they grow submerged in shallow marine environments. Except for some species that inhabit the rocky intertidal zone, seagrasses grow in shallow, subtidal, or intertidal sediments, and can extend over a large area to form seagrass beds (Garrison, 2004; Phillips & Meñez, 1988). Seagrass beds provide important ecosystem services as a structure-forming keystone species (Arnold et al., 2012; Buhl-Mortensen et al., 2010; U.S. National

Response Team, 2010). They provide suitable nursery environment for commercially important organisms (e.g., crustaceans, fish, and shellfish) and are also a food source for numerous species (e.g., turtles) (Nagaoka et al., 2012). Seagrass beds combat coastal erosion, promote nutrient cycling through the breakdown of detritus (Dawes, 1998; Dawes et al., 1997), and improve water quality. Seagrasses also contribute a high level of primary production to the marine environment, which supports high species diversity and biomass (Spalding et al., 2003). Seagrasses are uprooted by dredging and scarred by boat propellers (Hemminga & Duarte, 2000; Spalding et al., 2003), which can take years to recover. In Hawaii, the most common seagrasses are Hawaiian seagrass (*Halophila hawaiiana*) and paddle grass (*Halophila decipiens*). Hawaiian seagrass is a native species found at less than 3.3 ft. (1 m) in subtidal, sandy areas surrounding reefs, in bays, or in fishponds. It occurs in coastal waters of Oahu near Mamala Bay (southern coast), in Maunalua Bay (southeastern coast), in Kaneohe Bay (northeast coast), in coastal waters of Maui, in the inner reef flats of southern Molokai, at Anini Beach on the northern shore of Kauai, and at Midway Atoll in the Northwestern Hawaiian Islands (Phillips & Meñez, 1988). Paddle grass is possibly a nonnative species that occurs only on Oahu in waters to 114.8 ft. (35 m) deep; it is apparently restricted to the southern shore of Oahu (Preskitt, 2001, 2002b).

Seagrasses that occur in the coastal areas of the California Study Area in the California Current Large Marine Ecosystem include eelgrass (*Zostera marina* and *Zostera pacifica*), surfgrass (*Phyllospadix scouleri* and *Phyllospadix torreyi*), widgeon grass (*Ruppia maritima*), and shoal grass (*Halodule wrightii*) (Jones et al., 2013; Spalding et al., 2003). The distribution of underwater vegetation is patchy along the California coast. In the California Study Area, eelgrass and surfgrass are the dominant native seagrasses (Wyllie-Echeverria & Ackerman, 2003).

Eelgrass covers most of the available nearshore area in San Diego Bay (Figure C-3). Beds of eelgrass (*Zostera marina*) form an important and productive benthic habitat in San Diego Bay. Eelgrass habitats rank among the most productive habitats in the ocean (Nybakken, 1993) and are an important component of the San Diego Bay food web. As has occurred in bays and estuaries all along the Pacific coast and elsewhere in the world, eelgrass beds have suffered substantial losses and impacts due to their location in sheltered waters where human activity is concentrated. However, these losses were historic due to bay fill and deepening.

Today, various state and federal regulatory frameworks protect eelgrass beds, and any impacts are fully mitigated. For example, National Marine Fisheries Service (NMFS) policy recommends no net loss of eelgrass habitat function in California and encourages the use of eelgrass mitigation banking and in-lieu fee programs when impacts on eelgrass habitat cannot be avoided (National Marine Fisheries Service, 2014a). In San Diego Bay, the range of eelgrass bed growth is from surface to depths of approximately 10 m, depending on light levels and turbidity; eelgrass bed losses have ceased (U.S. Department of the Navy, 2013d). The recovery of the eelgrass habitat within San Diego Bay is largely attributed to restoration efforts as well as reduction in waste discharges since the 1970s. San Diego Bay currently supports approximately 15 percent of the eelgrass habitat and 50 percent of total eelgrass resources for the State of California (Merkel & Associates Inc., 2014). The Navy established an eelgrass mitigation bank in San Diego Bay in 2008 as mitigation for an action that was unrelated to the Proposed Action in this Environmental Impact Statement/Overseas Environmental Impact Statement (EIS/OEIS.)

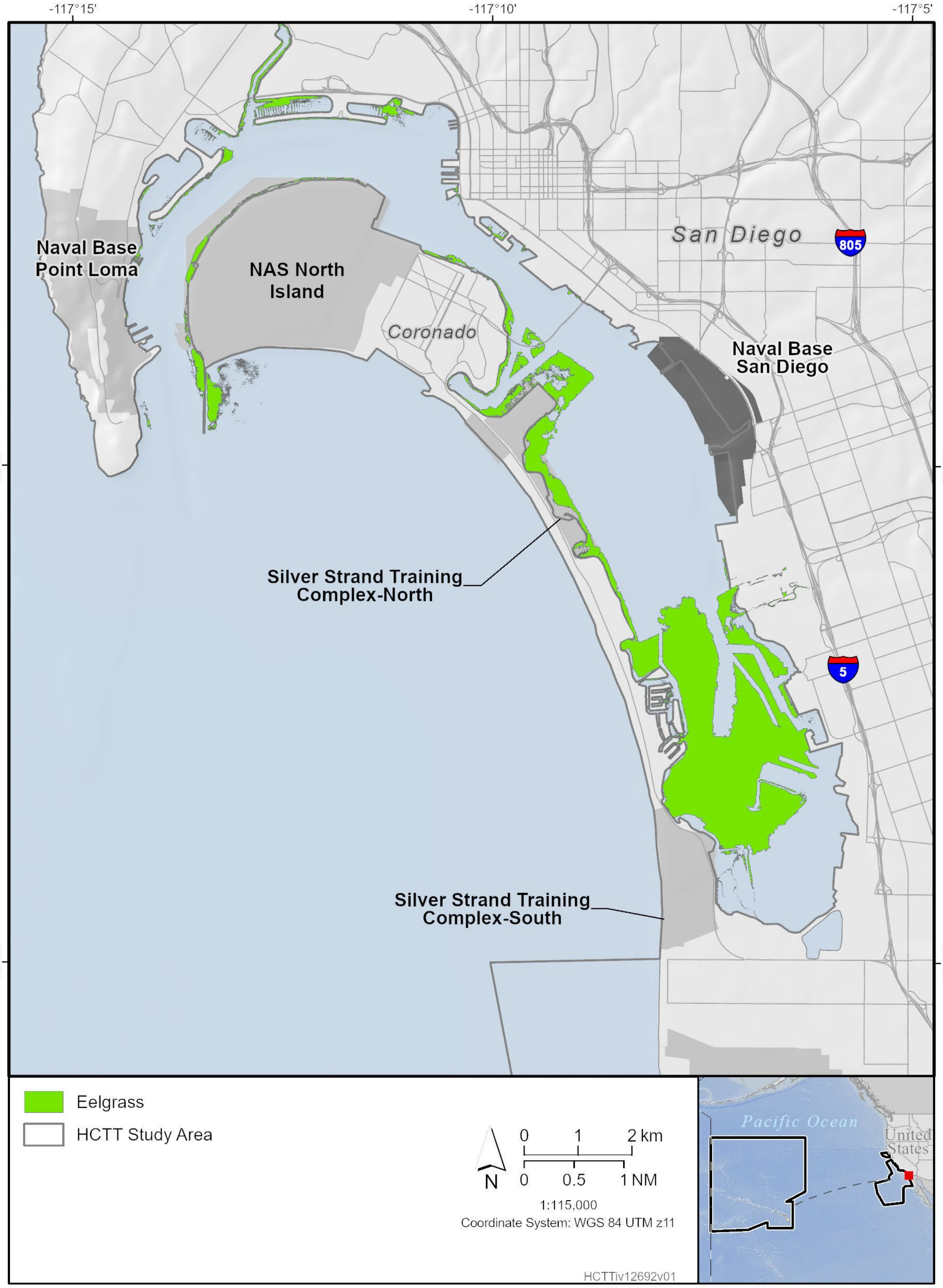

Figure C-3: Eelgrass Beds in San Diego Bay

C.2.2.8.2 Cordgrasses

Cordgrasses are temperate salt-tolerant land plants that inhabit salt marshes, mudflats, and other softbottom coastal habitats (Castro & Huber, 2000c). Cordgrasses are not present in the Hawaii Study Area. California cordgrass (*Spartina foliosa*) can be found in salt marshes and mudflats within the California Study Area. The Atlantic cordgrass (*Spartina alterniflora*), which is an invasive species in California, has not been documented within the study area (Calflora, 2016; California Invasive Plant Council, 2016). Salt marshes develop in intertidal, protected low energy environments, usually in coastal lagoons, tidal creeks, rivers, or estuaries (Mitsch & Gosselink, 2007). The structure and composition of salt marshes provide important ecosystem services. Salt marshes support commercial fisheries by providing habitat for wildlife, protecting the coastline from erosion, filtering fresh water discharges into the open ocean, taking up nutrients, and breaking down or binding pollutants before they reach the ocean (Dreyer & Niering, 1995; Mitsch et al., 2009b). Salt marshes also are carbon sinks (carbon reservoirs) and facilitate nutrient cycling (Bouillon et al., 2009; Chmura, 2009). Carbon sinks are important in reducing the impact of climate change (Laffoley & Grimsditch, 2009), and nutrient cycling facilitates the transformation of important nutrients through the environment. However, sinking salt marshes may damage cordgrasses, a process known as marsh subsidence.

C.2.2.8.3 Mangroves

Mangroves are a group of woody plants that have adapted to brackish water environments (where salt water and freshwater mix) (Ruwa, 1996). All mangrove trees have root systems that stick up in the air for oxygen intake in oxygen-poor soils and secrete salts from the leaves to process fresh water from the saline environment. Mangroves can trap sediments and pollution from terrestrial environments and can shield and stabilize coastlines from wave action. There are no native mangroves in the Hawaii Study Area. The red mangrove (*Rhizophora mangle*) and several other species of mangroves were introduced to Hawaii (Chimner et al., 2006); these species are invasive species. No mangroves are known to occur within California coastal environments.

C.3 Invertebrates

C.3.1 General Background

C.3.1.1 Habitat Use

Marine invertebrates live in all of the world's oceans, from warm shallow waters to cold deep waters. They inhabit the bottom and all depths of the water column in both large marine ecosystems (Insular Pacific-Hawaiian and California Current) and the open-ocean area (North Pacific Subtropical Gyre) that occur in the Study Area (Brusca & Brusca, 2003a). Many species that occur in the water column are either microscopic or not easily observed with the unaided eye (e.g., protozoans, copepods, and the larvae of larger invertebrate species). Many invertebrates migrate to deeper waters during the day, presumably to decrease predation risk. However, some invertebrates, such as some jellyfish and squid species, may occur in various portions of the water column, including near the surface, at any time of day. In addition, under certain oceanographic conditions, other types of invertebrates (e.g., pelagic crabs and by-the-wind sailors [*Velella velella*]) may occur near the surface during the day. The Study Area extends from the bottom up to the mean high tide line (often termed mean high water (MHW) in literature). The description of habitat use in this section pertains to common marine invertebrates found in the different habitats. This section also identifies marine invertebrates that form persistent habitats, which are structures that do not quickly disintegrate or become incorporated into soft or mixed substrate after the death of the organism. The principal habitat-forming invertebrates are corals and shellfish species (e.g., oysters, mussels). In a strict sense, individual invertebrates with hard shells (e.g., molluscs), outer skeletons (e.g., crabs), tubes (e.g., annelid worms), or cavities (e.g., sponges) also may be habitat-forming, providing attachment surfaces or living spaces for other organisms.

Marine invertebrate distribution in the Study Area is influenced by habitat (e.g., abiotic substrate, topography, biogenic [formed by living organisms] features), ocean currents, and physical and water chemistry factors such as temperature, salinity, and nutrient content (Levinton, 2009a). Distribution is also influenced by distance from the equator (latitude) and distance from shore. In general, the number of marine invertebrate species (species richness) increases toward the equator (Cheung et al., 2005; Macpherson, 2002). Species richness and overall biomass are typically greater in coastal water habitats compared to the open ocean due to the increased availability of food and protection that coastal habitats provide (Levinton, 2009a).

The diversity and abundance of Arthropoda (e.g., crabs, lobsters, and barnacles) and Mollusca (e.g., snails, clams, and squid) are highest on the bottom over the continental shelf due to high productivity and availability of complex habitats relative to typical soft bottom habitat of the deep ocean (Karleskint et al., 2006). Organisms occurring in the bathyal and abyssal zones of the ocean are generally small and have sparse populations (Nybakken, 1993). The deep ocean has a limited food supply for sedentary deposit or filter feeders. The only areas of the deep ocean known to be densely populated are hydrothermal vents and cold seeps.

Sandy coastal shores are dominated by species that are adapted to living in shifting substrates, many of which are highly mobile and can burrow. In Hawaii, mole crabs (species name was not provided in the report, but was presumably the Pacific mole crab, *Hippa pacifica*), polychaete worms, and auger snails (*Terebra* species) were identified as common species in the swash zone of sandy beaches (Hawaii Department of Land and Natural Resources, 1980). Studies of coastal locations on Molokai (primarily rocky intertidal areas but also including some sand patches) identified various crabs, amphipods, isopods, worms, and molluscs as observed or expected species (Godwin & Bolick, 2006; Minton & Carnevale, 2006). Common invertebrates of southern California beaches include common sand crab (*Emerita analoga*) and a variety of isopods, amphipods, bivalves, snails, worms, and insects (Dugan et al., 2000; Dugan et al., 2015). Inland soft shores consist of mud flats and sand flats that occur in areas sheltered from strong currents and waves. Soft shore habitats may support a wide variety of invertebrate species including crabs, shrimp, clams, snails, and numerous species of worms. Polychaete worms and crabs are common invertebrates on tidal mud flats in Hawaii (U.S. Fish and Wildlife Service, 2011c). Invertebrates documented in tidal flats in southern California include numerous taxa of worms, crustaceans, and molluscs (Talley et al., 2000; Thompson et al., 1993). California horn snail (*Cerithidea californica*) is the dominant invertebrate of mud flats.

Mixed (e.g., cobble, gravel) and rocky shores provide habitat for a variety of marine invertebrates (e.g., sea anemones, barnacles, chitons, limpets, mussels, urchins, sea stars, sponges, tunicates, and various worms). Rocky intertidal invertebrates may be attached or free living/mobile, and use various feeding strategies (filter-feeders, herbivores, carnivores, scavengers). Many invertebrates occurring in rocky intertidal zones are preyed upon by fish, birds, and other invertebrates. The black abalone (*Haliotis cracherodii*) and white abalone (*Haliotis sorenseni*), which are listed as endangered species under the Endangered Species Act (ESA), occur infrequently in southern California rocky intertidal and subtidal habitats. Hard artificial structures such as pier pilings and seawalls can have a community of invertebrates that is similar to that of rocky habitats.

Vegetated habitats, such as eelgrass in embayments and protected soft bottom coastal areas, surfgrass on rocky intertidal and nearshore subtidal habitat, and kelp forests in nearshore subtidal habitats, support a wide variety of marine invertebrate species. Eelgrass provides important habitat for invertebrates in southern California (Bernstein et al., 2011). More than 50 species of invertebrates occur in surfgrass beds of San Diego County (Stewart & Myers, 1980). Surfgrass also serves as the primary nursery habitat for the commercially important California spiny lobster (*Panuliris interruptus*). Several hundred species of invertebrates have been reported in giant kelp forests of California, in association with rocky substratum, kelp holdfasts, and as epiphytes on kelp blades (Foster & Schiel, 1985). Conspicuous or commonly observed invertebrates in kelp forests include cnidarians (sea anemones, gorgonian sea fans), sponges, arthropod crustaceans (crabs, California spiny lobster), molluscs (abalone, keyhole limpet, octopus, nudibranchs, sea hares), echinoderms (sea cucumbers, sea stars, sea urchins), and tunicates.

Rocky reefs and other rocky habitats may occur in subtidal zones. Invertebrate species composition associated with rocky subtidal habitats may be influenced by depth, size, and structural complexity of the habitat. Hundreds of invertebrate species may occur in rocky habitats, which provide attachment sites for sessile (attached to the bottom) species such as barnacles, bryozoans, limpets, sea anemones, sea fans, sponges, and tunicates, among others. Other invertebrates move about or shelter in crevices, including crustaceans (e.g., crabs, lobsters), echinoderms (e.g., brittle stars, sea cucumbers, sea urchins, sea stars), and molluscs (e.g., snails, nudibranchs, sea hares, octopus).

Shallow-water coral reefs are formed by individual corals with symbiotic, structure-forming algae that require both light and a mean annual water temperature greater than about 64 degrees Fahrenheit (°F) (National Ocean Service, 2016b; Nybakken, 1993). Shallow-water corals occur in the euphotic zone, which is the upper layer of the ocean where light levels are sufficient to support photosynthesis in the symbiotic algae. Shallow-water coral species typically occur in water depths less than 30 m. Shallow-water coral reefs occur on hard substrate throughout the Hawaii Study Area. In addition to the presence of many individual corals, coral reefs also support hundreds of other marine invertebrate species, including representatives of most taxa. The amount of hard reef structure covered by living corals, species richness, and species diversity in the Main Hawaiian Islands remained steady over the time period of 1999 to 2012, with total coverage estimated at about 24 percent, although there was notable variation at individual islands (Rodgers et al., 2015). Coral coverage is below 20 percent at most surveyed locations in the Northwestern Hawaiian Islands, and the coverage appears to have remained stable over the time period of 1981 to 2005, based on survey results at established monitoring sites (Friedlander et al., 2008a). Coral bleaching and mortality events were documented in portions of the Hawaiian archipelago in 2014, 2015, and 2019 (Bahr et al., 2015a; Bahr et al., 2017; National Marine Fisheries Service, 2022b), reducing the amount of live coral coverage in some areas. Surveyed areas that were affected by coral bleaching generally appeared to recover by the end of 2016, but researchers caution that potential future increases in severity and frequency of bleaching events could result in decreased coral coverage in the region (Bahr et al., 2015b; Bahr et al., 2017). Seven species of shallow-water corals dominate waters of the Main Hawaiian Islands: lobe coral (*Porites lobata*), finger coral (*P. compressa*), rice coral (*Montipora capitata*), sandpaper rice coral (*M. patula*), blue rice coral (*M. flabellata*), cauliflower coral (*Pocillopora meandrina*), and corrugated coral (*Pavona varians*) (Friedlander et al., 2008b). Lobe coral is the dominant species at numerous locations in the Northwestern Hawaiian Islands, while table coral (*Acropora cytherea*), cauliflower coral, and rice coral are abundant at some locations (Friedlander et al., 2008a).

Deep-water corals occur in water depths where there is low or no light penetration and therefore typically lack symbiotic algae. As such, deep-water corals do not typically form biogenic reefs, but rather form mounds of mixed (cobble-sized) substrate termed "lithoherms" over hard bottom areas (Lumsden et al., 2007). Differences in water clarity and the resulting light penetration at various locations affect the specific depth at which deep-water corals are found. However, in general, deep-water species are considered to occur at depths below 50 m (National Ocean and Atmospheric Administration, 2016; National Oceanic and Atmospheric Administration & National Marine Fisheries Service, 2008). To build their supporting structures, stony corals require calcium carbonate in the form of aragonite or calcite, which they obtain from seawater where carbonate is in solution. Combinations of temperature and pressure result in a boundary, often called the saturation depth, below which aragonite and calcite tend to dissolve. Therefore, corals (and other invertebrates) occurring below this boundary have difficulty forming persistent structures that contain calcium carbonate, and the aragonite saturation boundary imposes a depth limit for stony coral occurrence. The depth of the saturation boundary varies in different locations, ranging from about 200 to 3,000 m. Accordingly, deep-water corals are found in the depth range of about 50 to 3,000 m (Bryan & Metaxas, 2007; Lumsden et al., 2007; Quattrini et al., 2015; Tittensor et al., 2009). The primary taxa of deep-water corals include hexacorals (stony corals, black corals, and gold corals), octocorals (e.g., true soft corals, gorgonians, and sea pens), and hydrocorals (e.g., lace corals) (Hourigan et al., 2017a). Of the approximately 600 coral species that occur at depths below 50 m, about 20 are considered structure forming (Hourigan et al., 2017a). Stony corals such as *Enallopsammia rostrata* provide three-dimensional structure that may be utilized by other marine species. However, taxa such as black corals, gorgonians, and sea pens may also provide habitat for other marine species, particularly when they occur in dense aggregations. With the exception of sea pens, which occur in soft substrate, deep-water corals generally attach to hard or mixed substrates exposed to strong currents that provide a steady supply of plankton (algae and small animals that drift in the water) to feed on, and that reduce sedimentation that would inhibit colonization and growth of these slow-growing species (Bryan & Metaxas, 2007; Tsao & Morgan, 2005).

A transition zone of reduced light levels, called the mesophotic zone, occurs between the water depths typically associated with shallow-water and deep-water corals. Mesophotic coral communities are composed of stony corals, soft corals, and other structure-forming organisms such as algae and sponges. Some corals with symbiotic, photosynthetic algae occur in the mesophotic zone, although the algae often undergo photosynthesis at reduced rates and the corals, therefore, rely more heavily on planktonic food capture compared to individuals that occur in the euphotic zone. Black corals and octocorals, which do not contain photosynthetic algae, are also characteristic of mesophotic communities. The depth range of the mesophotic zone depends on water clarity, but it is generally considered to extend from 30 m to about 100 to 150 m. Mesophotic communities may occur as deeper extensions of shallow-water reefs or other hard bottom communities (typically in the coastal zone), or they may occur in offshore locations with no connection to shallow-water communities. Mesophotic reefs are usually not detectable on satellite images, which increases the difficulty of identifying and mapping these features. The highest concentrations of stony corals typically occur on persistent, high-relief bottom features that represent a small subset of the hard and, to a lesser extent, mixed substrates of the Study Area. In the Study Area, mesophotic coral communities occur throughout the Hawaiian Archipelago (Baker et al., 2016a). Due to water clarity, corals containing photosynthetic algae occur at depths up to about 150 m in some portions of the Hawaii region.

Chemosynthetic communities may support a relatively high biomass of marine invertebrates. Instead of using photosynthesis driven by sunlight, chemosynthetic organisms derive energy from chemicals originating from the earth’s crust. The primary types of habitats supporting chemosynthetic communities are hydrothermal vents and cold seeps. Hydrothermal vents form when seawater permeates downward through the earth’s crust and upper mantle, becomes superheated, and removes minerals and chemicals from the crust. The heated fluid may then rise through fissures in the crust and reach cold ocean water at the seafloor, where metals and other minerals precipitate out to form mounds or chimneys. Communities of microbes, such as bacteria, may colonize these structures and use chemicals occurring in the fluid (primarily hydrogen sulfide or methane) to make energy. The microbes may then become the base of a food web that contains invertebrates such as crabs, clams, mussels, worms, snails, and shrimp (Ross et al., 2012; Woods Hole Oceanographic Institution, 2015). Cold seeps are similar to hydrothermal vents, but the fluid exiting the crust is cooler, typically moves at a slower rate, and may spread over a larger area. Methane hydrates (ice-like structures that contain methane) are associated with some chemosynthetic communities. Cold seeps are generally associated with hard substrate on offshore shelf breaks, submarine canyons, and seamounts.

In the Hawaiian Islands, a hydrothermal vent field was documented on the summit of Lo’ihi Seamount (renamed in 2021 to Kamaʻehuakanaloa) is located near the Hawaii Island (Garcia et al., 2005). In 1996, seismic events formed a large crater on the summit and destroyed the vent area; however, new vents later re-formed (Wheat et al., 2000). Cold seeps have been found in association with multiple fault systems off southern California, including the San Clemente (Bernardino & Smith, 2010; Torres et al., 2002), San Pedro (Paull et al., 2008), and San Diego Trough faults (Grupe et al., 2015).

C.3.1.2 Movement and Behavior

Marine benthic and epibenthic (animals that live on the surface of the substrate) invertebrates may be sessile, sedentary (limited mobility), or highly mobile (but typically slower than large vertebrates). Several beach invertebrates (e.g., sand crabs, Pismo clams [*Tivela stultorum*], polychaete worms) recruit to beaches during spring and summer and seasonally move to shallow nearshore waters during late fall and winter. Some subtidal epibenthic invertebrates undergo seasonal onshore-offshore migrations associated with reproduction (e.g., California spiny lobster).

Pelagic marine invertebrates include plankton (organisms that do not swim or generally cannot swim faster than water currents) and nekton (active swimmers that can generally swim faster than water currents). Planktonic animals commonly undergo daily migrations to surface waters at dusk and return to deeper waters at dawn. This includes small, microscopic zooplankton and larvae, larger crustaceans (e.g., small shrimp), and jellyfish. Planktonic organisms vary in their swimming abilities, ranging from weak (e.g., larvae) to substantial (e.g., box jellyfish). Nekton such as prawns, shrimps, and squid have relatively strong swimming ability, although they are typically slower than most vertebrate animals.

C.3.1.3 Sound Sensing and Production

In general, organisms may detect sound by sensing either the particle motion or pressure component of sound, or both (refer to Appendix D, Acoustic and Explosive Concepts, for an explanation of these sound components). Aquatic invertebrates probably do not detect pressure since many are generally the same density as water and few, if any, have air cavities that would respond to pressure (Budelmann, 1992b; Popper et al., 2001). Marine invertebrates are generally thought to perceive sound via either external sensory hairs or internal statocysts. Many aquatic invertebrates have ciliated “hair” cells that may be sensitive to water movements, such as those caused by currents or water particle motion very close to a sound source (Budelmann, 1992a, 1992b; Mackie & Singla, 2003). This may allow sensing of nearby prey or predators, or help with local navigation. Detection of particle motion is thought to occur in mechanical receptors found on various body parts (Roberts et al., 2016). Aquatic invertebrates that are able to sense local water movements with ciliated cells include cnidarians, flatworms, segmented worms, molluscs, and arthropods (Budelmann, 1992a, 1992b; Popper et al., 2001). Crustaceans in particular seem to have extensive occurrence of these structures. The sensory capabilities of adult corals are largely limited to detecting water movement using receptors on their tentacles (Gochfeld, 2004), and the exterior cilia of coral larvae likely help them detect nearby water movements (Vermeij et al., 2010).

Some aquatic invertebrates have specialized organs called statocysts that enable an animal to determine orientation, balance, and, in some cases, linear or angular acceleration. Statocysts allow the animal to sense movement and may enable some species, such as cephalopods and crustaceans, to be sensitive to water particle movements associated with sound or vibration (Hu et al., 2009; Kaifu et al., 2008; Montgomery et al., 2006; Normandeau Associates, 2012; Popper et al., 2001). Because any acoustic sensory capabilities, if present, are apparently limited to detecting the local particle motion component of sound (Edmonds et al., 2016), and because water particle motion near a sound source falls off rapidly with distance, aquatic invertebrates are probably limited to detecting nearby sound sources rather than sound caused by pressure waves from distant sources.

In addition to hair cells and statocysts that allow some marine invertebrates to detect water particle motion, some species also have sensory organs called chordotonal organs that can detect substrate vibrations. Chordotonal organs are typically attached to connective tissue of flexible appendages such as antennae and legs (Edmonds et al., 2016). The structures are connected to the central nervous system and can detect some movements or vibrations that are transmitted through substrate.

Available information indicates that aquatic invertebrates are primarily sensitive to low-frequency sounds. Both behavioral and auditory brainstem response studies suggest that crustaceans may sense sounds up to 3 kilohertz (kHz), but greatest sensitivity is likely below 200 hertz (Hz) (Goodall et al., 1990; Lovell et al., 2005; Lovell et al., 2006). Most cephalopods (e.g., octopus and squid) likely sense low-frequency sound below 1 kHz, with best sensitivities at lower frequencies (Budelmann, 1992b; Mooney et al., 2010; Packard et al., 1990). A few cephalopods may sense frequencies up to 1.5 kHz (Hu et al., 2009). Squid did not respond to playbacks of odontocete (e.g., toothed whales) ultrasonic echolocation clicks, likely because these clicks were outside of squid hearing range (Wilson et al., 2007). Although information on the frequency range of the clicks was not provided, ultrasonic sound typically refers to high frequency sounds above the limit of human hearing (greater than about 20 kHz). Similarly, squid did not respond to killer whale echolocation clicks ranging from 199 to 226 decibels (dB) referenced to 1 micropascal (dB re 1 µPa) (Wilson et al., 2007) (refer to Appendix D, Acoustic and Explosive Concepts, for an explanation of this and other acoustic terms). The frequency of the clicks was not provided. However, killer whale echolocation clicks have been reported to be mostly between 45 and 80 kHz (Au et al., 2004). Some researchers have suggested sensitivity to sounds of higher frequencies in some species, although study results are inconclusive. European spiny lobsters (*Palinurus elephas*), some of which were exposed to predators, were found to produce ultrasound signals up to about 75 kHz (Buscaino et al., 2011). The investigators speculated that the signals might have an anti-predator function or might be used in intraspecific communication, although these functions (particularly communication) were considered hypothetical. The results of another study suggest that European spiny lobsters likely use acoustic signals to aggregate (frequency was not specified, although lobsters in the study produced sounds of up to 30 kHz) (Filiciotto et al., 2014). However, information currently available indicates that invertebrates are likely sensitive only to local water movement and to low-frequency particle accelerations generated in their close vicinity (Normandeau Associates, 2012; Solé et al., 2023).

Although many types of aquatic invertebrates produce sound and at least some species have the ability to detect low-frequency particle motion, little is known about the use of sound or whether all sound production is purposeful or merely incidental in some cases (Hawkins et al., 2015; Normandeau Associates, 2012). Some invertebrates have structures that appear to be designed specifically for sound production, and the results of various studies (summarized in the following paragraphs) indicate that sound is used for communication or other behaviors in some species. For example, it has been suggested by numerous researchers that the larvae of some marine species (e.g., crustaceans, molluscs, and corals) use sound cues for directional orientation (Budelmann, 1992a, 1992b; Montgomery et al., 2006; Popper et al., 2001).

Aquatic invertebrates may produce and use sound in territorial behavior, to detect or deter predators, and in reproduction (Popper et al., 2001). Some crustaceans produce sound by rubbing or closing hard body parts together (Au & Banks, 1998; Heberholz & Schmitz, 2001; Latha et al., 2005; Patek & Caldwell, 2006). The snapping shrimp chorus makes up a significant portion of the ambient noise in many locations (Au & Banks, 1998; Cato & Bell, 1992; Heberholz & Schmitz, 2001). Each snapping shrimp click is up to 215 dB re 1 µPa at 1 m (root mean square [rms] is implied, but the authors did not explicitly state sound pressure level or peak sound pressure level), with a peak around 2 to 5 kHz. Some crustaceans, such as the American lobster (*Homarus americanus*) and California mantis shrimp (*Hemisquilla californiensis*), may also produce sound by vibrating the carapace (Henninger & Watson, 2005; Patek & Caldwell, 2006). Spiny lobsters typically produce low-frequency rasps by moving a structure at the base of the antennae over a rigid file (Buscaino et al., 2011). Other crustaceans make low-frequency rasping or rumbling noises, perhaps used in defense or territorial display (Patek & Caldwell, 2006; Patek et al., 2009), or perhaps used incidental to a visual display. The aquatic isopod *Cymodoce japonica* produces sound by rubbing body parts together (Nakamachi et al., 2015).

Reef noises, such as fish pops and grunts, sea urchin grazing (around 1 kHz), parrotfish grazing, and snapping shrimp noises (around 5 kHz) (Radford et al., 2010), may be used as a cue by some aquatic invertebrates. Nearby reef noises were observed to affect movements and settlement behavior of coral and crab larvae (Jeffs et al., 2003; Radford et al., 2007; Stanley et al., 2010; Vermeij et al., 2010), although chemical cues and substrate color are also used by some species (Foster & Gilmour, 2016). Larvae of other crustacean species, including pelagic and nocturnally emergent species that benefit from avoiding coral reef predators, appear to avoid reef noises (Simpson et al., 2011). Detection of reef noises is likely limited to short distances. Low-frequency sound pressure and particle motion have been measured near a coral reef off Maui, Hawaii (Kaplan & Mooney, 2016). Results indicate that adult cephalopod species would not be able to detect the low level of particle acceleration at the measurement point nearest the reef (50 m). The specific particle acceleration levels detected by marine invertebrate larvae are unknown, but the authors suggest that invertebrate larvae would be unlikely to detect particle acceleration at distances beyond 150 m at this reef. Playback of reef sounds increased the settlement rate of eastern oyster (*Crassostrea virginica*) larvae (Lillis et al., 2013). Green-lipped mussel (*Perna canaliculus*) larvae settlement rate increased when exposed to underwater noise produced by a ferry (Wilkens et al., 2012).

C.3.1.4 General Threats

General threats to marine invertebrates include overexploitation and destructive fishing practices (Halpern et al., 2008b; Jackson et al., 2001; Kaiser et al., 2002; Miloslavich et al., 2011; Pandolfi et al., 2003), habitat degradation resulting from pollution and coastal development (Cortes & Risk, 1985; Downs et al., 2009; Mearns et al., 2011), disease (Porter et al., 2001), invasive species (Bryant et al., 1998; Galloway et al., 2009; Wilkinson, 2002) (which may be introduced as a result of growth on vessel hulls or bilge water discharge), oil spills (Yender et al., 2010), global climate change and ocean acidification (Hughes et al., 2003), and possibly human-generated noise (Brainard et al., 2011; Vermeij et al., 2010). A relatively new threat to marine invertebrates is bioprospecting, which is the collection of organisms in pursuit of new compounds for development of pharmaceutical products (Radjasa et al., 2011). Numerous bioactive products have been isolated from marine invertebrates collected in the Hawaii Exclusive Economic Zone (EEZ) (Leal et al., 2012).

Compared to many other invertebrate taxa, the threats to corals are well-studied. Numerous natural and human-caused stressors may affect corals of the Main Hawaiian Islands, including thermal stress, disease, tropical storms, coastal development and pollution, erosion and sedimentation, tourism/recreation, fishing, trade in coral and live reef species, vessel anchoring or groundings, marine debris, predation, invasive species, military and other security-related activities, and hydrocarbon exploration (Center for Biological Diversity, 2009; National Oceanic and Atmospheric Administration, 2008a, 2008b). Stressors associated with the Northwestern Hawaiian Islands are similar but, in the case of direct human-caused impacts, lesser in degree because the islands are more remote.

Coral bleaching, which occurs when corals expel the symbiotic algae living in their tissues, is a stress response to changes in environmental parameters such as temperature or light. Compared to other regions of the world, few major coral bleaching events have occurred in the Hawaiian Islands. The first known large-scale bleaching event occurred in 1996, primarily affecting portions of the Main Hawaiian Islands. A second event occurred in 2002 in the Northwestern Hawaiian Islands (Jokiel & Borwn, 2004). More recently, bleaching events were documented at Kane’ohe Bay on the northeast coast of Oahu in 2014 (Bahr et al., 2015b) and other portions of the Main Hawaiian Islands in 2014, 2015, and 2019 (National Oceanic and Atmospheric Administration, 2015b, 2022). In Kane’ohe Bay, susceptibility to bleaching, severity of impacts, and recovery time was strongly influenced by the type of symbiotic algae, varying coping mechanisms in individual corals, and abiotic (e.g., hydrodynamics) factors (Cunning et al., 2016).

Factors that seem to be important for coral reef resilience (ability of a reef to resist and recover from environmental disturbance) were identified by McClanahan et al. (2012). Some factors are large in scale and difficult to manage, while others, such as fishing methods and adjacent watershed pollution, are more easily affected by local management practices. NOAA’s Pacific Islands Fisheries Science Center evaluated numerous areas of the Main Hawaiian Islands in relation to these factors and developed composite resiliency scores. Generally, the highest scores were associated with sparsely populated areas (e.g., Ni’ihau, portions of Maui), while the lowest scores were associated with densely populated areas (e.g., portions of O’ahu) (Pacific Islands Fisheries Science Center, 2014). Primary threats to deep-water or cold-water corals include bottom fishing, hydrocarbon exploration, cable and pipeline placement, and waste disposal (e.g., discarded or lost rope and fishing equipment, dredged sediments) (Freiwald et al., 2004).

Threats related to water quality, marine debris, and climate change are further described in the subsections below.

Water Quality

Invertebrates may be affected by changes in water quality resulting from pollution, turbidity and increased particle deposition that may occur as a result of sediment disturbance, and waste discharge. Stormwater runoff and point source discharges associated with coastal development may introduce pollutants into bays and other nearshore coastal areas. The pollutants may degrade sediment and water quality, which in turn can impact marine invertebrate communities. Sediment disturbance may result from activities such as dredging, which can affect sensitive species such as some corals (Erftemeijer et al., 2012). In addition to dredging, erosion due to storm runoff may cause changes in the frequency or magnitude of sedimentation in areas in proximity to ocean outfalls, estuarine inlets, and major river discharges.

Ship discharges may affect water quality and invertebrates associated with the impacted water. Discharged materials include sewage, bilge water, graywater, ballast water, and solid waste (e.g., food and garbage). Discharges may originate from military, commercial, and recreational vessels. Under provisions of the CWA, the USEPA and the DoD have developed Uniform National Discharge Standards to address discharges from U.S. military vessels.

Marine invertebrates can be impacted by exposure to oil due to runoff from land, natural seepage, or accidental spills from offshore drilling/extraction or tankers (White et al., 2012). Reproductive and early life stages are especially sensitive to oil exposure. Factors such as oil type, quantity, exposure time, and season can affect the toxicity level. Experiments using corals indicate that oil exposure can result in death, decreased reproductive success, altered development and growth, and altered behavior (White et al., 2012; Yender et al., 2010).

C.3.1.4.1 Climate Change

The primary concerns of climate change in the context of impacts on marine invertebrates include increased water temperature, ocean acidification, increased frequency or intensity of cyclonic storm events, and sea level rise.

Increases in ocean temperature can lead to coral stress, bleaching, and mortality (Lunden et al., 2014). Bleaching of corals and other invertebrates that contain symbiotic algae in their tissues (e.g., some anemones and clams) is often tied to atypically high sea temperatures (Lough & van Oppen, 2009; National Ocean Service, 2016a). Bleaching events have increased in frequency in recent decades. Coral bleaching on a global scale occurred during the summers of 2014, 2015, 2016, and 2023–2024 (Eakin et al., 2016; National Oceanic and Atmospheric Administration, 2024b). In addition to elevated sea temperatures, atypically low sea temperatures may also cause mortality to corals and most other reef organisms (Colella et al., 2012; Lirman et al., 2011; National Ocean Service, 2016a), suggesting that widening climate extremes could cause more coral bleaching. In one experiment, three coral species that experienced bleaching had reduced ability to remove sediments from their tissue surface (Bessell-Browne et al., 2017). Response to thermal stress may differ across species or within different environmental contexts, with some species or taxa being more tolerant than others (Bahr et al., 2016; Guest et al., 2016; Hoadley et al., 2015). For example, in the Caribbean Sea, while numerous stony corals may be negatively affected by increased water temperature, some gorgonian corals have been found to persist or increase in abundance under similar conditions (Goulet et al., 2017). The results of one study suggest that some corals may acclimate to increased water temperature over time, exhibiting less temperature sensitivity and resulting bleaching activity (McClanahan, 2017). Skeletal formation of post-settlement individuals of the plate coral *Acropora spicifera* was not affected by increased water temperature (Foster et al., 2016). However, exposure to lowered pH was found to increase the potential for negative effects associated with subsequent water temperature increase in one stony coral species (Towle et al., 2016). In addition to potential physiological effects, the distribution of some invertebrates may be affected by changing water temperature. Northern and southern shifts in the geographic center of abundance of some benthic invertebrates along the U.S. Atlantic coast have occurred over the last 20 years, presumably in response to increased water temperature (Hale et al., 2017).

Ocean acidification has the potential to reduce calcification and growth rates in species with calcium carbonate skeletons, including shellfish (e.g., clams, oysters), corals, and sponges (Clark & Gobler, 2016; Cohen et al., 2009), and crustose coralline algae that contain calcite in their cell walls (Roleda et al., 2015). For example, newly settled individuals of the plate coral *A. spicifera* that were exposed to elevated CO_2 and lowered pH levels showed decreased mineral deposition and evidence of skeletal malformation (Foster et al., 2016), and water acidification decreased the survival, size, and weight of bay barnacles (*Balanus improvises*) (Pansch et al., 2018). The results of one study suggest that community-level effects to corals can be more evident than effects to individual corals (Carpenter et al., 2018). Many species within these taxa are important structure-building organisms. In addition to corals and shellfish, acidification may also affect weakly calcified taxa such as lobsters and sea cucumbers (Small et al., 2016; Verkaik et al., 2016).

Some climate change models predict that the depth below which corals are unable to form calcium carbonate skeletons will become limited as the oceans acidify and temperatures increase, potentially decreasing the occurrence and habitat-forming function of corals and other invertebrates. Deep-sea scleractinian stony corals could be particularly vulnerable due to habitat loss and decreased larvae dispersal (Miller et al., 2011). However, a recent study of successive generations of shallow-water reef-building corals exposed to increased water temperature and acidification suggests some corals may be able to tolerate rapidly changing environmental conditions better than previously thought (Putnam & Gates, 2015).

In addition to physical effects, increased acidity may result in behavioral changes in some species. For example, acidification of porewater was found to affect burrowing behavior and juvenile dispersal patterns of the soft-shell clam (*Mya arenaria*) (Clements et al., 2016), and increased acidity caused a reduction in the loudness and number of snaps in the snapping shrimp *Alpheus novaezelandiae* (Rossi et al., 2016). As discussed for thermal stress, some invertebrate species may be more tolerant of changing acidity levels than others (Bahr et al., 2016). One study found that lowered pH caused a significant decrease in black band disease progression in mountainous star coral (Muller et al., 2017). Another study of three Arctic marine bivalves concluded that at least two of the species are generally resilient to decreased pH (Goethel et al., 2017). A study of the deep-water stony coral *Desmophyllum dianthus* found that the species was not affected by increased acidity under conditions of ambient water temperature but that stress and decreased calcification occurred when acidity and water temperature were both increased (Murray et al., 2016). Gelatinous invertebrates such as jellyfish generally seem to be tolerant of increased water acidity (Treible et al., 2018).

Although the potential effects that climate change could have on future storm activity is uncertain, numerous researchers suggest that rising temperatures could result in little change to the overall number of storms, but that storm intensity could increase (Voiland, 2013). Increased storm intensity could result in increased physical damage to individual corals and reefs constructed by the corals (which support numerous other invertebrate taxa), overturning of coral colonies, and a decrease in structural complexity due to disproportionate breakage of branching species (Heron et al., 2008; The Nature

Conservancy, 2015). However, large storms such as hurricanes may also have positive impacts on corals, such as lowering the water temperature and removing less resilient macroalgae from reef structures, which can overgrow corals.

Sea level rise could affect invertebrates by modifying or eliminating habitat, particularly estuarine and intertidal habitats bordering steep and artificially hardened shorelines (Fujii, 2012). It is possible that intertidal invertebrates would colonize newly submerged areas over time if suitable habitat were present. Coral reef growth may be able to keep pace with sea level rise because accretion rates of individual corals are generally greater than projected potential rates of sea level rise (The Nature Conservancy, 2016). Corals are currently subjected to tidal fluctuations of up to several meters (The Nature Conservancy, 2015; U.S. Geological Survey, 2016). However, the overall net accretion rate of coral reefs may be much slower than the rate of individual corals, decreasing the overall ability of reefs to keep pace with rising water levels. In addition, the compounding effect of other stressors (e.g., ocean acidification) is unknown. In an evaluation of threats to corals previously petitioned for listing under the ESA, sea level rise was considered a low to medium influence on extinction risk (Brainard et al., 2011).

Additional concerns include the potential for changes in ocean circulation patterns that affect the planktonic food supply of filter- and suspension-feeding invertebrates (e.g., corals) (Etnoyer, 2010). An increase in the future incidence of diseases in marine organisms is also theorized (Harvell et al., 2002). In addition, there is concern that cumulative effects of threats from fishing, pollution, and other human disturbance may reduce the tolerance of corals to global climate change (Ateweberhan et al., 2013; Ateweberhan & McClanahan, 2010).

C.3.1.4.2 Marine Debris

Marine debris (especially plastics) is a threat to many marine ecosystems, particularly in coastal waters adjacent to urban development. Microplastics (generally considered to be particles less than 5 mm in size), which may consist of degraded fragments of larger plastic items or intentionally manufactured items (e.g., microbeads), are of concern because of their durability and potential to enter marine food webs (Setala et al., 2016). Field and laboratory investigations have documented ingestion of microplastics by marine invertebrates including bivalve molluscs; crustacean arthropods such as lobsters, shore crabs, and amphipods; annelid lugworms; and zooplankton (Browne et al., 2013; Setala et al., 2014; Von Moos et al., 2012; Watts et al., 2014). While animals with different feeding modes have been found to ingest microplastics, laboratory studies suggest that filter-feeding and deposit feeding benthic invertebrates are at highest risk (Setala et al., 2016).

Marine debris, including large amounts of plastic, is present in surface waters around the Main Hawaiian Islands and Northwestern Hawaiian Islands, and is found on coral reefs as well (Cooper & Corcoran, 2010; Dameron et al., 2007). The Hawaiian Archipelago is located within the North Pacific Gyre, which consolidates debris originating in various areas of the Pacific Ocean. However, there have been no surveys specifically conducted to investigate marine debris on the seafloor in Hawaii. A visual survey of the seafloor that included a portion of the Navy's SOCAL Range Complex was conducted as part of a 15-year quantitative assessment of marine debris on the seafloor off the California coast (Watters et al., 2010). Plastics were the most abundant material found and, along with recreational monofilament fishing line, dominated the debris encountered on the seafloor. U.S. Navy vessels have a zero-plastic discharge policy and return all plastic waste to appropriate disposal or recycling sites on shore. The visual survey encountered only a single object that was potentially "military" in origin (it appeared to be a shell casing). A survey conducted at Monterey Canyon off California found that items of military origin were among the least frequently encountered types of identified debris (Schlining et al., 2013).

Recent studies in the Southern California Bight found that marine debris (primarily plastic) occurred in about one-third of seafloor areas surveyed (McLaughlin et al., 2022; Moore et al., 2016). Microplastic particles were more prevalent in shallow nearshore areas (ports, marinas, bays, and estuaries) than in offshore areas. Another study of marine debris along the U.S. West Coast characterized the composition and abundance of man-made marine debris at 1,347 randomly selected stations during groundfish bottom trawl surveys that took place in 2007 and 2008 (Keller et al., 2010). The sample sites included some locations within the California portion of the HCTT Study Area. A subset of the sites sampled included historically used post-WWII dump sites. Recovered items identifying the sites as post-WWII era dump sites included equipment described as “helmets,” “gas masks,” “uniforms,” and other miscellaneous and diverse items such as “plastic,” “file cabinets,” and “buckets.” Since approximately the 1970s, items such as these are no longer disposed of at sea. The items listed here are not military expended materials and would not be expended during training and testing activities in the HCTT Study Area. For this reason, the characterization of “military debris” in the study has little if any relevance to the Proposed Action or to present-day standard Navy conduct that includes (among other procedures) restrictions on the discharge of plastics at sea.

C.3.1.4.3 Invasive Species

Introduction of invasive species is considered one of primary threats to ESA-listed species (Anttila et al., 1998; Costante, 2021; Dueñas et al., 2018; Noss et al., 2020; Pimentel et al., 2004; Wilcove & Chen, 1998). As early as 2001, 343 aquatic invasives species were documented as occurring within Hawaiian waters, with the origin of many of these species coming from Indo-Pacific and Philippine Islands region (Eldredge & Carlton, 2002; Eldredge & Smith, 2001). In the Hawaii Study Area, invasive marine invertebrates include Mediterranean mussel (*Mytilus galloprovincialis*), orange keyhole sponge (*Mycale armata*), upside-down jellyfish (*Cassiopea andromeda*), Samoa crab (*Scylla serrata*), American oyster (*Crassostrea virginica*), pulse coral (*Unomia stolonifera*), cabbage coral (*Montipora foliosa*), finger coral (*Montipora digitata*), and the Manjano anemone (*Anemonia manjano*) (Carlton & Eldredge, 2015; Carlton & Schwindt, 2024; Eldredge & Carlton, 2002; Eldredge & Smith, 2001). Many of these nonindigenous species are known to predate upon and outcompete species native to the Hawaiian region of the Study Area (Eldredge & Smith, 2001; Garcia et al., 2021; Kueffer et al., 2010).

Since at least 2001, non-indigenous species of octocorals have been identified in Hawaiian waters (Kahng, 2006; Kahng & Grigg, 2005). At that time, the octocoral *Carijoa riisei* was discovered overgrowing black coral in the Au’au Channel, which separates Maui and Lanai (Kahng & Grigg, 2005). In May 2020, three new nonindigenous species of coral (*Montipora foliosa*, *Montipora digitata*, and the third species either *Montipora stellata* or *Monitipora carinata*) and an anemone (*Anemonia manjano*) were confirmed in Kāneʻohe Bay (Division of Aquatic Resources, 2021). Even more recently, colonies of the invasive octocorals *Unomia stolonifera* and *Capnella cf. spicata* were detected in the southern region of the Main Channel at JBPHH, Hawaii in the vicinity of Bishop Point (Miller et al., 2023). The Navy funded combined remotely operated vehicle, drop camera, and diver surveys to define the extent of their colonization, which could be used to assist in determining appropriate control measures. These surveys estimated that the total area where these two nonindigenous octocorals occur was approximately 336,000 square meters (Miller et al., 2023).

California's extensive coastline and diverse marine ecosystems make it particularly vulnerable to marine invasive species. Certain areas along the coast, referred to as hotspots, are at a higher risk of invasion due to factors such as increased shipping activity, aquaculture operations, and recreational use (California Department of Fish and Wildlife, 2024b). Located off the coast of Southern California, the Channel Islands are also at risk of marine invasions (California Department of Fish and Wildlife, 2024b). Notable invasive species in the area include the Mediterranean fanworm (*Sabella spallanzanii*) and the clubbed tunicate (*Styela clava*) (California Department of Fish and Wildlife, 2024b).

C.3.2 Endangered Species Act-Listed Species

C.3.2.1 Black Abalone (*Haliotis cracherodii*)

C.3.2.1.1 Status and Management

The black abalone (*Haliotis cracherodii*) was listed as endangered under the ESA in 2009. A dramatic decline in abundance, likely caused by a disease known as withering syndrome (explained in more detail below), prompted closure of both the commercial and recreational fisheries in California. The State of California imposed a moratorium on black abalone harvesting throughout California in 1993 and on all abalone harvesting in central and southern California in 1997 (Butler et al., 2009). Numerous California State Marine Protected Areas provide additional protection for abalone. An Abalone Recovery Management Plan was adopted by the State of California in 2005.

NMFS prepared a status review for this species in 2009 (Butler et al., 2009), 2017 (National Marine Fisheries Service, 2018e), and announced in 2023 the intent to prepare an updated status review (Endangered and Threatened Species; Initiation of 5-Year Review for the Endangered Black Abalone and the Endangered White Abalone, 88 *Federal Register* [FR] 43308 [July 7, 2023]). Critical habitat was designated for black abalone by NMFS in 2011 (Endangered and Threatened Wildlife and Plants: Final Rulemaking to Designate Critical Habitat for Black Abalone, 76 FR 66806–66844 [October 27, 2011]).

Various projects are in place to monitor the status of the species, to understand and address withering disease, to improve reproduction, and to minimize illegal harvest. For instance, the Navy monitors black abalone populations on SCI and San Nicolas Islands (SNI) and Point Loma, San Diego, and the species is managed under both the SCI Integrated Natural Resources Management Plan (INRMP) and SNI INRMP. The Navy has conducted and continues to provide funding for ongoing surveys of rocky intertidal areas on SCI, SNI, and Point Loma, including surveys specifically for black abalone (Fletcher et al., 2021; Graham et al., 2014; Hong et al., 2023a; Hong et al., 2023b; Kenner & Yee, 2022; Tierra Data, 2008; U.S. Department of the Navy, 2022).

Critical Habitat

Critical habitat for black abalone was designated on November 28, 2011, and includes approximately 360 km^2 of rocky intertidal and subtidal habitat within five segments of the California coast between the Del Mar Landing Ecological Reserve to the Palos Verdes Peninsula, as well as on the Farallon Islands, Año Nuevo Island, San Miguel Island, Santa Rosa Island, Santa Cruz Island, Anacapa Island, Santa Barbara Island, and Santa Catalina Island (76 FR 66806). This designation also includes rocky intertidal and subtidal habitats from the MHW to a depth of – 6 m (relative to the mean lower low water line), as well as the coastal marine waters encompassed by these areas (76 FR 66806). In addition, two specific areas, SNI and SCI, are excluded from the designation, based on INRMPs for these areas that provide benefits to black abalone.

Based on the best available scientific information, the following primary constituent elements essential for the conservation of black abalone include:

1. Rocky substrate
2. Food resources

3. Juvenile settlement habitat
4. Suitable water quality
5. Suitable nearshore circulation patterns

Recovery Goals

In 2020, NMFS finalized the recovery plan for black abalone (National Marine Fisheries Service, 2020b). The goal of the recovery plan is to restore black abalone populations in the wild such that the species can be downlisted to threatened status and subsequently delisted (i.e., removed from the Endangered Species List). NMFS also developed recovery objectives used to describe the conditions necessary to achieve the recovery goal. These objectives include:

- Increase the abundance, productivity, spatial structure, and diversity of black abalone populations to levels that support the species' long-term survival, viability, and resilience to threats.
- Sufficiently address the threats of concern (e.g., contaminant spills, spill response activities, illegal harvest, and potential introductions of pathogens).

C.3.2.1.2 Habitat and Geographic Range

The distribution of black abalone ranges approximately from Point Arena in northern California to Bahia Tortugas and Isla Guadalupe in Mexico (Butler et al., 2009). Although the geographic range of black abalone extends to northern California, the most abundant populations historically have occurred in the Channel Islands (Butler et al., 2009). A map of the black abalone range can be accessed on NMFS Office of Protected Resources website.

Black abalone live on rocky substrates in the high to low intertidal zone (with most animals found in the middle and lower intertidal) within the California portion of the HCTT Study Area. They occur among other invertebrate species, including California mussels (*Mytilus californianus*), gooseneck barnacles (*Pollicipes polymerus*), and sea anemones (e.g., giant green anemone (*Anthopleura xanthogrammica*). Of the species of abalone in the waters of California, the black abalone inhabits the shallowest areas. It is rarely found deeper than 6 m, and smaller individuals generally inhabit the higher intertidal zones. Complex surfaces with cracks and crevices may be crucial habitat for juveniles, and appear to be important for adult survival as well (Butler et al., 2009).

The black abalone diet varies with life history stage. As larvae, black abalone receive nourishment from their egg yolks and do not actively feed. Settled abalone clamp tightly to rocky substrates and feed on crustose coralline algal matter that they scrape from the rocks. Young juveniles feed on bottom-dwelling diatoms, bacterial films, and microflora. As they increase in size and become less vulnerable to predation, abalone move into more open locations on rocks (though still cryptic) to forage. Adult black abalone feed primarily on drifting plant fragments and attached macroalgae (Butler et al., 2009; Smith et al., 2003). The primary predators of abalone are fish, sea otters, sea stars, and a variety of invertebrates, as well as humans through illegal harvesting (National Oceanic and Atmospheric Administration Fisheries, 2018; Smith et al., 2003).

C.3.2.1.3 Population Trends

Black abalone were generally abundant before 1985 in the coastal waters throughout the species' range, although abundance has historically not been considered high north of San Francisco. Substantial populations also occurred in the coastal waters of the Channel Islands of southern California. In the early

1970s, the black abalone constituted the largest abalone fishery in California. Black abalone populations south of Monterey County, California, have experienced 95 percent or greater declines in abundance since the mid-1980s as a result of fishing pressure in combination with withering syndrome (Neuman et al., 2010). Withering syndrome is caused by the bacteria species *Candidatus Xenohaliotis californiensis*, which attacks the lining of the abalone's digestive tract, inhibiting the production of digestive enzymes, which ultimately causes the muscular "foot" to wither and atrophy. This impairs the abalone's ability to adhere to rocks (Butler et al., 2009), making it more vulnerable to predation or starvation.

Major declines in abundance in the Channel Islands, the primary fishing grounds for this species before closure of the abalone fishery, have severely reduced the population as a whole (Butler et al., 2009). Surveys of rocky intertidal habitat at SCI have resulted in a total population estimate of approximately 100 to 300 individuals of black abalone, representing less than 0.1 percent of historical levels on the island (Raimondi et al., 2012; Tierra Data, 2008). Surveys of rocky shores between 2016 and 2020 indicate black abalone continue to be rare at SCI (Fletcher et al., 2021).

In 2018, the Black Abalone Status Review Team noted that some locations that were historically impacted by overfishing and more recently severely impacted by withering syndrome, showed increased recruitment and increased numbers (National Marine Fisheries Service, 2018a). They indicated that successful fertilization and recruitment is possible in populations that are below this minimum density. SNI is one of the only locations in southern California where black abalone have been increasing and where multiple recruitment events have occurred since 2005 (Butler et al., 2009). However, the Review Team concluded that the status of black abalone largely remains the same as it was at the time of the listing, and therefore recommended the classification of endangered remains the same (National Marine Fisheries Service, 2018a).

C.3.2.1.4 Population Threats

The black abalone population is declining because of historical overfishing and ongoing threats of withering syndrome, illegal harvest, pollution, and natural predation. The spread of withering syndrome is enhanced by periods of ocean warming, such as El Niño events (Neuman et al., 2010). Although there is no documented causal link between withering syndrome and long-term climate change, historical patterns suggest that ocean warming may increase the susceptibility of black abalone to the disease. Decreased population density is an additional factor in the species decline (Neuman et al., 2010). The black abalone is a broadcast spawner (gametes released into the water and fertilization occurs externally), and simultaneous spawning by males and females in close proximity (within a few ft.) is required for successful reproduction. In areas where black abalone have been overfished or otherwise reduced, the distance between adult males and females may be too great or the population density too low to sustain local populations (Butler et al., 2009; Neuman et al., 2010). There is some concern that the invasive macroalga *Sargassum horneri*, first documented off southern California in 2003 and currently distributed in coastal waters from Santa Barbara to central Baja California, Mexico, has the potential to affect black abalone populations. Long-term ecological implications of the presence of the invasive species are uncertain but potentially include displacement of native kelp (Kaplanis et al., 2016; Marks et al., 2015), which is a food source for black abalone.

C.3.2.2 White Abalone (*Haliotis sorenseni*)

C.3.2.2.1 Status and Management

The white abalone (*Haliotis sorenseni*) was listed as endangered under the ESA in 2001 (66 FR 29046) and is recognized as one stock (Hobday & Tegner, 2000). Overfishing in the 1970s reduced the population to such low densities that successful reproduction was severely restricted. White abalone populations continue to be threatened primarily by reproductive failure (Hobday et al., 2001; National Marine Fisheries Service & Southwest Regional Office, 2008b). Critical habitat is not designated for this species.

The State of California suspended all forms of harvesting of the white abalone in 1996 and, in 1997, imposed an indefinite moratorium on the harvesting of all abalone in central and southern California (National Marine Fisheries Service & Southwest Regional Office, 2008b). NMFS determined that informing the public of the locations of critical habitat, which includes areas where white abalone still exist, would increase the risk of illegal harvesting of white abalone (National Marine Fisheries Service & Southwest Regional Office, 2008b). Potential habitat may exist between Point Conception, California, and the California/Mexico border, with much of it occurring in the isolated, deep waters off the Channel Islands. In reaction to concerns over the status of white abalone, the White Abalone Restoration Consortium was formed to propagate a captive-reared stock to enhance the depleted wild stock (National Marine Fisheries Service & Southwest Regional Office, 2008b). There is now a captive breeding program at the Bodega Bay Marine Laboratory, University of California Davis, in partnership with several facilities throughout California.

The Navy's INRMP for Naval Auxiliary Landing Field SCI (U.S. Department of the Navy, 2013b) includes specific management mandates for black and white abalone, including conducting periodic island-wide nearshore surveys, multibeam sonar nearshore habitat surveys, and quantification of suitable deep rocky habitats. The Navy cooperates with white abalone co-managers California Department of Fish and Wildlife and NMFS to monitor white abalone populations and habitat at SCI.

In July 2016, the Navy and NMFS entered into a 7-year Memorandum of Agreement to fund projects benefitting white abalone recovery (U.S. Department of the Navy & National Oceanic and Atmospheric Administration, 2016). The activities, which included field and laboratory projects, focused on Tanner and Cortes Banks, but also occurred at SCI and Point Loma. Programs included in the agreement consisted of field surveys and management assessments, development of tagging methods, disease studies, genetic evaluation, and outplanting monitoring. In November 2019, experimental outplanting of white abalone occurred (National Marine Fisheries Service, 2024h). Captive-bred juvenile white abalone were released for the first time into coastal waters off southern California. Since then, additional juveniles have been outplanted and monitored to assess their survival, movements, and growth over time.

Critical Habitat

Critical habitat has not been designated for white abalone.

Recovery Goals

In 2008, NMFS finalized a white abalone recovery plan (National Marine Fisheries Service & Southwest Regional Office, 2008a). The goal of the recovery plan is to increase wild white abalone abundance in California to viable and self-sustaining levels such that the species can be downlisted to threatened status and subsequently removed from the Endangered Species List. NMFS also developed recovery objectives used to describe the conditions necessary to achieve the recovery goal. These objectives include the following:

- Improve our knowledge of the current status of white abalone in the wild so that extinction risks can be estimated, and a plan can be developed for carrying out recovery actions in a way that will most efficiently achieve the delisting criteria.
- Reduce or eliminate existing threats to white abalone in the wild.
- Downlist and eventually delist white abalone by defining a safe population level (i.e., delisting criteria) which includes viable/sustainable subpopulations at a number of locations throughout the former range of the species.

C.3.2.2.2 Habitat and Geographic Range

The white abalone is a well-concealed, sessile, bottom-dwelling species that prefers reefs and rock piles with low relief areas surrounded by sandy areas (Hobday & Tegner, 2000). White abalone in the Southern California Bight typically inhabit depths ranging from about 20 to 60 m, with the highest densities occurring between 40 and 50 m (Butler et al., 2006a). White abalone were found in waters deeper than other west coast abalone species (Hobday et al., 2001). Overall, habitat associations of white abalone depend on its main food sources, drift macroalgae and a variety of red algae (National Oceanic and Atmospheric Administration, 2018b). Thus, depth distribution is limited by water clarity and light penetration as well as by the availability of hard substrate or anchoring points on the bottom (Butler et al., 2006a). Evidence suggests that white abalone prefer the sand and rock interface at the reef's edge, rather than the middle sections of reefs. Sand channels may be important for movement and concentration of drifting fragments of macroalgae and red algae (National Marine Fisheries Service & Southwest Regional Office, 2008b). Postlarval and juvenile individuals often occur in sheltered areas to decrease susceptibility to predation, while adults occur in more open areas.

White abalone were historically found between Point Conception, California, and Punta Abreojos, Baja California, Mexico, at depths as shallow as 5 m (National Marine Fisheries Service & Southwest Regional Office, 2008b). The northern portion of the range includes the San Clemente (Navy-owned) and Santa Catalina Islands in the northeastern corner of the California portion of the HCTT Study Area (Butler et al., 2006a; National Marine Fisheries Service & Southwest Regional Office, 2008b). On the southern end of the range, the species was common around a number of islands, including Isla Cedros and Isla Natividad, Mexico (Hobday & Tegner, 2000). The current range in California appears similar to that of the historical range, although the species occurs in extremely reduced numbers. Information on the current range off Baja California is not available (National Marine Fisheries Service & Southwest Regional Office, 2008b).

Except for some isolated survivors, within the Study Area, the species is largely only distributed around the Channel Islands and along various banks (Behrens & Lafferty, 2005; Butler et al., 2006b; Hobday & Tegner, 2000; Raimondi, 2016; Rogers-Bennett et al., 2002; Stierhoff et al., 2014b). The species is known to occur off SCI, Santa Catalina, and Santa Barbara Islands and at Tanner and Cortes Banks (approximately 50 mi southwest of SCI). Both these banks are underwater mountains that occur off the coast of southern California. One study documented 5 square miles (mi.^{2}) of available white abalone habitat at Tanner Bank, 4 mi.^{2} at Cortes Bank, and 3 mi.^{2} on the western side of SCI (Butler et al., 2006a).

The white abalone diet varies with life history stage. As larvae, white abalone do not actively feed while in the planktonic stage. After settling on suitable substrate, abalone clamp tightly to rocky substrates and feed on algal matter scraped from the rocks or trapped under their shells. Young juveniles feed on bottom-dwelling diatoms, bacterial films, and benthic microflora. As they increase in size and become less vulnerable to predation, abalone leave their sheltered habitat to forage. Adult white abalone feed primarily on drifting fragments and attached macroalgae (National Marine Fisheries Service &

Southwest Regional Office, 2008b). Predators of white abalone include sea otters, fish, sea stars, crabs, spiny lobsters, and octopuses, as well as humans through illegal harvesting (Hobday & Tegner, 2000).

C.3.2.2.3 Population Trends

White abalone were once abundant throughout their range but were more common and abundant along the coast in the northern and southern portions. Since the 1970s, the white abalone population has experienced a 99 percent reduction in density (National Marine Fisheries Service & Southwest Regional Office, 2008b), with an annual decline in the population estimated at 12 percent (Catton et al., 2016). An increase in the size distribution over this same time period suggests individuals in the white abalone population are growing larger (which indicates increased age) with little or no indication of adequate recruitment success (Catton et al., 2016). With a dispersed population of aging individuals, prospects for reproduction, and therefore recruitment, remain low without management intervention, such as outplanting of healthy, captive-bred white abalone in suitable habitat (National Marine Fisheries Service, 2018e; Stierhoff et al., 2012). Captive breeding programs are currently in place to develop white abalone for introduction into the ocean (National Oceanic and Atmospheric Administration, 2018b; University of California Davis, 2017).

Various researchers have conducted submersible surveys off Tanner and Cortes Banks to map abalone habitat structure, examine distribution, and estimate the population size (Butler et al., 2006a; Davis et al., 1998a; Hobday & Tegner, 2000). They recorded 258 animals, with 168 recorded on Tanner Bank in 2002, at depths ranging from 32 to 55 m. In 2004, 35 individuals were recorded at Tanner Bank, 12 at Cortes Bank, and 5 off SCI. The 2012 population estimate of 564 individuals at SCI represented a moderate increase from the estimate of 353 individuals in 2005 (Stierhoff et al., 2014a).

C.3.2.2.4 Population Threats

White abalone face similar threats to those of the black abalone (i.e., historical overharvesting, current low population densities, withering syndrome, competition with urchins and other abalone species for food, and illegal harvest). Low population density and illegal harvest are considered the primary current threats (National Marine Fisheries Service & Southwest Regional Office, 2008b). However, because of the small population of white abalone, impacts on the remaining population are magnified.

C.3.2.3 Sunflower Sea Star (*Pycnopodia helianthoides*)

C.3.2.3.1 Status and Management

The sunflower sea star (*Pycnopodia helianthoides*) is proposed for listing (as Threatened) under the ESA (88 FR 21600).

C.3.2.3.2 Habitat and Geographic Range

Sunflower sea stars are found throughout the Northeastern Pacific Ocean, from the Aleutian Islands to Baja California (Sakashita, 2021). The large geographic range of sunflower sea star suggests this species is well adapted for a wide variety of environmental conditions and habitat types (Lowry et al., 2022), including kelp forests and rocky intertidal habitats. Sunflower sea stars also occupy a wide range of benthic substrates, including mud, sand, shell, gravel, and rocky bottoms (Konar et al., 2019; Lambert, 2000). They inhabit the low intertidal and subtidal zones to a depth of 435 m (1,427 ft) but are most common at depths less than 25 m (82 ft.) and rare in waters deeper than 120 m (394 ft) (Gravem et al., 2021; Hemery et al., 2016; Lambert, 2000).

The sunflower sea star is among the largest sea stars in the world, reaching over 1 m (3 ft.) in total diameter from ray tip to ray tip across the central disk (88 FR 16212). Very few predators are known to consume adult sunflower sea stars (Lowry et al., 2022). Predation risk is likely highest during the planktonic larval phase when indiscriminate filter feeders consume small larvae and selective pickers target larger, more developed individuals. Additionally, while the fecundity of the sunflower sea star is not well known, even conservative estimates suggest that an individual female likely produces millions of eggs in a single spawning event. As such, predation is not likely to substantially contribute to extinction risk, now or in the foreseeable future (Lowry et al., 2022).

The diet of adult sunflower sea stars generally consists of benthic and mobile epibenthic invertebrates, including sea urchins, snails, crab, sea cucumbers, and other sea stars (Mauzey et al., 1968; Shivji et al., 1983), and appears to be driven largely by prey availability.

C.3.2.3.3 Population Trends

The global abundance of sunflower sea star prior to 2013 was estimated at several billion animals, but from 2013 to 2017, sea star wasting syndrome (SSWS) reached pandemic levels, killing an estimated 90 percent of the population (Hamilton et al., 2021; Heady et al., 2022; Lowry et al., 2022). While population impacts varied by region across the range of the species, SSWS generally progressed from south to north and by 2017, sunflower sea stars became rare south of Cape Flattery (Washington), in areas where it had long been a conspicuous and ecologically important component of benthic marine ecosystems (Lowry et al., 2022). Most authors cited above comment that sunflower sea star is functionally extinct (greater than 99.2 percent) in its southern range from Baja Mexico to Washington state.

Within southern California, sunflower sea stars were frequently seen historically at low numbers at all sites during kelp forest monitoring surveys at SNI within the California portion of the HCTT Study Area (Kenner & Tomoleoni, 2021). In 2014, sunflower sea stars disappeared from all kelp forest monitoring sites due to SSWS and have not been observed since (Kenner & Tomoleoni, 2021). In addition, no sunflower sea stars were observed in the 2012–2013 subtidal surveys off SCI (Pondella et al., 2015). During 18 surveys at SCI from 2014 to 2022, no sunflower sea stars were sighted (Navy, unpublished data).

C.3.2.3.4 Population Threats

Species-specific threats to the sunflower sea star largely come from habitat destruction and modification in nearshore areas, overutilization from commercial, artisanal, and recreational fisheries (including bycatch), trade, and disease (Lowry et al., 2022). Disease, specifically SSWS, is the single greatest threat affecting the persistence of *P. helianthoides* now and into the foreseeable future (Lowry et al., 2022).

C.3.2.4 Globiceps Coral (*Acropora globiceps*)

C.3.2.4.1 Status and Management

Acropora globiceps was listed as threatened under the ESA in 2014 (79 FR 53852) and critical habitat was proposed for this species on November 30, 2023 (88 FR 83644).

C.3.2.4.2 Habitat and Geographic Range

A. globiceps has a relatively broad distribution, occurring in 39 marine ecosystems throughout the world (National Marine Fisheries Service, 2024b). In the U.S., this species occurs in Guam, the Commonwealth of the Northern Mariana Islands (CNMI), American Samoa, the Pacific Remote Island Area, and the Northwestern Hawaiian Islands.

C.3.2.4.3 Population Trends

Current information indicates that *A. globiceps* has a rangewide relative abundance of uncommon to common and current information suggests that abundances of this species is decreasing across all the ecoregions (National Marine Fisheries Service, 2024b).

C.3.2.4.4 Predator and Prey Interactions

The only known predator on *A. globiceps* is the crown-of-thorns seastar (79 FR 53852).

C.3.2.4.5 Species-Specific Threats

The threats that contributed to the listing of *A. globiceps* include ocean warming, ocean acidification, disease, fishing, predation, and inadequacy of existing regulatory mechanisms (79 FR 53851). In addition, current information indicates that collection and trade is also impacting the status of the species (National Marine Fisheries Service, 2024b).

C.3.3 Species Not Listed under the Endangered Species Act

C.3.3.1 Foraminifera, Radiolarians, Ciliates (Kingdom Protozoa)

Foraminifera, radiolarians, and ciliates are miniscule singled-celled organisms, sometimes forming colonies of cells, belonging to the kingdom Protozoa (Appeltans et al., 2010; Castro & Huber, 2000b). They are found in the water column and on the bottom of the world's oceans, and while most are microscopic, some species grow to approximately 20 cm (Hayward et al., 2016). In general, the distribution of foraminifera, radiolarians, and ciliates is patchy, occurring in regions with favorable growth conditions.

Foraminifera such as the genus *Globergerina* occur in the waters of the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems. Foraminifera form diverse and intricate shells out of calcium carbonate, organic compounds, or sand or other particles cemented together (University of California Berkeley, 2010c). The shells of foraminifera that live in the water column eventually sink to the bottom, forming soft bottom sediments known as foraminiferan ooze. Foraminifera feed on diatoms and other small organisms. Their predators include copepods and other zooplankton.

Radiolarians are microscopic zooplankton that form shells made of silica. Radiolarian ooze covers large areas of soft bottom habitat on the ocean floor (Pearse et al., 1987; University of California Berkeley, 2010e). Many radiolarian species contain symbiotic dinoflagellates (a type of single-celled organism) or algae. Radiolarians may also trap small particles or other organisms (e.g., diatoms) that drift in the water column.

Ciliates are protozoans with small hair-like extensions that are used for feeding and movement. They are a critical food source for primary consumers and are considered important parasites of many marine invertebrates. Ciliates feed on bacteria and algae, and some species contain symbiotic algae.

C.3.3.2 Sponges (Phylum Porifera)

Sponges include approximately 8,550 marine species worldwide and are classified in the Phylum Porifera (Van Soest et al., 2012; World Register of Marine Species Editorial Board, 2015). Sponges are bottom-dwelling, multicellular animals that can be best described as an aggregation of cells that perform different functions. Sponges are largely sessile, and are common throughout the Study Area at all depths. Sponges are typically found on mixed bottoms (unconsolidated substrate that is mostly gravel or cobble-sized) to hard bottoms, artificial structures, and biotic reefs. Sponges reproduce both sexually and asexually. Water flow through the sponge provides food and oxygen, and removes wastes (Pearse et al., 1987; University of California Berkeley, 2010a). This filtering process is an important coupler of processes and ecological function that occur in the water column and on the bottom (Pawlik & McMurray, 2020; Perea-Blázquez et al., 2012). Many sponges form calcium carbonate or silica spicules or bodies embedded in cells to provide structural support (Castro & Huber, 2000b; Van Soest et al., 2012). Sponges provide homes for a variety of animals including shrimp, crabs, barnacles, worms, brittle stars, sea cucumbers, and other sponges (Colin & Arneson, 1995b). Common native species in the Insular Pacific-Hawaiian Large Marine Ecosystem include lobate sponge (*Suberites zeteki*) and *Spongia oceania* (De Laubenfels, 1950, 1951), although some introduced species have become widespread as well. Sponges in the genera *Farrea*, *Hyalonema*, and *Suberites* occur in the waters of the California Current Large Marine Ecosystem (Clarke et al., 2015). Some sponge species are harvested commercially.

C.3.3.3 Corals, Hydroids, Jellyfish (Phylum Cnidaria)

There are over 10,000 marine species within the phylum Cnidaria worldwide (World Register of Marine Species Editorial Board, 2015), although there is taxonomic uncertainty within some groups (Veron, 2013). Cnidarians are organized into four classes: Anthozoa (corals, sea anemones, sea pens, sea pansies), Hydrozoa (hydroids and hydromedusae), Scyphozoa (true jellyfish), and Cubozoa (box jellyfish, sea wasps). Individuals are characterized by a simple digestive cavity with an exterior mouth surrounded by tentacles. Microscopic stinging capsules known as nematocysts are present (especially in the tentacles) in all cnidarians and are a defining characteristic of the phylum. The majority of species are carnivores that eat zooplankton, small invertebrates, and fishes. However, many species feed on plankton and dissolved organic matter, or contain symbiotic dinoflagellate algae (zooxanthellae) that produce nutrients by photosynthesis (Brusca & Brusca, 2003b; Dubinsky & Berman-Frank, 2001; Lough & van Oppen, 2009; National Oceanic and Atmospheric Administration & NOAA's Coral Reef Conservation Program, 2016). Representative predators of cnidarians include sea slugs, snails, crabs, sea stars, coral- and jellyfish-eating fish, and marine turtles. Cnidarians may be solitary or may form colonies.

Cnidarians have many diverse body shapes, but may generally be categorized as one of two basic forms: polyp and medusa. The polyp form is tubular and sessile, attached at one end with the mouth surrounded by tentacles at the free end. Corals and sea anemones are examples of the polyp form. The medusa form is bell- or umbrella-shaped (e.g., jellyfish), with tentacles typically around the rim. The medusa form generally is pelagic, although there are exceptions. Many species alternate between these two forms during their life cycle. All cnidarian species are capable of sexual reproduction, and many cnidarians also reproduce asexually. The free-swimming larval stage is usually planktonic, but is benthic in some species.

A wide variety of cnidarian species occur throughout the Study Area at all depths and in most habitats, including hard and mixed shores; soft, mixed, and hard bottom; aquatic vegetation beds; and artificial substrates. Some cnidarians form biotic habitats that harbor other animals and influence ecological processes, the primary examples being shallow-water and deep-water stony corals. In this section, corals are discussed in terms of individual coral polyps or early life stages, where "coral" is defined as follows: Species of the phylum Cnidaria, including all species of the orders Antipatharia (black corals), Scleractinia (stony corals), Gorgonacea (horny corals), Stolonifera (organ pipe corals and others), Alcyonacea (soft corals), and Helioporacea (blue coral) of the class Anthozoa; and all species of the families Milleporidea (fire corals) and Stylastreridae (*stylasterid hydrocorals*) of the class Hydrozoa.

Precious corals are non-reef building and inhabit depth zones below the euphotic zone. They are found on solid substrate in areas that are swept relatively clean by moderate-to-strong (greater than 25 cm/second) bottom currents. Precious corals may be divided into deep- and shallow-water species. Deep-water precious corals are generally found between 350 and 1,500 m and include pink coral (*Corallium secundum*), gold coral (*Gerardia* spp. and *Parazoanthus* spp.), and bamboo coral (*Lepidisis olapa*). Shallow water species occur between 30 and 100 m and consist primarily of three species of black coral: *Antipathes dichotoma*, *Antipathes grandis*, and *Antipathes ulex*.

Corals occur throughout the Hawaiian Archipelago. Approximately 250 species of corals are found in the region, including 59 scleractinian stony corals, 137 species of octocorals, 14 genera of black coral, 12 species of soft coral, and 4 species of stylasterid hydrocorals (Maragos et al., 2004). Dominant coral species in the Main Hawaiian Islands include *M. capitata*, *M. flabellata*, *M. patula*, *P. meandrina*, *P. compressa*, *P. lobata*, and *P. varians* (Franklin et al., 2013; Friedlander et al., 2008b). Common scleractinian corals of mesophotic reefs (Spalding et al., 2019) include several species of the genus *Leptoseris* (Kahng & Maragos, 2006). Coral coverage is generally highest in the southern portion of the archipelago (Friedlander et al., 2008b). However, more species of stony corals have been documented in the Northwestern Hawaiian Islands (57) than in the Main Hawaiian Islands (50) (Friedlander et al., 2008a; Friedlander et al., 2008b; Jokiel, 2008).

Although corals in temperate waters are not reef-building, the corals provide vertical relief and habitat that supports many organisms. For example, a single dead colony of Christmas tree black coral (*Antipathes dendrochristos*) observed by submersible off southern California was colonized by over 2,500 individual invertebrates, including other cnidarians (sea anemones and corals), crustaceans, echinoderms, molluscs, and polychaete worms (Love et al., 2007). Surveys using trawls, submersibles, and remotely operated vehicles conducted on outer continental shelf bank and rock outcrops off southern California have documented numerous coral species, including scleractinian stony corals, antipatharian black corals, gorgonian octocorals (sea fans), alcyonacean soft corals, pennatulacean octocorals (sea pens), and stylasterine hydrocoral (Etnoyer & Morgan, 2003; Whitmire & Clarke, 2007; Yoklavich et al., 2013).

Corals that are associated with tropical shallow reefs and temperate rocky habitats are vulnerable to a range of threats, including fishing impacts, pollution, erosion/sedimentation, coral harvesting, vessel damage, temperature increase, and climate change. Fishing practices such as blast fishing and trapping may be particularly destructive to coral reefs. In addition, removal of herbivorous fishes may result in overgrowth of coral reefs by algae (DeMartini & Smith, 2015). Because corals are slow growing and can survive for hundreds of years (Love et al., 2007; Roberts & Hirshfield, 2003), recovery from damage could take many years. Corals that occur in association with shallow-water coral reefs are protected by Executive Order [EO] 13089, *Coral Reef Protection*, and managed by the Coral Reef Task Force (EO 13089, *Coral Reef Protection*, 63 *FR* 32701–32703 [June 16, 1998]). The Navy is the DoD representative to the U.S. Coral Reef Task Force and also carries out the Coral Reef Protection Implementation Plan (Lobel & Lobel, 2000).

Deep-water corals are azooxanthellate (lack symbiotic algae) and thus do not form consolidated biogenic substrate, but rather form mounds of mixed substrate over hard bottom areas. Deep-water coral taxa in the Study Area consist primarily of hexacorals (stony corals, black corals, and gold corals), octocorals (e.g., true soft corals, gorgonians, sea pens), and hydrocorals (e.g., lace corals) (Hourigan et al., 2017a). Deep-water corals are widely distributed throughout the U.S. Pacific Island region, including the Hawaiian Archipelago (Parrish et al., 2015b; Parrish et al., 2022). In general, deep corals in the

Hawaii region do not form the extensive three-dimensional reef structures observed in the Atlantic and South Pacific. Octocorals and antipatharians (black corals) have been found in high densities at numerous sites, particularly on topographically high areas. Deep-sea coral communities are prevalent throughout the entire Hawaiian Archipelago (Etnoyer & Morgan, 2003) and have been found at all depths investigated (maximum of about 1,800 m) where suitable substrate exists (Baco, 2007). Approximately 200 species of deep corals (octocorals, antipatharians, and zoanthids) have been found in the Hawaiian Archipelago region (Parrish & Baco, 2007; Parrish et al., 2015a). Precious corals, black corals, and various octocoral species appear to be the most numerous deep-water corals at depths less than about 600 m, while octocorals dominate below 600 m (Parrish et al., 2015a). Study results indicate that stony corals are relatively rare at all depths and that most species are solitary (non-colonial). Gorgonians are the most common group of deep-sea corals in the Hawaiian Islands.

Most of the habitat-forming deep-sea corals in the California portion of the Study Area are anthozoans and hydrozoans (Etnoyer & Morgan, 2003; Etnoyer & Morgan, 2005). Deep-water corals have been documented throughout the SOCAL Bight (generally considered to be the area between Point Conception and San Diego, California), although the corals appear to be more restricted in the region near San Diego. Deep-water areas off the California coast, including the Channel Islands National Marine Sanctuary, support numerous corals such as sea fans (gorgonians), *Lophelia pertusa*, scleractinians such as the cup coral *Caryophyllia arnoldi*, and black corals (National Oceanic and Atmospheric Administration Fisheries & Southwest Fisheries Science Center, 2010; Whitmire & Clarke, 2007). At least 26 taxa of deep corals were recorded at a site within the Channel Islands sanctuary (Clarke et al., 2015). Large populations of hydrocorals occur at Tanner, Cortes, and Farnsworth Banks, offshore of southern California (Southern California Marine Institute, 2016). Much of the rocky area of Farnsworth Bank to depths of 66 m was found to be covered by the hydrocoral *Stylaster californicus* (Clarke et al., 2015). Surveys of a rocky bank south of Anacapa Island (depths of 97 to 314 m) found gorgonians and the black coral *A. dendrochristos* to be relatively abundant. Additional surveys of a nearby bank at depths of 275 to 900 m documented numerous corals, primarily including *A. dendrochristos*, the soft mushroom coral *Heteropolypus ritteri*, several sea fan species, *L. pertusa*, the cup coral *Desmophyllum dianthus*, and the sea pen *Halipteris californica* (on soft sediment only). Numerous species, including gold coral species, have been documented during various other surveys of banks off southern California.

The greatest threat to deep-water coral is physical strike and disturbance resulting from human activities. Deep corals are susceptible to physical disturbance due to the branching and fragile growth form of some species, slow growth rate (colonies can be hundreds of years old), and low reproduction and recruitment rates. Fishing activities, particularly trawling, are the primary threats to deep corals (Boland et al., 2016; Hourigan et al., 2017b; Packer et al., 2017; Rooper et al., 2016; Yoklavich et al., 2017). Marine debris is also a potential threat. For example, during one study in the Atlantic Ocean, a fishing trap, fishing line, balloon remnants, and ribbon was observed either lying on or wrapped around deep-sea corals located off the northeastern U.S. (Quattrini et al., 2015). Other potential human-caused threats to deep-water corals include coral harvesting (e.g., black corals), hydrocarbon exploration and extraction, cable and pipeline installation, and other bottom-disturbing activities (Boland et al., 2016; Clarke et al., 2015; Parrish et al., 2015a). Natural threats consist of sedimentation and bioerosion of the substrate.

C.3.3.4 Flatworms (Phylum Platyhelminthes)

Flatworms include between 12,000 and 20,000 marine species worldwide (World Register of Marine Species Editorial Board, 2015) and are the simplest form of marine worm (Castro & Huber, 2000b). The largest single group of flatworms are parasites commonly found in fishes, seabirds, and marine mammals (Castro & Huber, 2000b; University of California Berkeley, 2010d). The life history of parasitic flatworms plays a role in the regulation of populations of the marine vertebrates they inhabit. Ingestion by the host organism is the primary dispersal method for parasitic flatworms. Parasitic forms are not typically found in the water column outside of a host organism. The remaining groups are non-parasitic carnivores, living without a host. Flatworms are found throughout the Study Area living on rocks in tide pools and reefs, or within the top layer of sandy areas. Dominant genera of flatworms in the Insular Pacific-Hawaiian Large Marine Ecosystem include *Pseudobiceros* and *Pseudoceros*. Flatworms in the genera *Waminoa* and *Freemania* occur in the waters around the California Current Large Marine Ecosystems. Several species of wrasses and other reef fish prey on flatworms (Castro & Huber, 2000a, 2000b).

C.3.3.5 Ribbon Worms (Phylum Nemertea)

Ribbon worms include over 1,300 marine species worldwide (World Register of Marine Species Editorial Board, 2015). Ribbon worms, with their distinct gut and mouth parts, are more complex than flatworms (Castro & Huber, 2000b). A unique feature of ribbon worms is the extendable proboscis (an elongated, tubular mouth part), which can be ejected to capture prey, to aid in movement, or for defense (Brusca & Brusca, 2003b). Most ribbon worms are active, bottom-dwelling predators of small invertebrates such as annelid worms and crustaceans (Brusca & Brusca, 2003b; Castro & Huber, 2000b). Some are scavengers or symbiotic (parasites or commensals). Some ribbon worms are pelagic, with approximately 100 pelagic species identified from all oceans (Roe & Norenburg, 1999). Pelagic species generally drift or slowly swim by undulating the body. Ribbon worms exhibit a variety of reproductive strategies, including direct development with juveniles hatching from egg cases and indirect development from planktonic larvae (Brusca & Brusca, 2003b). In addition, many species are capable of asexual budding or regeneration from body fragments. Ribbon worms have a relatively small number of predators, including some birds, fishes, crabs, molluscs, squid, and other ribbon worms (McDermott, 2001). Ribbon worms are found throughout the Study Area. They occur in most marine environments, although usually in low abundances. They occur in embayments; soft, mixed, and rocky shores and subtidal habitats of coastal waters; and deep-sea habitats. Some are associated with biotic habitats such as mussel clumps, coral reefs, kelp holdfasts, seagrass beds, and worm burrows (Thiel & Kruse, 2001). Approximately 10 species of ribbon worms from the classes *Anopla* and *Enopla* are known from Hawaii (Hawaiisfishes, 2017), and a total of 64 species have been identified in intertidal habitats of California (Bernhardt, 1979).

C.3.3.6 Round Worms (Phylum Nematoda)

Round worms include over 7,000 marine species (World Register of Marine Species Editorial Board, 2015). Round worms are small and cylindrical, abundant in sediment habitats such as soft to mixed shores and soft to mixed bottoms, and also found in host organisms as parasites (Castro & Huber, 2000b). Round worms are some of the most widespread marine invertebrates, with population densities of up to 1 million or more organisms per square meter of sediment (Levinton, 2009a). This group has a variety of food preferences, including algae, small invertebrates, annelid worms, and organic material from sediment. Like parasitic flatworms, parasitic nematodes play a role in regulating populations of other marine organisms by causing illness or mortality. Species in the family Anisakidae infect marine fish, and may cause illness in humans if fish are consumed raw without proper precautions (Castro & Huber, 2000b). Round worms are found throughout the Study Area.

C.3.3.7 Segmented Worms (Phylum Annelida)

Segmented worms include approximately 14,000 currently accepted marine species worldwide in the phylum Annelida, although the number of potentially identified marine species is nearly 25,000 (World Register of Marine Species Editorial Board, 2015). Most marine annelids are in the class Polychaeta. Polychaetes are the most complex group of marine worms, with a well-developed respiratory and gastrointestinal system (Castro & Huber, 2000b). Different species of segmented worms may be highly mobile or burrow in the bottom (soft to mixed shore or bottom habitats) (Castro & Huber, 2000b). Polychaete worms exhibit a variety of life styles and feeding strategies, and may be predators, scavengers, deposit-feeders, filter-feeders, or suspension feeders (Jumars et al., 2015). The variety of feeding strategies and close connection to the bottom make annelids an integral part of the marine food web (Levinton, 2009a). Burrowing and agitating the sediment increases the oxygen content of bottom sediments and makes important buried nutrients available to other organisms. This allows bacteria and other organisms, which are also an important part of the food web, to flourish on the bottom. Benthic polychaetes also vary in their mobility, including sessile attached or tube-dwelling worms, sediment burrowing worms, and mobile surface or subsurface worms. Some polychaetes are commensal or parasitic. Many polychaetes have planktonic larvae.

Polychaetes are found throughout the Study Area inhabiting rocky, sandy, and muddy areas of the bottom, vegetated habitats, and artificial substrates. Some are associated with biotic habitats such as mussel clumps, coral reefs, and worm burrows. Some species of worms build rigid (e.g., *Diopatra* spp.) or sand-encrusted (*Phragmatapoma* spp.) tubes, and aggregations of these tubes form a structural habitat. Giant tube worms (*Riftia pachyptila*) are chemosynthetic (using a primary production process without sunlight) reef-forming worms living on hydrothermal vents of the abyssal oceans. Their distribution is poorly known in the Study Area. A total of 20 taxa of annelid worms were documented at intertidal locations of Oahu, compared to 71 taxa in central California (Zabin et al., 2013).

C.3.3.8 Bryozoans (Phylum Bryozoa)

Bryozoans include approximately 6,000 marine species worldwide (World Register of Marine Species Editorial Board, 2015). They are small box-like, colony-forming animals that make up the “lace corals.” Colonies can be encrusting, branching, or free-living. Bryozoans may form habitat similar in complexity to sponges (Buhl-Mortensen et al., 2010). Bryozoans attach to a variety of surfaces, including mixed and hard bottom, artificial structures, and algae, and feed on particles suspended in the water (Hoover, 1998b; Pearse et al., 1987; University of California Berkeley, 2010b). Bryozoans are of economic importance for bioprospecting (the search for organisms for potential commercial use in pharmaceuticals). As common biofouling organisms, bryozoans also interfere with boat operations and clog industrial water intakes and conduits (Hoover, 1998b; Western Pacific Regional Fishery Management Council, 2001). Bryozoans occur throughout the Study Area but are not expected at depths beyond the continental slope (Ryland & Hayward, 1991). Habitat-forming species are most common on temperate continental shelves with relatively strong currents (Wood et al., 2012). Common species in the Insular Pacific-Hawaiian Large Marine Ecosystem are violet encrusting bryozoan (*Disporella violacea*) and lace bryozoan (*Reteporellina denticulata*). Species that occur in the California Current Large Marine Ecosystem include arborescent bryozoans of the genus *Bugula* and encrusting bryozoans of the genus *Schizoporella*.

C.3.3.9 Squid, Bivalves, Sea Snails, Chitons (Phylum Mollusca)

The phylum Mollusca includes approximately 45,000 marine species worldwide (World Register of Marine Species Editorial Board, 2015). These organisms occur throughout the Study Area, including open ocean areas, at all depths. Sea snails and slugs (gastropods), clams and mussels (bivalves), chitons (polyplacophorans), and octopus and squid (cephalopods) are examples of common molluscs in the Study Area. Snails and slugs occur in a variety of soft, mixed, hard, and biogenic habitats. Chitons are typically found on hard bottom and artificial structures from the intertidal to littoral zone but may also be found in deeper water and on substrates such as aquatic plants. Many molluscs possess a muscular organ called a foot, which is used for mobility. Many molluscs also secrete an external shell (Castro & Huber, 2000b), although some molluscs have an internal shell or no shell at all (National Oceanic and Atmospheric Administration Fisheries, 2015). Sea snails and slugs eat fleshy algae and a variety of invertebrates, including hydroids, sponges, sea urchins, worms, other snails, and small crustaceans, as well as detritus (Castro & Huber, 2000b; Colin & Arneson, 1995a; Hoover, 1998b). Clams, mussels, and other bivalves are filter feeders, ingesting suspended food particles (e.g., phytoplankton, detritus) (Castro & Huber, 2000b). Chitons, sea snails, and slugs use rasping tongues, known as radula, to scrape food (e.g., algae) off rocks or other hard surfaces (Castro & Huber, 2000b; Colin & Arneson, 1995a). Squid and octopus are active swimmers at all depths and use a beak to prey on a variety of organisms including fish, shrimp, and other invertebrates (Castro & Huber, 2000b; Hoover, 1998b; Western Pacific Regional Fishery Management Council, 2001). Octopuses mostly prey on fish, shrimp, eels, and crabs (Wood & Day, 2005).

Important commercial, ecological, and recreational species of molluscs in the Insular Pacific-Hawaiian Large Marine Ecosystem include: various species of squid, the endemic cuttlefish (*Euprymna scolopes*), bivalves (clams and mussels), and limpets (*Cellana exarata* and *Cellana sandwicensis*) (Western Pacific Regional Fishery Management Council, 2001). Important commercial, ecological, and recreational species of molluscs in the California Current Large Marine Ecosystem include multiple abalone species, California market squid (*Doryteuthis opalescens*) (Clark et al., 2005), keyhole limpet (*Megathura crenulata*), Kellet's whelk (*Kelletia kelletia*), various species of octopus, sea hare (*Aplysia* spp.), snails (*Lithopoma undosum*, *Tegula* spp.), and Pismo clam (*Tivela stultorum*). Only one species of abalone, the red abalone (*Haliotis rufescens*), is currently fished recreationally, north of San Francisco County. The abalone fishery is closed to all commercial fishing. Black abalone and white abalone are listed under the ESA, while the green abalone (*Haliotis fulgens*) and pink abalone (*Haliotis corrugata*) are designated as species of concern.

C.3.3.10 Shrimp, Crab, Lobster, Barnacles, Copepods (Phylum Arthropoda)

Shrimp, crabs, lobsters, barnacles, and copepods are animals with an exoskeleton, which is a skeleton on the outside of the body (Castro & Huber, 2000b), and are classified as crustaceans in the Phylum Arthropoda. The exoskeletons are made of a polymer called chitin, similar to cellulose in plants, to which the animals add other compounds to achieve flexibility or hardness. There are over 57,000 marine arthropod species, with about 53,000 of these belonging to the subphylum Crustacea (World Register of Marine Species Editorial Board, 2015). These organisms occur throughout the Study Area at all depths. Crustaceans may be carnivores, omnivores, predators, or scavengers, preying on molluscs (primarily gastropods), other crustaceans, echinoderms, small fishes, algae, and seagrass (Waikiki Aquarium, 2009a, 2009b, 2009c; Western Pacific Regional Fishery Management Council, 2009). Barnacles and some copepods are filter feeders, extracting algae and small organisms from the water (Levinton, 2009a). Copepods may also be parasitic, affecting most phyla of marine animals (Walter & Boxshall, 2017). As a group, arthropods occur in a wide variety of habitats. Shrimp, crabs, lobsters, and copepods may be associated with soft to hard substrates, artificial structures, and biogenic habitats. Barnacles inhabit hard and artificial substrates.

Important commercial, ecological, and recreational species of Crustacea in the Hawaii Study Area include several lobster species from the taxonomic groups Palinuridae (spiny lobsters) and Scyllaridae (slipper lobsters) (Western Pacific Regional Fishery Management Council, 2009). Lobsters occur primarily within the subtidal zone, although their range can extend slightly deeper. Most species occur throughout the tropical oceans of the world, while the endemic Hawaiian spiny lobster is found only in Hawaii and Johnston Atoll (Polovina et al., 1999). Important commercial, ecological, and recreational species of Crustacea in the California Current Large Marine Ecosystem include the spot shrimp (*Pandalus platyceros*), ridgeback rock shrimp (*Sicyonia ingentis*), rock crab (*Cancer* species), sheep crab (*Loxorhynchus grandis*), and California spiny lobster (Clark et al., 2005).

C.3.3.11 Sea Stars, Sea Urchins, Sea Cucumbers (Phylum Echinodermata)

Organisms in this phylum include over 7,000 marine species, such as sea stars, sea urchins, and sea cucumbers (World Register of Marine Species Editorial Board, 2015). Asteroids (e.g., sea stars), echinoids (e.g., sea urchins), holothuroids (e.g., sea cucumbers), ophiuroids (e.g., brittle stars and basket stars), and crinoids (e.g., feather stars and sea lilies) are symmetrical around the center axis of the body (Mah & Blake, 2012). Echinoderms occur at all depth ranges from the intertidal zone to the abyssal zone and are almost exclusively benthic, potentially found on all substrates and structures. Most echinoderms have separate sexes, but a few species of sea stars, sea cucumbers, and brittle stars have both male and female reproductive structures. Many species have external fertilization, releasing gametes into the water to produce planktonic larvae, but some brood their eggs and release free-swimming larvae (Mah & Blake, 2012; McMurray et al., 2012). Many echinoderms are either scavengers or predators on sessile organisms such as algae, stony corals, sponges, clams, and oysters. Some species, however, filter food particles from sand, mud, or water (Hoover, 1998a). Predators of echinoderms include a variety of fish species (e.g., triggerfish, eels, rays, sharks), crabs, shrimps, octopuses, birds, and other echinoderms (sea stars).

Echinoderms are found throughout the Study Area. Important commercial, ecological, and recreational species in the Insular Pacific-Hawaiian Large Marine Ecosystem include helmet urchin (*Colobocentrotus atratus*), burrowing sea urchin (*Echinometra mathaei*), sea cucumbers, and sea stars. The crown-of-thorns sea star (*Acanthaster planci*) is a carnivorous predator that feeds on coral polyps and can devastate coral reefs. In 1969, crown-of-thorns sea stars infested reefs off southern Molokai but did not cause extensive damage to living coral polyps of cauliflower coral (Gulko, 1998; Hoover, 1998b). Important commercial, ecological, and recreational species of echinoderms in the California Current Large Marine Ecosystem include California sea cucumbers (*Parastichopus californicus*), sea stars (*Pisaster* spp.), red sea urchin (*Strongylocentrotus franciscanus*), and purple sea urchin (*S. purpuratus*) (Clark et al., 2005). Beginning in 2013, large numbers of sea stars have died along the west coast of North America due to sea-star wasting disease (Hewson et al., 2014; Miner et al., 2018). The virus causing the disease has also been found in sea urchins and sea cucumbers, although mass die-offs have not been documented for these taxa.

C.4 Habitats

C.4.1 General Background

Abiotic marine habitats vary according to geographic location, underlying geology, hydrodynamics, atmospheric conditions, and suspended particles and associated biogenic features. Sediments may be derived from material eroded from land sources associated with coastal bluff erosion and sediment flows from creeks and rivers, which may create channels, tidal deltas, intertidal and subtidal flats, and shoals of unconsolidated material along the shorelines and estuaries. Sediments derived from volcanic rock are common in the Hawaiian Islands and occur in localized areas of southern California (i.e., SCI) within the Study Area. In the Hawaiian Islands, nearshore sediments also are derived from living sources (i.e., corals).

The influence of land-based nutrients on habitat type and sediment increases with proximity to streams, bays and harbors, and nearshore waters. In the open ocean, gyres, eddies, and oceanic currents influence the distribution of organisms. Major bottom features in the offshore areas of the range complexes include shelves, banks, breaks, slopes, canyons, plains, and seamounts. Geologic features such as these affect the hydrodynamics of the ocean water column (i.e., currents, gyres, upwellings) as well as living resources present.

C.4.1.1 Shore Habitats

C.4.1.1.1 Description

C.4.1.1.1.1 Soft Shores

Soft shores include all aquatic habitats that have three characteristics: (1) unconsolidated substrates with less than 25 percent areal cover of stones, boulders, or bedrock, (2) unconsolidated sediment composed of predominantly sand or mud, and (3) primarily intertidal water regimes (Cowardin et al., 1979). Note that a shoreline covered in vegetation (e.g., marsh) could still have a soft substrate foundation. Soft shores include beaches, tidal flats/deltas, and streambeds of the tidal riverine and estuarine systems.

Intermittent or intertidal channels of the riverine system and intertidal channels of the estuarine system are classified as streambed. Intertidal flats, also known as tidal flats or mudflats, consist of loose mud, silt, and fine sand with organic-mineral mixtures that are regularly exposed and flooded by the tides (Karleskint et al., 2006). Muddy and fine sediment tends to be deposited where wave energy is low, such as in sheltered bays and estuaries (Holland & Elmore, 2008). Mudflats are typically unvegetated, but may be covered with encrusting microscopic algae (e.g., diatoms) or sparsely vegetated with low-growing aquatic plants (e.g., macroalgae/seaweed, seagrass). Muddy intertidal habitat occurs most often as part of a patchwork of intertidal habitats that may include rocky shores, tidal creeks, sandy beaches, salt marshes, and mangroves. A flat area of unconsolidated sediment that is covered in aquatic plants could be considered an aquatic bed growing on soft shore habitat. While river deltas are created by soil deposits forming from the outflow of the water, such as at the mouth of the Mississippi River, tidal deltas are depositions of sediment left by the diurnal tides and their resulting currents. Therefore, tidal (or tide-dominated) deltas typically occur in locations of large tidal ranges or high tidal current speeds (SEPM Strata, 2018).

Beaches form through the interaction of waves and tides, as particles are sorted by size and are deposited along the shoreline (Karleskint et al., 2006). Wide flat beaches with fine-grained sands occur where wave energy is limited. Narrow steep beaches of coarser sand form where energy and tidal ranges are high (Speybroeck et al., 2008). Three zones characterize beach habitats: (1) dry areas above the MHW, (2) wrack lines (the area where seaweed and debris is deposited at high tide) and (3) a high-energy intertidal zone (area between high and low tide).

C.4.1.1.1.2 Mixed Shores

Mixed shores include all aquatic habitats with the following three characteristics: (1) substrates with at least 25 percent cover in particles smaller than stones, (2) unconsolidated substrate is predominantly gravel or cobble-sized, and (3) primarily intertidal water regimes. These areas may or may not be stable enough for attached vegetation or invertebrates, depending on overlying hydrology and water quality. Note that a shoreline with vegetation (e.g., macroalgae, seagrass) could still have a mixed substrate foundation. Hard corals may grow in these habitats in the Hawaiian Islands.

C.4.1.1.1.3 Hard Shores

Rocky shores include intertidal aquatic habitats characterized by bedrock, stones, and/or boulders that cover 75 percent or more of an area (Cowardin et al., 1979). Note that a shoreline covered in vegetation could still have a hard substrate foundation. Rocky intertidal shores are areas of bedrock occupying the area between high and low tide lines (Menge & Branch, 2001). Extensive rocky shorelines can be interspersed with sandy areas, estuaries, or river mouths.

Environmental gradients between hard shorelines and subtidal habitats are determined by wave action, depth, frequency of tidal inundation, and stability of substrate (Cowardin et al., 1979). Where wave energy is extreme, only rock outcrops may persist. In lower energy areas, a mixture of rock sizes will occur in the intertidal zone. Intertidal rocky shores provide substrate for attached macroalgae and sessile invertebrates.

C.4.1.1.2 Distribution

C.4.1.1.2.1 Soft Shores

Tidal flats occur on a variety of scales in virtually all estuaries and bays in the Hawaii and California Study Areas. In the Hawaiian portion of the Study Area, beaches are common along the lagoon reaches of atoll islets, along the coasts, and in embayment's of the main and Northwestern Hawaiian Islands. Significant sandy beach habitat occurs primarily on the western and southern sides of the islands (Maragos, 2000). About 82 percent of Southern California's coastline is sandy beach habitat (Allen & Pondella, 2006). The California Study Area has extensive beaches, although few stretches are undisturbed by human activity (U.S. Department of Commerce et al., 2008).

C.4.1.1.2.2 Mixed Shores

In the Hawaii Study Area, mixed intertidal habitat, including unconsolidated limestone and volcanic rock, occurs throughout the Hawaiian Islands in localized areas, typically near hard shorelines where physical conditions prevent sand from accumulating (Maragos, 2000). Mixed intertidal habitats occur on the Channel Islands and along the mainland within the Study Area. The majority of mixed shores occur in transitional areas between hard shores and soft shores. Mixed shorelines also may occur at beaches where hard substrate underlies sand and rocks become exposed during periods of shoreline erosion (e.g., several beaches in San Diego County).

C.4.1.1.2.3 Hard Shores

In the Hawaii Study Area, rocky intertidal habitat including limestone and volcanic rock occurs throughout the Hawaiian Islands in localized areas wherever physical conditions prevent sand from accumulating (Maragos, 2000). In the Southern California portion of the Study Area, rocky intertidal habitat is most extensive on the offshore Channel Islands. Hard shores are localized in distribution along the mainland of southern California. In numerous locations within the HCTT Study Area, artificial hard substrates (e.g., rock riprap, seawalls) have been placed to reduce storm damage and erosion along shorelines and in estuaries.

C.4.1.2 Artificial Structures

C.4.1.2.1 Distribution

Offshore artificial structures in the Hawaii Study Area include shipwrecks, sunken military vessels and aircraft, and artificial reefs. Shipwrecks located near the Island of Hawaii are concentrated along its northwestern coast and within Hilo Bay. Well-documented examples of the numerous submerged structures in the waters surrounding Oahu include the largely intact Sea Tiger, a World War II-era Japanese midget submarine; *Mahi*, a Navy minesweeper/cable layer scuttled off the Waianae Coast; and the YO-257, a Navy yard oiler built in the 1940s that was intentionally sunk off Waikiki in 1989 to create an artificial reef. Major sunken vessels in Pearl Harbor include the USS ARIZONA, the USS UTAH, and the USS BOWFIN, which are listed in the National Register of Historic Places. There may be as many as 60 vessels known lost among the atolls and at least 67 naval aircraft sunk in the Northwestern Hawaiian Islands (National Oceanic and Atmospheric Administration, 2017b). At least 14 ships have run aground in the Northwestern Hawaiian Islands since 1957 (Friedlander et al., 2009).

Most artificial structures in the California Study Area include shipwrecks and artificial reefs. A prominent artificial structure area offshore San Diego County, known as Wreck Alley, includes several types of structures, including six vessels (*El Rey, Ruby E., Shooter's Fantasy, Strider, Yukon,* and a barge), a P-38 aircraft, old Naval Ocean Systems Center tower, and dumped bridge and roadway materials (California Wreck Divers, 2017). The largest artificial reef in the Study Area was built offshore San Clemente; the Wheeler North Artificial Reef mostly consists of boulder-sized quarry rock deposited in a module design that covers a 174-acre area (Reed et al., 2010). Most artificial reefs in marine waters have been placed and monitored by individual state programs.

C.5 Fishes

C.5.1 General Background

Marine fishes can be broadly categorized by their distributions within the water column or habitat usage. Moyle and Cech (2004) define the major marine habitat categories as estuaries, coastal habitats, reefs, the epipelagic zone, the deep sea, and the Polar regions. In the Study Area, the major habitat categories include all the aforementioned except the Polar regions. Many marine fishes that occur in the Study Area are either demersal species (i.e., close to the seafloor) associated with nearshore coastal reefs, or are more oceanic and live in surface waters (pelagic) further offshore (Schwartz, 1989). The highest number and diversity of fishes typically occur where the habitat has structural complexity (reef systems, continental slopes, deep canyons), biological productivity (areas of nutrient upwelling), and a variety of physical and chemical conditions (water flow, nutrients, DO, and temperature) (Bergstad et al., 2008; Helfman et al., 2009; Moyle & Cech, 2004; Parin, 1984; Pyle et al., 2019). Some of the marine fishes that occur in the coastal zone migrate between marine and freshwater habitats (Helfman et al., 2009). Other distribution factors, including predator/prey relationships, water quality, and refuge (e.g., physical structure or vegetation cover) operate, on more regional or local spatial scales (Reshetiloff, 2004). Also, fishes may move among habitats throughout their lives based on changing needs during different life stages (Schwartz, 1989).

Some fish species in the U.S. are protected under the ESA and are managed by either the U.S. Fish and Wildlife Service (USFWS) or NMFS. The recreational and commercial fisheries are managed within a framework of overlapping international, federal, state, interstate, and tribal authorities. Individual states and territories generally have jurisdiction over managed fisheries located in marine waters within 3 NM of their coast. Federal jurisdiction includes managed fisheries in marine waters inside the U.S. EEZ. The area stretches from the outer boundary of state waters out to 200 NM offshore of any U.S. coastline, except where intersected closer than 200 NM by bordering countries.

The Magnuson-Stevens Fishery Conservation and Management Act and Sustainable Fisheries Act led to the formation of eight regional fishery management councils that coordinate with NMFS to manage and conserve certain fisheries in federal waters. Together with NMFS, the councils maintain fishery management plans for species or species groups comprised of fish, invertebrates, and vegetation to regulate commercial and recreational harvest within their geographic regions. The Study Area overlaps with the jurisdiction of two regional fishery management councils, as well as the range of the highly migratory species (e.g., sharks, billfishes, swordfish, and tunas), which are managed directly by NMFS.

- **The Western Pacific Regional Fishery Management Council** includes Hawaii, American Samoa, Guam, and the Northern Mariana Islands.
- **The Pacific Fishery Management Council** includes Washington, Oregon, and California.

- **NMFS, Office of Sustainable Fisheries** includes all federally managed waters of the U.S. where highly migratory species occur.

C.5.1.1 Habitat Use

Fishes inhabit most of the world's oceans, from warm shallow coastal habitat to cold deep-sea waters, and are found on the surface, in the water column, and at the bottom in the Study Area. The description of habitat use in this section pertains to common fishes found in the different habitats.

Fish distribution is restricted by biotic factors (competition or predation) or by abiotic components, such as temperature, salinity, DO, and pH. A species can be excluded from a suitable habitat by competitors, predators, parasites, or a lack of available prey (Moyle & Cech, 2004). For example, Catano et al. (2015) found that a loss of corals and the resulting decline in structural complexity, as well as management efforts to protect reefs, could alter the territory dynamics and reproductive potential of important herbivorous fish species.

Marine and diadromous fishes inhabit the diverse coastal habitats on or near the edges of the continents, from the intertidal regions to the edge of the continental shelf (Moyle & Cech, 2004). The most abundant and conspicuous types of coastal habitats are hard bottom (e.g., rocky reefs which can include shell beds), soft bottom (e.g., sand, mud, silt), submerged aquatic vegetation (e.g., mangroves, salt marshes, seagrass beds, macroalgae beds), and floating macroalgae. Each of these coastal habitats has distinct types of fishes associated with it. Common fishes inhabiting hard bottom habitats in the Study Area include gobies (Gobiidae), rockfishes (Scorpaenidae), and sculpins (Cottidae), while flounder (Bothidae) and stingrays (Dasyatidae) are found on soft bottoms. Pipefishes (Syngnathidae) and kelpfish (Clinidae) are common inhabitants of submerged aquatic vegetation habitat. Species commonly found under offshore floating macroalgae include ocean sunfishes, tunas, sharks, and mahi mahi.

Somewhere between 30 and 40 percent of all fish species are associated with hard bottom habitats (tropical and subtropical) such as reefs, and anywhere from 250 to 2,200 species are likely to be found in, on, or near a major complex of reefs. Coral reef habitats are found between latitudes 30° North (N) and 30° South (S) in shallow water (usually less than 164 ft.) that is warm enough to support the growth of corals and clear enough to allow photosynthesis at moderate depths. Most reef habitats are surrounded by nutrient-poor oceanic waters. Compared to the total number of species of carnivorous fishes that inhabit low-latitude coral reefs, the number of herbivores is small (20 percent), but they are often the most noticeable fishes (Moyle & Cech, 2004). Damselfishes (Pomacentridae), parrotfishes (Scaridae), and surgeonfishes (Acanthuridae) are examples of herbivorous fishes found in reef habitat (Moyle & Cech, 2004). In the Study Area, commonly recognized reef fishes include butterflyfishes (Chaetodontidae), puffers (Tetraodontidae), tangs (Acanthuridae), triggerfishes (Balistidae), and wrasses (Labridae).

The upper 200 m of the ocean is known as the photic or epipelagic zone. Sunlight penetrates sufficiently to support the growth of phytoplankton or macro algae. The area between 200 m and 1,000 m is referred to as the mesopelagic zone, where light penetration is minimal. Below the mesopelagic zone is the bathypelagic or aphotic zone, where sunlight does not penetrate. A lack of habitat complexity can limit the number of fish species that inhabit the epipelagic zone. Less than two percent of all fish species inhabit the nutrient-poor waters, with most occurring in the upper 100 m of the water column, where light can penetrate and permit phytoplankton growth and visual predators to see their prey. Epipelagic fishes are divided for convenience into nearshore and oceanic groups. Nearshore epipelagic fishes are overall the most commercially valuable group of fishes to humans because they typically occur in large schools, such as herring (Clupeidae) and anchovies (Engraulidae), or are particularly favored as food, such as tunas (Scombridae) and salmon (Salmonidae). Predators on nearshore epipelagic fishes include billfishes and swordfishes (Xiphiidae), sharks (Carcharhinidae), and others. Epipelagic fishes that inhabit the open ocean spend their entire life cycle either free swimming or associated with drifting seaweed e.g., kelp (Moyle & Cech, 2004). In the Study Area, examples of epipelagic open ocean fishes include sharks, tunas, sauries (Scomberesocidae), and ocean sunfish (Molidae).

Mesopelagic habitats are found below the well-lighted, well-mixed epipelagic zone. Between about 120 m and 1,000 m in depth, light gradually fades to extinction, and the water temperatures decreases to 39°F. Below 1,000 m, bathypelagic habitats are characterized by complete darkness, low temperatures, low nutrients, low DO, and great pressure. This environment is the most extensive aquatic habitat on earth. The vastness of the deep-sea habitat, coupled with its probable stability through geological time, has led to the development of a diverse fish community, which accounts for 11 percent of all recorded fish species in the oceans. Lanternfishes (Myctophidae), with about 240 species, are an important group of mesopelagic deep sea fishes in terms of diversity, distribution, and numbers of individuals (Helfman et al., 2009). These species make up a large fraction of the deep scattering layer, so-called because the sonic pulses of a sonar can reflect off the millions of swim bladders, often giving the impression of a false bottom (Moyle & Cech, 2004). Generally, deep-sea fishes are divided into two groups, those that are found in the water column and others associated with the seafloor. In the Study Area, the cookie cutter shark (Dalatiidae), fangtooths (Anoplogastridae), hatchetfishes (Sternoptychidae), and lanternfishes inhabit the water column while the seafloor is inhabited with grenadiers or rattails (Macrouridae), hagfishes (Myxinidae), rays (Rajidae), and some rockfishes (Sebastidae).

Some fishes use one habitat type over their entire life cycle, while others associate with different habitat types by life stage. Anadromous fishes, such as Pacific salmon, hatch and rear in freshwater rivers as fry, with early juveniles inhabiting estuaries for short periods as they transition into late-juvenile life stages before entering the ocean to mature into adults. Many other marine fishes inhabit the water column as larvae, settling onto soft bottom habitat as juveniles and remaining there as adults (e.g., flatfishes). The reef-associated Pacific seahorse (*Hippocampus ingens*) and oceanic Pacific bluefin tuna (*Thunnus orientalis*) provide examples of species closely connected to one habitat category across their life cycle.

C.5.1.2 Movement and Behavior

Fishes exhibit a rich array of sophisticated behavior (Meyer et al., 2010). Fishes have been shown to cooperate in a variety of ways during foraging, navigation, reproduction, and predator avoidance (Fitzpatrick et al., 2006; Huntingford et al., 2006; Johnstone & Bshary, 2004). Some examples of the common types of behavior exhibited by fishes include movement or migration, schooling, feeding, and resting (Moyle & Cech, 2004).

Migratory behavior consists of mass movements from one place to another and can range in occurrence from daily to seasonal, depending on the species. Tunas, salmonids, and eels migrate thousands of miles in short periods of time (e.g., a few months). Daily or seasonal migrations are typically for feeding and/or predator avoidance and can also be referred to as movement patterns. Some common movement patterns include coastal migrations, open ocean migrations, onshore/offshore movements, vertical water column movements, and life stage-related migrations (e.g., eggs and larvae as part of the plankton/nekton). Migratory behavior occurs in response to changing environmental conditions, particularly temperature, or the movement and abundance of food organisms. The destinations of migratory events are often feeding or reproductive grounds. Many fishes have the ability to find their way back to a “home” area and some species use olfactory and visual cues, as well as chemicals released by the other fishes to return home. Highly migratory species such as hammerhead shark (*Sphyrna* species), albacore (*Thunnus alalunga*), wahoo (*Acanthocybium solandri*) and swordfish (*Xiphias gladius*), may move across thousands of miles of open ocean (Beamish et al., 2005; Duncan et al., 2006; Theisen et al., 2008). Other migratory species such as Pacific salmonids exhibit seasonal movement patterns throughout coastal continental shelf waters and beyond.

A shoal is defined as any group of fishes that remain together for social reasons, while a school is a polarized, synchronized shoal (Moyle & Cech, 2004), often swimming together in tight formations. Schools can change shape when traveling, feeding, resting, or avoiding predators. Vision and the lateral-line system play roles in assisting schooling by allowing fish to visually orientate to one another and also sense water movements when visibility is reduced. Schooling behavior may provide protection against predators. Schooling may also be beneficial in terms of reproduction since little energy has to be expended to find a mate when sexes school together (Moyle & Cech, 2004).

Feeding behavior of fishes is influenced by many factors, including characteristics of the environment, predators, and prey. When food is scarce, individual fish have been observed capturing prey items of all sizes, for which there is likely to be a net gain of energy for the fish. However, when food is abundant, a fish will typically seek the prey item that produces the most energy for the least amount of effort. The body shape of a fish species, specifically the mouth, reflects the general method of feeding. Many fishes must swallow their prey whole and have mouths specialized for their prey depending on the prey’s size and shape (Price et al., 2015). Fishes with their mouth on the underside of their body (e.g., sturgeon, rays, skates, etc.) are typically bottom feeders, while fishes with their mouths near the top of their head (e.g., mullets, halfbeaks, etc.) are typically surface feeders. Fishes that typically feed in the water column, which includes most species, have mouths that are centered in their head. Common types of feeding behavior include ambushing, drift feeding, and filter feeding; fishes may regularly switch between two or more modes of feeding behavior depending on the abundance of prey (Moyle & Cech, 2004).

C.5.1.3 Hearing and Vocalization

Refer to Appendix D (Acoustic and Explosive Effects Supporting Information) for a summary and details regarding the hearing and vocalization of fishes.

C.5.1.4 General Threats

Fish populations can be influenced by various natural factors and human activities. There can be direct effects, from disease or from commercial and recreational activities such as fishing, or indirect effects, such as those associated with reductions in prey availability or lowered reproductive success of individuals. Human-made impacts are widespread throughout the world's oceans, such that very few habitats remain unaffected by human influence (Halpern et al., 2008a). Direct and indirect effects have shaped the condition of marine fish populations, particularly those species with large body size, late maturity ages, or low fecundity such as sharks, Pacific cod (*Gadus macrocephalus*), and Pacific bluefin tuna, making these species especially vulnerable to habitat losses and fishing pressure (Reynolds et al., 2005). Human-induced stressors (e.g., threats) can be divided into four components, which often act on fish populations simultaneously: habitat alteration, exploitation, introduction of non-native species, and pollution (Moyle & Cech, 2004). Climate change and its resulting effects on the marine environment are additional stressors on fish populations.

Coastal development, deforestation, road construction, dam development, water control structures, and agricultural activities are types of habitat alteration that can affect fishes and their environment. These activities may affect the water quality of the nearshore marine environment.

C.5.1.4.1 Water Quality

Parameters such as temperature, DO, salinity, turbidity, and pH define the water quality as a component of habitat quality for fishes. Some land-based activities can directly and indirectly impact water quality in rivers, estuaries, and in the coastal waters. Sediment from activities on land may be transported to the marine environment. Sediment can impact water quality by increasing turbidity and decreasing light penetration into the water column, as well as transport contaminants into the marine environment (Allen, 2006). Increases in sediment can decrease the survival and reproduction of plankton and have food web and ecosystem level effects.

Hypoxia (low DO concentration) is a major impact associated with poor water quality. Hypoxia occurs when waters become overloaded with nutrients such as nitrogen and phosphorus, which enter oceans from agricultural runoff, sewage treatment plants, bilge water, and atmospheric deposition. An overabundance of nutrients can stimulate algal blooms, resulting in a rapid expansion of microscopic algae (phytoplankton) and can cause anoxic events leading to fish kills (Corcoran et al., 2013). Over the last several decades, coastal regions throughout the world have experienced an increase in the frequency of algal blooms that are toxic or otherwise harmful. Commonly called red tides, these events are now grouped under the descriptor harmful algal blooms (Anderson et al., 2002). Harmful algal blooms can produce toxins, causing human illness and massive fish and other animal mortalities.

C.5.1.4.1.1 Pollution

Chemicals and debris are the two most common types of pollutants in the marine environment. Global oceanic circulation patterns result in the accumulation of a considerable amount of pollutants and debris scattered throughout the open ocean and concentrated in gyres and other places (Crain et al., 2009). Pollution initially impacts fishes that occur near the sources of pollution, but may also affect future generations from effects to reproduction and increased mortality across life stages.

Chemical pollutants in the marine environment that may impact marine fishes include organic pollutants (e.g., pesticides, herbicides, polycyclic aromatic hydrocarbons, flame retardants, and oil) and inorganic pollutants (e.g., heavy metals) (Pew Oceans Commission, 2003). High chemical pollutant levels in marine fishes may cause behavioral changes, physiological changes, or genetic damage (Goncalves et al., 2008; Moore, 2008; Pew Oceans Commission, 2003). Bioaccumulation is the net buildup of substances (e.g., chemicals or metals) in an organism from inhabiting a contaminated habitat or from ingesting food or prey containing the contaminated substance (Newman, 1998), or from ingesting the substance directly (Moore, 2008). Biomagnification can also be a concern for fishes. Biomagnification occurs when higher trophic organisms (predators) consume many lower trophic organisms that have accumulated toxins, potentially resulting in a higher toxin load for higher trophic organisms. Bioaccumulation and biomagnification of pollutants (e.g., metals and organic pollutants) is also a concern to human health because people consume top predators with high pollutant loads.

C.5.1.4.1.2 Oil Spills

Groups of fish typically impacted by oil spills include surface-oriented or surface dwelling species, nearshore (within 3 NM of the shoreline) species, and species whose spawning time coincided with an oil spill (Yender et al., 2010). Fishes can be impacted by the oil directly through the gills, or by consuming oil or oiled prey. Potentially harmful physiological effects to fishes from oil spills include reduced growth, enlarged livers, changes to heart and respiration rate, fin erosion, and reproductive impairment. The most damaging effects of oil on fish populations may be in harming eggs and larvae, because these stages are highly sensitive to oil at the surface, in the water column, or on the seafloor, and are subject to increased mortality and morphological deformities and impaired growth (Greer et al., 2012; Ingvarsdottir et al., 2012; National Oceanic and Atmospheric Administration, 2014; Ocean Conservancy, 2010a; Restore the Gulf, 2010). Discharges from ballast water and bilge water during routine ship operations and illegal dumping of solid waste are other sources of oil in the marine environment.

C.5.1.4.2 Commercial and Recreational Activities

Exploitation by commercial and recreational fishing is the single biggest cause of changes in fish populations and communities (Moyle & Cech, 2004). Historic and current overfishing largely contributed to the listing of ESA-protected marine species (Crain et al., 2009; Kappel, 2005). Overfishing of a fishery resource results from both legal and illegal fishing (poaching) and bycatch of resources in quantities above a sustainable level. At the end of 2017, 30 managed fish stocks in the U.S. were on the overfishing list and 35 stocks were on the overfished list, while the number of rebuilt fish stocks since 2000 increased to 44 (National Marine Fisheries Service, 2016v, 2018d).

In recent decades, commercial fisheries have targeted the larger, predatory, and sometimes higher-priced fish species. Gradually, this fishing pressure could make the larger species more scarce, and fishing will move towards the smaller species (Pauly & Palomares, 2005). Other factors, such as fisheries-induced evolution and intrinsic vulnerability to overfishing, have been shown to reduce the abundance of some populations (Kauparinen & Merila, 2007). Fisheries-induced evolution is a change in genetic composition of the population that results from intense fishing pressure, such as a reduction in the overall size and growth rates of fishes in a population. Intrinsic vulnerability is when certain life history traits (e.g., large body size, late maturity age, low growth rate, low offspring production) result in a species being more susceptible to overfishing than others (Cheung et al., 2007).

Other threats from commercial industries to fishes include vessel strikes, sea farming, and energy production activities. Large commercial passenger vessels (e.g., cruise liners) pose threats to large, slow-moving open ocean fishes while moving along the sea surface. Whale sharks (*Rhincodon typus*), basking sharks (*Cetorhinus maximus*), sturgeons (Acipenseridae), manta rays (*Manta* spp), and ocean sunfish (*Mola mola*) are vulnerable to ship strikes (National Marine Fisheries Service, 2010a; Rowat et al., 2007; Stevens, 2007).

The threats of aquaculture operations on wild fish populations include reduced water quality, competition for food, predation by escaped or released farmed fishes, spread of disease and parasites, and reduced genetic diversity (Kappel, 2005). These threats become apparent when farmed fish escape and enter the natural ecosystem (Hansen & Windsor, 2006; Ormerod, 2003). NOAA (2011) published the Marine Aquaculture Policy which provides direction to enable the development of sustainable marine aquaculture.

Energy production and offshore activities associated with power-generating facilities results in direct and indirect injury and/or mortality of fishes. Injury and mortality sources include entrainment of eggs and larvae during water withdrawal and impingement of juveniles and adults (U.S. Environmental Protection Agency, 2004). Acoustic impacts from offshore wind energy development are additional sources of injury and mortality (Madsen et al., 2006).

C.5.1.4.3 Disease and Parasites

Fishes in poor quality environments have higher incidences of disease, due to increased stress levels and decreased immune system function and are less resilient to fight the disease. Parasites, bacteria, aquaculture conditions, environmental influences, and poor nourishment contribute to fish disease levels (National Oceanic and Atmospheric Administration, 2016a). Disease outbreaks in fishes are influenced by environmental conditions, which typically are more variable in inland waters compared to the open ocean (Snieszko, 1978). Areas with higher density fish populations, such as marine protected areas and fish farms, are at higher risk for disease compared to areas with lower densities (National Oceanic and Atmospheric Administration, 2016c; Wootton et al., 2012). Additionally, introduced species may expose native species to new diseases and parasites. In Hawaii, the introduction of the bluestripe snapper (*Lutjanus kasmira*) native to the Indian Ocean introduced a parasitic nematode (*Procamallanus istiblenni*) that has spread to native fish species (Gaither et al., 2013).

C.5.1.4.4 Invasive Species

Native fish populations are affected by invasive (introduced, non-native) species by predation, competition and hybridization (Moyle & Cech, 2004). Non-native fishes pose threats to native fishes when they are introduced into an environment lacking natural predators and then either compete with native marine fishes for resources or prey upon the native marine fishes (Coleman et al., 2014; Crain et al., 2009). Marine invasions by other non-fish species also may impact fish populations. Invasive marine algae have been found to alter the health status of native fishes feeding on the algae, which could impact the reproduction success of those populations (Felline et al., 2012).

In the Study Area, some of the invasive species include the peacock grouper (*Cephalopholis argus*), introduced to Hawaii, the yellowfin goby (*Acanthogobius flavimanus*), and the rainwater killifish (*Lucania parva*). The yellowfin goby is native to eastern Asia and the rainwater killifish is native to the U.S. Atlantic coast. Both of these fishes have also been introduced to the San Diego Bay (Gaither et al., 2013).

C.5.1.4.5 Climate Change

Global climate change is impacting and will continue to impact marine and estuarine fish and fisheries (Giddens et al., 2022; Intergovernmental Panel on Climate Change, 2014; Johnson et al., 2020; Roessig et al., 2004). Climate change is contributing to a shift in fish distribution from lower to higher latitudes

(Blanchard & Novaglio, 2024; Brander, 2010; Brander, 2007; Dufour et al., 2010; Giddens et al., 2022; Popper & Hastings, 2009; Wilson et al., 2010). Warming waters over the past quarter-century have driven fish populations in the northern hemisphere northward and to deeper depths (Asch, 2015; 2012; Heuer & Grosell, 2014; Inman, 2005; Peterson et al., 2014a).

Fishes with shifting distributions have faster life cycles and smaller body sizes than non-shifting species (Perry et al., 2005). In addition to affecting species ranges, increasing temperature has been shown to alter the sex-ratio in fish species that have temperature-dependent sex determination mechanisms (Ospina-Alvarez & Piferrer, 2008). Further temperature rises are likely to have profound impacts on commercial fisheries through continued shifts in distribution and alterations in community interactions (Perry et al., 2005). It appears that diadromous and benthic fish species are most vulnerable to climate change impacts (Hare et al., 2016).

Ocean acidification, the process whereby increasing atmospheric CO_2 concentrations reduces ocean pH and carbonate ion concentrations, may have serious impacts on fish development and behavior (Raven et al., 2005). Physiological development of fishes can be affected by increases in pH that can increase the size, density, and mass of fish otoliths (e.g., fish ear stones), which would affect sensory functions (Bignami et al., 2013b). Ocean acidification may affect fish larvae behavior and could impact fish populations (Munday et al., 2009). A range of behavioral traits critical to survival of newly settled fish larvae are affected by ocean acidification. Settlement-stage larval marine fishes exposed to elevated CO^2 were less responsive to threats than controls. This decrease in sensitivity to risk might be directly related to the impaired olfactory ability (Munday et al., 2009).

Beyond direct impacts on fishes from increasing pH, ocean acidification can cause changes to the ocean chemistry, which leads to increased algal blooms (Anderson et al., 2002). Ocean acidification can also lead to reef impacts, such as coral bleaching, and can also lead to reduced larval settlement and abundance (Doropoulos et al., 2012). Plankton are important prey items for many fish species and are also impacted by ocean acidification. Ocean acidification may cause a shift in phytoplankton community composition and biochemical composition that can impact the transfer of essential compounds to predators that eat the plankton (Bermudez et al., 2016) and can cause shifts in community composition (Anderson et al., 2002; Bermudez et al., 2016; Doropoulos et al., 2012; Fabry et al., 2008; Kroeker et al., 2013).

Another climate change effect is ocean deoxygenation. Netburn and Koslow (2015) found that the depth of the lower boundary of the deep scattering layer (so-called because the sonic pulses of a sonar can reflect off the millions of fish swim bladders) is most strongly correlated with DO concentration, and irradiance and oxygen concentration are the key variables determining the upper boundary. This study estimated the corresponding annual rate of change of deep scattering layer depths and hypothesized that if past trends continue, the upper boundary is expected to rise at a faster rate than the lower boundary, effectively widening the deep scattering layer. Cao et al. (2014) modeled different sensitivities of ocean temperature, carbonate chemistry, and oxygen, in terms of both the sign and magnitude to the amount of climate change. Model simulations in this study found by the year 2500, every degree increase of climate sensitivity will warm the ocean by 0.8 °C and will reduce ocean-mean DO concentration by 5.0 percent. Conversely, every degree increase of climate sensitivity buffers CO_2-induced reduction in ocean-mean carbonate ion concentration and pH by 3.4 percent and 0.02 units, respectively. These results have great implications for understanding the response of ocean biota to climate change. Keller et al. (2015b) suggested that within the California Current System, shoaling of the oxygen minimum zone is expected to produce complex changes and onshore movement of the oxygen minimum zone could lead to habitat compression for species with higher oxygen requirements while allowing expansion of species tolerant of low bottom DO.

C.5.1.4.6 Marine Debris

Marine debris is a widespread global pollution problem, and trends suggest that accumulations are increasing as plastic production rises (Rochman et al., 2013). Debris includes plastics, metals, rubber, textiles, derelict fishing gear, vessels, and other lost or discarded items. Debris such as abandoned nets and lines also pose a threat to fishes. Due to body shape, habitat use, and feeding strategies, some fishes are more susceptible to marine debris entanglement than others (Musick et al., 2000; Ocean Conservancy, 2010b). Entanglement in abandoned commercial and recreational fishing gear has caused declines for some marine fishes.

Microplastics (i.e., plastics less than 5mm in size) in the marine environment are well documented, and interactions with marine biota, including numerous fish species have been described worldwide (Lusher et al., 2016). Plastic waste in the ocean chemically attracts hydrocarbon pollutants such as PCB and DDT, which accumulate up to one million times more in plastic than in ocean water (Mato et al., 2001). Fishes and zooplankton can mistakenly consume these wastes containing elevated levels of toxins instead of their prey. Rochman et al., (2015) found marine debris in 28 percent of the individual fish examined and in 55 percent of all fish species analyzed. According to the California Coastal Commission, only 20 percent of the items found in the ocean can be linked to ocean-based sources, like commercial fishing vessels, cargo ships (discharge of containers and garbage), or pleasure cruise ships, while 80 percent of the debris is land based from sources like litter, industrial discharges, and garbage management (California Coastal Commission, 2018).

C.5.2 Endangered Species Act-Listed Species

ESA-listed fishes in the Study Area include three Evolutionarily Significant Units (ESU) of Chinook salmon, three ESUs of coho salmon, five Distinct Population Segments (DPS) of steelhead, green sturgeon, eulachon, oceanic whitetip shark, scalloped hammerhead shark (*Sphyrna lewini*), and giant manta (*Manta birostris*).

C.5.2.1 Chinook Salmon (*Oncorhynchus tshawytscha*)

Three ESA-listed ESUs of Chinook salmon have the potential to occur in the NOCAL Range Complex portion of the Study Area, including the California Coastal ESU, Central Valley Spring-Run ESU, and Sacramento River Winter-Run ESU.

C.5.2.1.1 California Coastal ESU

C.5.2.1.1.1 Status and Management

The California Coastal Chinook Salmon ESU was listed as threatened on September 16, 1999 (64 FR 50394); that status was reaffirmed on June 28, 2005 (70 FR 37160) and subsequently updated on April 14, 2014 (79 FR 20802). This ESU includes naturally spawned Chinook salmon originating from rivers and streams south of the Klamath River to and including the Russian River (79 FR 20802). No hatchery programs are currently included as part of this ESU.

Critical Habitat

NMFS designated critical habitat for the California Coastal Chinook salmon on September 2, 2005 (70 FR 52488). It includes multiple CALWATER hydrological units north from Redwood Creek and south to Russian River. The physical and biological features required by Chinook salmon are applicable to freshwater and estuaries (i.e., spawning sites, rearing sites, and migration corridors) and are outside the Study Area (Figure C-4). Therefore, designated critical habitat for the California Coastal ESU of Chinook salmon does not overlap spatially with the HCTT Study Area.

Recovery Goals

Recovery goals, objectives and criteria for the California Coastal Chinook salmon are fully outlined in NMFS (National Marine Fisheries Service, 2016l). Recovery plan objectives are to: 1) Reduce the present or threatened destruction, modification, or curtailment of habitat or range; 2) Ameliorate utilization for commercial, recreational, scientific, or educational purposes; 3) Abate disease and predation; 4) Establish the adequacy of existing regulatory mechanisms for protecting California Coastal Chinook salmon now and into the future (i.e., post-delisting); 5) Address other natural or manmade factors affecting the continued existence of California Coastal Chinook salmon; and 6) Ensure the status of California Coastal Chinook salmon is at a low risk of extinction based on abundance, growth rate, spatial structure and diversity.

C.5.2.1.1.2 Habitat and Geographic Range

Geographic range of spawning systems for this ESU: Naturally spawned Chinook salmon originating from rivers and streams south of the Klamath River to and including the Russian River (79 FR 20802).

Juvenile outmigrants

With the extirpation of spring-runs in this ESU, California Coastal Chinook are comprised almost entirely of ocean-type, fall-run fish (Crozier et al., 2019; Moyle et al., 2017; National Marine Fisheries Service, 2016e). The majority of California Coastal fall-run Chinook emerge from the gravel in the late winter or spring, then outmigrate to estuaries from March through August of their first year. Most of these fish enter marine waters by July (Chase et al., 2005; Gallagher, 2003), though in some smaller systems where sedimentation blocks their egress, they may remain until October or November (Madej et al., 2012). When these fish leave their natal systems and migrate to coastal marine habitats, marine entry occurs to the north and east of the northernmost portion of the California Study Area (Figure C-4).

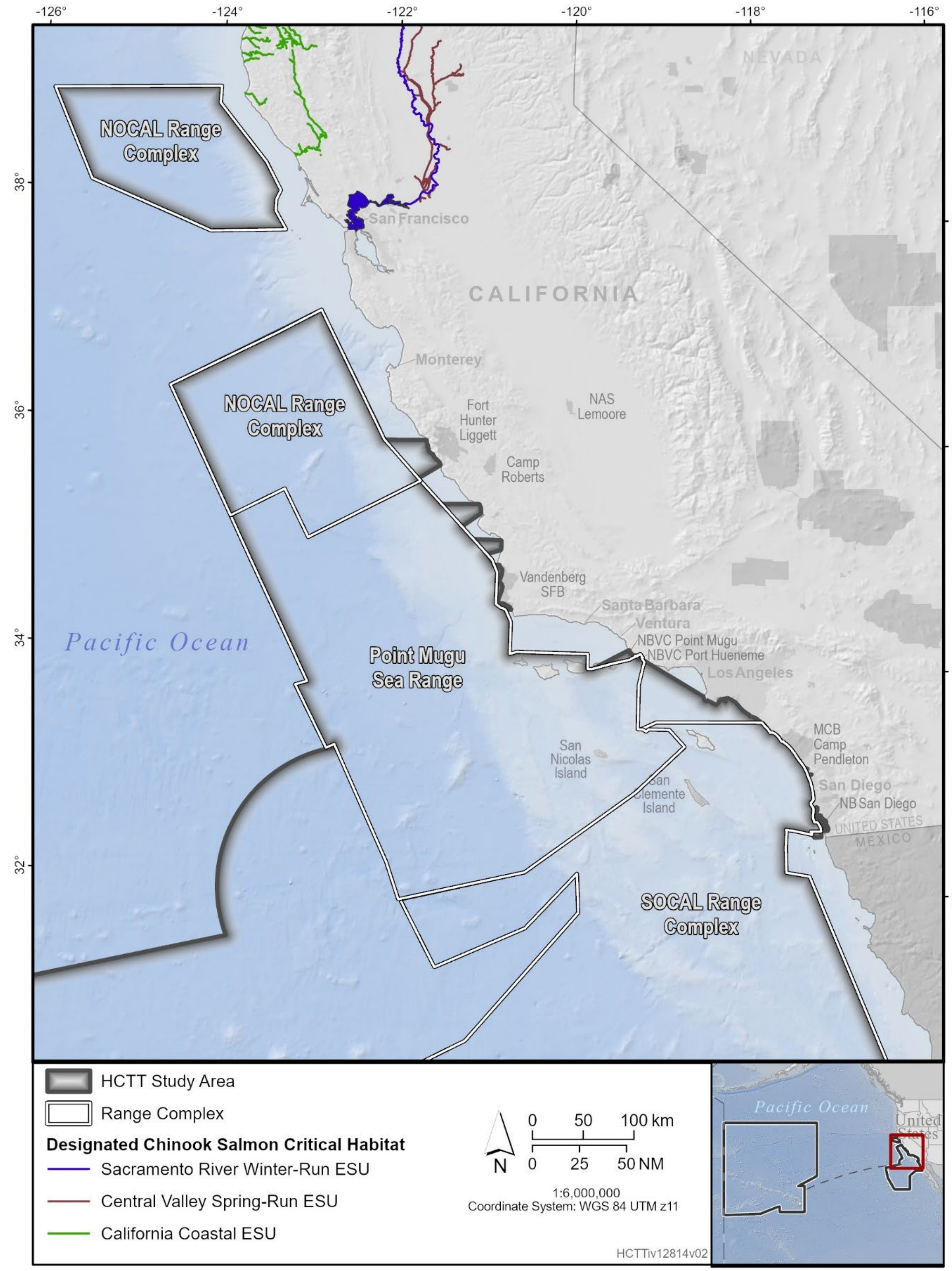

Figure C-4: Designated Chinook Salmon Critical Habitat

Ocean rearing

Chinook and coho along the California Current utilize the cooler, upwelled water of the coastal shelf for feeding and migrating (Bellinger et al., 2015; Hinke et al., 2005). Upon marine entry, California Coastal Chinook stay near the coastline, with many of these fish moving north and feeding in cool waters off the Klamath-Trinidad region (Moyle et al., 2017). Older juveniles continue to rear in these productive waters, typically occurring between Pt. Reyes and southern Oregon, with highest abundances in the Fort Bragg and Klamath subareas (Bellinger et al., 2015; Satterthwaite et al., 2015; Satterthwaite et al., 2014 Crozier, 2019, #15602), north of the Study Area. In an analysis of Oregon and California coastal fish, Bellinger et al., (2015) summarized the May to September stock-specific catch per unit effort for troll caught Chinook. They found that California Coastal Chinook were broadly distributed from May to August, from northern Oregon to Monterey Bay. The greatest and most consistent stock-specific catch per unit effort (SSCPUEs) for California Coastal Chinook occurred between Crescent City and Fort Bragg. Fisher et al., (2007) found that subyearling Chinook can be abundant in nearshore waters in June and July, with their distribution more tightly associated with depth, than temperature or salinity. Though limited catch data was available for yearling Chinook, Fisher et al., (2007), found that yearling Chinook along the California coast were found in very shallow waters and were relatively absent from both the warmest and coolest waters.

To investigate the marine habitat utilization of maturing Chinook salmon in the Gulf of Alaska, Seitz and Courtney (2022, 2023, 2024) attached 20 pop-up satellite archive tags (PSATS) on large immature Chinook at each of five different locations. Each fish was tracked for approximately 2–6 months. At each location, they found that maturing Chinook tended to occupy marine habitats over the continental shelf relative to those that occurred over continental slope or basin habitats. It is likely that Chinook in other regions of the Northeast Pacific Ocean display similar at-sea life history behaviors.

Return migration

Returning fall and late-fall maturing fish from the California Coastal ESU are typically age-3 and age-4 fish, with age-2 jacks representing approximately 5–10 percent of returning fish (Crozier et al., 2019; Myers et al., 1998). These fish leave coastal marine waters and enter estuaries as early as September and as late as January (Moyle et al., 2017; Myers et al., 1998), migrating upstream to habitats where spawning typically occurs between late October and December (Moyle et al., 2017).

C.5.2.1.1.3 Population Trends

The 2016 status review for this ESU indicates that there has been a mix in population trends, with some river system population escapement numbers increasing and others decreasing. Overall, there is a lack of compelling evidence to suggest that the status of these populations has improved or deteriorated appreciably since the previous status review (Williams et al., 2011; Williams et al., 2016c). At the ESU level, the loss of the spring-run life history type represents a significant loss of diversity within the ESU, as has been noted in previous status reviews (Good et al., 2005; Williams et al., 2011). Concern remains about the extremely low numbers of Chinook salmon in most populations of the North-Central Coast and Central Coast strata, which diminishes connectivity across the ESU. In summary, the new information available since the last status review does not appear to suggest there has been a change in extinction risk for this ESU (National Marine Fisheries Service, 2016g). Although conservation efforts for Chinook salmon have reduced some threats for this ESU, the threats have, with few exceptions, remained unchanged since the last review. Poor ocean conditions, drought, and marijuana cultivation have significant negative impacts on Chinook salmon populations in this ESU since the last review (National Marine Fisheries Service, 2016g).

C.5.2.1.1.4 Population Threats

Threats to Chinook populations are largely applicable across multiple ESUs. In addition to the primary concerns articulated in the recovery plan (National Marine Fisheries Service, 2016l), salmonids have many other concerns that affect their respective populations. Juvenile Chinook salmon are eaten by other fishes such as whiting and mackerel, and seabirds, while adult Chinook are preyed on by marine mammals such as sea lions and orcas, as well as other fishes such as sharks (National Marine Fisheries Service, 2024c; Seitz & Courtney, 2022, 2023, 2024).

Chinook salmon populations are threatened by warming temperatures and changing conditions in freshwater and ocean habitats. Ocean acidification, a climate change related process where increasing atmospheric CO_2 concentrations are reducing ocean pH and carbonate ion concentrations, may have serious impacts on fish development and behavior (Raven et al., 2005). Physiological development of fishes can be affected by increases in pH that can increase the size, density, and mass of fish otoliths (e.g., fish ear stones), which would affect sensory functions (Bignami et al., 2013a). Ocean acidification may affect fish larvae behavior and could impact fish populations (Munday et al., 2009). A range of behavioral traits critical to survival of newly settled fish larvae are affected by ocean acidification. Settlement-stage larval marine fishes exposed to elevated CO_2 were less responsive to threats than controls (Munday et al., 2009). This decrease in sensitivity to risk might be directly related to impaired olfactory ability (Munday et al., 2009). Ocean acidification may cause a shift in plankton community composition and biochemical composition that can impact the transfer of essential compounds to planktivorous organisms (Bednaršek et al., 2022; Bermudez et al., 2016; Mélançon et al., 2016) and can cause shifts in community composition up the food chain.

Another effect of climate change is ocean deoxygenation. Netburn and Koslow (2015) found that the depth of the lower boundary of the deep scattering layer (so-called because the sonic pulses of a sonar can reflect off the millions of fish swim bladders) is most strongly correlated with DO concentration. Cao et al. (2014) modeled different sensitivities of ocean temperature, carbonate chemistry, and oxygen, in terms of both the sign and magnitude, and correlated them to the amount of climate change. Model simulations in a study by Cao et al. (2014) found that, by the year 2500, every degree increase of climate sensitivity will warm the ocean by 0.8° C and will reduce ocean-mean DO concentration by 5.0 percent. Conversely, every degree increase of climate sensitivity buffers CO_2-induced reduction in ocean-mean carbonate ion concentration and pH by 3.4 percent and 0.02 units, respectively. These results have great implications for understanding the response of ocean biota to climate change. Keller et al. (2015a) suggested that within the California Current System, shoaling of the oxygen minimum zone (OMZ) is expected to produce complex changes and onshore movement of the OMZ that could lead to habitat compression for species with higher oxygen requirements while allowing expansion of species tolerant of low bottom DO. Ruz-Moreno (2023) found that alterations in region-specific OMZ are likely to have significant impacts on the zooplankton community and productivity.

C.5.2.1.2 Central Valley Spring-Run ESU

C.5.2.1.2.1 Status and Management

The Central Valley Spring-Run Chinook Salmon ESU was listed as threatened on September 16, 1999 (64 FR 50394); that status was reaffirmed on June 28, 2005 (70 FR 37160) and subsequently updated on April 14, 2014 (79 FR 20802). This ESU includes naturally spawned spring-run Chinook salmon originating from the Sacramento River and its tributaries, and spring-run Chinook salmon from the Feather River Hatchery Spring-run Chinook Program. This ESU does not include Chinook salmon that are designated as part of an experimental population (79 FR 20802). NMFS has not proposed any changes to the hatchery listing status for this ESU in their 2016 proposed rule (81 FR 72759).

Critical Habitat

NMFS published a final rule designating critical habitat for Central Valley Spring-Run Chinook salmon on September 2, 2005 (70 FR 52488). The physical and biological features required by Chinook salmon are applicable to freshwater and estuaries (i.e., spawning sites, rearing sites, and migration corridors) and are outside the Study Area (Figure C-4). Therefore, designated critical habitat for the Central Valley Spring-Run ESU of Chinook salmon does not overlap spatially with the HCTT Study Area.

Recovery Goals

Recovery goals, objectives and criteria for the Central Valley Spring-Run Chinook salmon are fully outlined in the 2014 recovery plan (National Marine Fisheries Service, 2014e). The ESU delisting criteria for the spring-run Chinook salmon are: 1) One population in the Northwestern California Diversity Group at low risk of extinction; 2) Two populations in the Basalt and Porous Lava Diversity Group at low risk of extinction; 3) Four populations in the Northern Sierra Diversity Group at low risk of extinction; 4) Two populations in the Southern Sierra Diversity Group at low risk of extinction; and 5) Maintain multiple populations at moderate risk of extinction.

C.5.2.1.2.2 Habitat and Geographic Range

Geographic range of spawning systems for this ESU: Naturally spawned spring-run Chinook salmon originating from the Sacramento River and its tributaries. Also, spring-run Chinook salmon from the Feather River Hatchery Spring-run Chinook Program. This DPS does not include Chinook salmon that are designated as part of an experimental population (79 FR 20802).

Juvenile outmigrants

The majority of Central Valley Spring-run Chinook emerge from the gravel by March and begin their downstream migration shortly thereafter (Crozier et al., 2019; Moyle, 2002). Predictive estuary rearing time and marine entry for these different spring-run age classes is unknown, with some juveniles remaining in the estuary for extended periods, whereas others in the same age class are believed to migrate directly towards the Pacific Ocean (Brandes & McLain, 2000; California Department of Fish and Game, 1998; Crozier et al., 2019; Moyle et al., 2017; National Marine Fisheries Service, 2014e; Whipple et al., 2012). Juvenile entry into marine waters for this ESU can occur as early as age-0 spring outmigrants, as fall or winter age-O outmigrants, or as late as age-1 spring outmigrants (Cordoleani et al., 2019; Cordoleani et al., 2021a; Cordoleani et al., 2021b; Cordoleani et al., 2020; Goertler et al., 2020; Moyle et al., 2017; Satterthwaite et al., 2017). Although Cordoleani et al. (2021b) noted that age-1 spring outmigrants were the rarest phenotype observed at juvenile monitoring traps (10 percent), these later migrating fish contributed to the majority of returning adult spring-run spawners (60 percent). The peak outmigration for spring-outmigrating fry occurs in April and May (Baker & Morhardt, 2001).

Ocean rearing

In a review of data from coded wire tag tags recovered in the marine environment, Weitkamp et al., (2009) found that the majority of Central Valley-origin Chinook remained in California waters, though some fish were also detected in Oregon and Washington waters. In an analysis of Oregon and California coastal troll caught fish, Bellinger et al., (2015) summarized the May to September SSCPUEs for Chinook. They found Central Valley Spring-run Chinook first appeared in the catch records in May, just south of San Francisco Bay. By June, they were detected from north of Monterey Bay to north of San Francisco Bay. By August and September, fish from this ESU were caught from the central Oregon coast to south of Monterey Bay. However, marine spatial distribution of rare stocks, such as Central Valley Spring-run Chinook, are limited due to their infrequent detection in the marine environment (Satterthwaite et al., 2015).

As discussed for California Coastal Chinook, Seitz and Courtney (2022, 2023, 2024) found that PSAT-tagged Chinook salmon in the Gulf of Alaska preferred to occupy habitats over the continental shelf relative to those that occurred over continental slope or basin habitats. It is likely that Chinook in other regions of the North Pacific Ocean display similar life history behaviors.

Return migration

Returning maturing fish from the Central Valley Spring-run ESU are typically age-3 and age-4 fish (Fisher, 1994; Moyle et al., 2017; National Marine Fisheries Service, 2014e). These fish enter the estuary to migrate up the Sacramento River from late January to early February, with peak detections in the Sacramento River from April to June (Crozier et al., 2019; Moyle et al., 2017). Characteristic for spring-run Chinook, these maturing fish hold in the river until conditions are suitable for spawning in the fall, typically beginning around October (California Department of Fish and Game, 1998; Moyle et al., 2017).

C.5.2.1.2.3 Population Trends

The Central Valley Technical Recovery Team delineated 18 or 19 independent populations of Central Valley Spring-Run Chinook salmon that occurred historically, along with a number of smaller dependent populations, within four diversity groups (Lindley et al., 2004). Of these populations, only three are extant (Mill, Deer, and Butte creeks) and they occur only in the Northern Sierra Nevada Diversity Group. With a few exceptions, Central Valley Spring-Run Chinook salmon populations have increased since the previous status review (2010/2011), which has moved the Mill and Deer creek populations from the high extinction risk category, to moderate, and Butte Creek has remained in the low risk of extinction category (National Marine Fisheries Service, 2016d; Williams et al., 2016c). Additionally, the Battle Creek and Clear Creek populations have continued to show stable or increasing numbers the last five years, putting them at moderate risk of extinction based on abundance. Overall, the Southwest Fisheries Science Center concluded in their viability report that the status of Central Valley Spring-Run Chinook salmon (through 2014) has probably improved since the 2010/2011 status review and that the ESU's extinction risk may have decreased, however the ESU is still facing significant extinction risk, and that risk is likely to increase over at least the next few years as the full effects of the recent drought are realized (Williams et al., 2016c).

C.5.2.1.2.4 Population Threats

Population threats are similar across many Chinook salmon ESUs. Please see the description of these threats described for the California Coastal Chinook Salmon ESU.

C.5.2.1.3 Sacramento River Winter-Run ESU

C.5.2.1.3.1 Status and Management

The Sacramento River Winter-Run Chinook Salmon ESU was listed as threatened on August 4, 1989 (54 FR 32085) and it was downgraded to endangered in 1994; that status was reaffirmed on June 28, 2005 (70 FR 37160) and subsequently updated on April 14, 2014 (79 FR 20802). This ESU includes all naturally spawned populations of winter-run Chinook salmon in the Sacramento River and its tributaries, as well as two conservation programs maintained at the Livingston-Stone National Fish Hatchery (79 FR 20802). NMFS has not proposed any significant changes to the hatchery listing status for this ESU in their 2016 proposed rule (81 FR 72759).

Critical Habitat

NMFS designated critical habitat for the Sacramento River Winter-Run Chinook salmon on June 16, 1993 (58 FR 33212). The physical and biological features required by Chinook salmon are applicable to freshwater and estuaries (i.e., spawning sites, rearing sites, and migration corridors) and are outside the Study Area (Figure C-4). Therefore, designated critical habitat for the Sacramento River Winter-Run ESU of Chinook salmon does not overlap spatially with the HCTT Study Area.

Recovery Goals

Recovery goals, objectives and criteria for the Sacramento River Winter-Run Chinook salmon are fully outlined in the 2014 recovery plan (National Marine Fisheries Service, 2014e). To achieve the downlisting criteria, the species would need to be composed of two populations – one viable and one at moderate extinction risk. Having a second population would improve the species' viability, particularly through increased spatial structure and abundance, but further improvement would be needed to reach the goal of recovery. The only delisting criteria for this ESU is to have three populations in the Basalt and Porous Lava Diversity Group at low risk of extinction.

C.5.2.1.3.2 Habitat and Geographic Range

Geographic range of spawning systems for this ESU: Naturally spawned winter-run Chinook salmon originating from the Sacramento River and its tributaries. Also, winter-run Chinook salmon from one artificial propagation program: the Livingston Stone National Fish Hatchery (79 FR 20802).

Juvenile outmigrants

Winter-run Chinook are unique to the Central Valley (Healey, 1991) and have significant plasticity in their life history. Sacramento River Winter-run Chinook fry emerge from July to mid-October and may rear in the lower Sacramento for a few months prior to migrating to marine waters from December to April (Crozier et al., 2019; National Marine Fisheries Service, 2014e; Phillis et al., 2018; Pyper et al., 2013).

Ocean rearing

Sacramento River Winter-run Chinook salmon have a more southerly ocean distribution relative to other California Chinook salmon populations and are primarily impacted by fisheries south of Point Arena, California (National Marine Fisheries Service, 2016f). For Central Valley Chinook, both winter-run and late fall-run fish tend to occupy waters off California, whereas fall- and spring-run fish are distributed well into Oregon waters (Satterthwaite et al., 2013). In an analysis of Oregon and California coastal troll caught fish, Bellinger et al., (2015) summarized the May to September stock-specific catch per unit effort for Chinook. Except for a few June detections south of San Francisco Bay, the authors found that winter Chinook originating from the Central Valley (including the Sacramento watershed) were absent from the catch from May to July. During August and September, when their detections were greatest, they were only caught in the vicinity of Monterey Bay, and were slightly more prevalent to the south of the bay, relative to the north (Bellinger et al., 2015). These findings might have indicated that winter-run Chinook originating from the Central Valley do not typically rear off coastal California and only occur as they are entering or exiting the estuary. Satterthwaite et al., (2013) also reviewed coded wire tag recoveries for this region. The authors found that although winter-run Chinook recoveries were low, and the greatest number of detections occurred during three years (2000, 2004, and 2005), that they appeared to be restricted to California coastal waters south of Point Arena, with 72 percent of the age-3 winter fish detected in June occurring in this area. Taken in combination, these studies suggest winter-run Chinook appear to have a more southerly distribution than spring-run Chinook, and they appear to be more abundant in the catch record in fall months.

As discussed for California Coastal Chinook, Seitz and Courtney (2022, 2023, 2024) found that PSAT-tagged Chinook salmon in the Gulf of Alaska preferred to occupy habitats over the continental shelf relative to those that occurred over continental slope or basin habitats. It is likely that Chinook in other regions of the North Pacific Ocean display similar life history behaviors.

Return migration

As age-3, maturing Sacramento River Winter-run Chinook typically enter the San Francisco Bay-Delta from December through July, with peak arrival occurring in March (Crozier et al., 2019; O'Farrell et al., 2012; Satterthwaite et al., 2017; Yoshiyama et al., 1998). These fish will hold in the Sacramento River and tributaries until spawning the following spring and summer (Crozier et al., 2019; Fisher et al., 2014; Killam & Mache, 2018; National Marine Fisheries Service, 2014e; Yoshiyama et al., 1998).

C.5.2.1.3.3 Population Trends

The Central Valley Technical Recovery Team delineated four historical independent populations of this ESU. The spawning areas for three of these historical populations are above the impassable Keswick and Shasta dams, while the fourth population (Battle Creek) is presently unsuitable for Sacramento River Winter-run Chinook salmon due to high summer water temperatures. Lindley et al. (2007) developed viability criteria for Central Valley salmonids and, using data through 2004, found that the mainstem Sacramento River population was at low risk of extinction, but that the ESU as a whole remained at a high risk of extinction because there is only one naturally-spawning population, and it is not within its historical range. The ESU's status has declined since the 2010 status review, with the single spawning population on the mainstem Sacramento River no longer at a low risk of extinction (National Marine Fisheries Service, 2016r). New information indicates an increased extinction risk to Sacramento River Winter-run Chinook salmon. The larger influence of the hatchery broodstock in addition to the rate of decline in abundance over the past decade has placed the population at a moderate risk of extinction. In summary, the most recent biological information suggests that the extinction risk of this ESU has increased since the last status review largely due to extreme drought and poor ocean conditions. The best available information on the biological status of the ESU and new threats to the ESU indicate that its ESA classification as an endangered species is appropriate and should be maintained (National Marine Fisheries Service, 2016r).

C.5.2.1.3.4 Population Threats

Population threats are similar across many Chinook salmon ESUs. Please see the description of these threats described for the California Coastal Chinook Salmon ESU.

C.5.2.2 Coho Salmon (*Oncorhynchus kisutch*)

Three ESA-listed ESUs of coho salmon have the potential to occur in the Study Area, including the Oregon Coast ESU, Southern Oregon and Northern California Coast ESU, and Central California Coast ESU.

C.5.2.2.1 Oregon Coast ESU

C.5.2.2.1.1 Status and Management

The Oregon Coast Coho Salmon ESU was listed as threatened on February 11, 2008 (63 FR 42587); that status was retained on June 20, 2011 (76 FR 35755) and subsequently updated on April 14, 2014 (79 FR 20802). This ESU includes naturally spawned coho salmon originating from coastal rivers south of the Columbia River and north of Cape Blanco, and also coho salmon from one artificial propagation program: Cow Creek Hatchery Program (Oregon Department of Fish and Wildlife Stock #18) (79 FR 20802). NMFS has not proposed any significant changes to the hatchery listing status for this ESU in their 2016 proposed rule (81 FR 72759).

Critical Habitat

NMFS published a final rule designating critical habitat for Oregon Coast coho salmon on February 11, 2008 (73 FR 7816). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Oregon Coast ESU of coho salmon does not overlap spatially with the HCTT Study Area. The only ESA-listed coho salmon with critical habitat designated within 50 NM of the HCTT Study Area is the Central California Coast Coho ESU (Figure C-5).

Recovery Goals

See the 2016 recovery plan for detailed descriptions of the recovery goals and delisting criteria for Oregon Coast coho salmon (National Marine Fisheries Service, 2016l). In the simplest terms, NMFS will remove the Oregon Coast coho salmon from federal protection under the ESA when NMFS determines that:

- The species has achieved a biological status consistent with recovery—the best available information indicates it has sufficient abundance, population growth rate, population spatial structure, and diversity to indicate it has met the biological recovery goals.
- Factors that led to ESA listing have been reduced or eliminated to the point where federal protection under the ESA is no longer needed, and there is reasonable certainty that the relevant regulatory mechanisms are adequate to protect Oregon Coast coho salmon sustainability.

C.5.2.2.1.2 Habitat and Geographic Range

Geographic range of spawning systems for this ESU: Naturally spawned coho salmon originating from coastal rivers south of the Columbia River and north of Cape Blanco. This ESU includes coho from one artificial propagation program; the Cow Creek Hatchery Program (79 FR 20802).

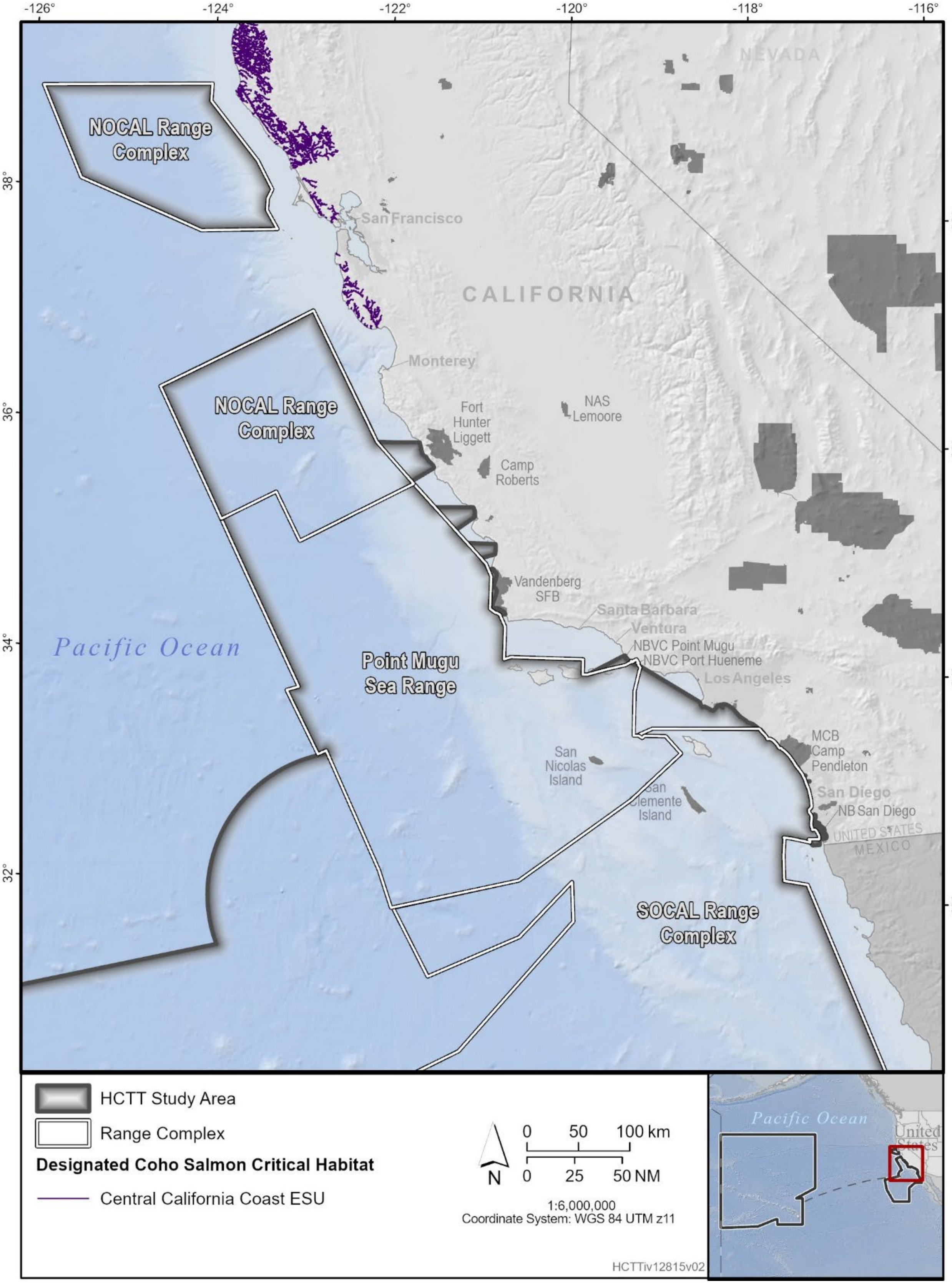

Figure C-5: Designated Coho Salmon Critical Habitat

Juvenile outmigrants

The broad geographic area and highly variable environmental conditions where coho salmon spawn and rear as juveniles results in substantial variability within the life history traits displayed by individual populations in given regions, requiring generalizations, or averages of some traits (Weitkamp et al., 1995). Oregon Coast coho fry emerge in early spring, then typically rear in freshwater lakes, coastal rivers, wetlands, and estuaries for at least one year before migrating to the ocean as age-1 fish from late March through June, with peak outmigration occurring in May (National Marine Fisheries Service, 2016n; Weitkamp et al., 1995). Only a very small proportion of fish from this ESU enter the ocean as sub-yearlings (National Marine Fisheries Service, 2016n; Stout et al., 2012; Weitkamp et al., 1995).

Ocean rearing

Oregon Coast coho typically spend 18–24 months at sea, females typically return as age-3 fish, whereas males can return after only 6 months at sea as age-2 or age-3 fish (Crozier et al., 2019; National Marine Fisheries Service, 2016b, 2016n; NOAA Fisheries Protected Resources Division, 2007; Weitkamp et al., 1995). Given their relatively short period in marine waters, Oregon Coast coho are broadly distributed. This ESU has been found to range from as far south as southern Monterey Bay to as far north as the Gulf of Alaska (Morris et al., 2007; Van Doornik et al., 2007; Weitkamp & Neely, 2002) though the majority of adult ocean harvest occurs off the Oregon Coast (National Marine Fisheries Service, 2016n). Coded wire tagged (CWT) coho from this region display a more northerly marine distribution than populations to the south (Weitkamp & Neely, 2002; Weitkamp et al., 1995). The region-specific recovery of tags by percentages for Oregon Coast-origin coho, as reported by Weitkamp et al. (1995), is California (27–39 percent), Washington (2–9 percent), British Columbia (2–6 percent), and Alaska (<1 percent). During NMFS summer salmon trawl surveys conducted from 2010 to 2014, coho captured in California waters were much more abundant in more northern California waters in the Klamath-Trinidad region than the Lost Coast region, with very few coho captured in the Gulf of Farallones region (Harding, 2015). Coho captured during these surveys occurred at lower abundance for deeper offshore trawl stations relative to shallower stations (Harding, 2015).

Pearcy and Fisher (1988) evaluated 1980–1985 Washington and Oregon purse seine and gill net salmon catch data to determine characteristics of coho salmon in the marine environment. Their data indicate more northerly movement of juvenile coho early in late spring to summer (e.g., May) and more southerly movement towards the end of summer or early fall (e.g., August to September). Juveniles were larger at northern sampling locations relative to those in the south near the California border. They found that half of the juvenile coho were captured in the top 2 m of the water column, though as many as 5 percent of the catch occurred at 9–12 m depth. Pearcy and Fisher reviewed prior literature indicating that coho typically occur from 0 to 20 m depth and that 79 percent of maturing coho occurred at depths from 0 to 50 m.

Return migration

Adult return migration and spawn timing can also be highly variable, with river entry correlated with fall rains elevating river levels (Weitkamp et al., 1995). Mature Oregon Coast coho typically enter estuaries from late September through November (National Marine Fisheries Service, 2016n; Weitkamp et al., 1995) to begin their migration to upstream spawning grounds where spawning occurs from January through March (Crozier et al., 2019; National Marine Fisheries Service, 2016n; NOAA Fisheries Protected Resources Division, 2007; Weitkamp et al., 1995).

C.5.2.2.1.3 Population Trends

Many positive improvements to Oregon Coast coho salmon are described in National Marine Fisheries Service (2016m), including positive long-term abundance trends and escapement. Increases in ESU scores for persistence and sustainability also clearly indicate the biological status of the ESU is improving, due in large part to management decisions (reduced harvest and hatchery releases) and favorable environmental variation (i.e., high marine survival). Lawson (1993) cautioned that variation in ocean productivity can mask the true benefits of stream restoration projects; increased abundances are incorrectly attributed to stream restoration when the increases resulted from high marine survival. Consequently, it is only when marine survival is low that it becomes apparent whether habitat quality and quantity are sufficient to support self-sustaining populations. With marine survival rates expected to decrease for Oregon Coast coho salmon entering the ocean in 2014 (Peterson et al., 2014b; Peterson et al., 2014c), it may be advisable to wait to observe how populations fare during this potential downturn before deciding to change their status (National Marine Fisheries Service, 2015b).

C.5.2.2.1.4 Population Threats

Coho salmon on the west coast of the U.S. have experienced dramatic declines in abundance during the past several decades from human-induced and natural factors (Alaska Department of Fish and Game, 2024; National Marine Fisheries Service, 2016b, 2016n; Weitkamp et al., 2020). Water storage, withdrawal, conveyance, and diversions for agriculture, flood control, domestic, and hydropower purposes have greatly modified, reduced or eliminated historically accessible habitat (National Marine Fisheries Service, 2016b, 2016n; Weitkamp et al., 2020). Physical features of dams, such as turbines and sluiceways, have resulted in increased mortality of both adults and juvenile salmonids (Alaska Department of Fish and Game, 2024). In addition, coho salmon predators include marine mammals such as orcas, sea lions and sea otters, as well as other fishes such as sharks.

C.5.2.2.2 Southern Oregon and Northern California Coast ESU

C.5.2.2.2.1 Status and Management

The Southern Oregon and Northern California Coast Coho (SONCC) Salmon ESU was listed as threatened on May 6, 1997 (62 FR 24588); that status was reaffirmed on June 28, 2005 (70 FR 37160) and subsequently updated on April 14, 2014 (79 FR 20802). This ESU includes naturally spawned coho salmon originating from coastal streams and rivers between Cape Blanco, Oregon, and Punta Gorda, California, as well as coho salmon from three artificial propagation programs (79 FR 20802). NMFS has not proposed any significant changes to the hatchery listing status for this ESU in their 2016 proposed rule (81 FR 72759).

Critical Habitat

NMFS designated critical habitat for the SONCC ESU of coho salmon on May 5, 1999 (64 FR 24049). Critical habitat includes juvenile summer and winter rearing areas, juvenile migration corridors, areas for growth and development to adulthood, adult migration corridors, and spawning areas. The physical or biological features that characterize these sites include substrate, water quality, water quantity, water temperature, water velocity, cover/shelter, food, riparian vegetation, space, and safe passage conditions. Designated critical habitat for the SONCC ESU of coho salmon does not overlap spatially with the HCTT Study Area. The only ESA-listed coho salmon with critical habitat designated within 50 NM of the HCTT Study Area is the Central California Coast Coho ESU (Figure C-5).

Recovery Goals

See the 2014 recovery plan for detailed descriptions of the recovery goals and delisting criteria for Southern Oregon and Northern California Coast coho salmon (National Marine Fisheries Service, 2014e).

The following goals guide recovery of SONCC coho salmon as described in recovery documents from the State of Oregon, the State of California, and NMFS.

- First, each SONCC population must reach desired levels of biological viability and the recovery effort must sufficiently reduce the impact of the stresses and threats in order to warrant removal of the SONCC coho salmon ESU from the threatened and endangered species list (referred to in this plan as either delisting or ESA recovery).
- Second, the States of California and Oregon seek to rebuild wild populations to reach 'broad sense recovery' to provide for sustainable fisheries and other ecological, cultural, and social benefits.

C.5.2.2.2.2 Habitat and Geographic Range

Geographic range of spawning systems for this ESU: Naturally spawned coho salmon originating from coastal streams and rivers between Cape Blanco, Oregon, and Punta Gorda, California. This ESU includes coho from three artificial propagation programs, the Cole Rivers Hatchery Program, the Trinity River Hatchery Program, and the Iron Gate Hatchery Program (79 FR 20802).

Juvenile outmigrants

A number of the stream systems where the SONCC coho spawn are small and subject to highly variable flow regimes. As adults delay migrating into these systems until flows are sufficient, adult run-timing and spawning are highly variable within and between these coastal systems (Moyle et al., 2017). Fry typically emerge from the gravel in spring (National Marine Fisheries Service, 2014c). However, in larger watersheds like the Klamath River, fry merge from the gravel from February through July, with typical peaks in March and April. Juveniles may rear in streams or estuaries for at least a year, with some fish remaining in freshwater habitats through age-3 (Crozier et al., 2019; National Marine Fisheries Service, 2014c). Smolts migrate to the ocean as young of the year, one-year-old, or two-year-old fish (Moyle et al., 2017), typically enter marine waters in spring and summer (NMFS 2014), with peak outmigration occurring in April or May (Moyle et al., 2017; Weitkamp et al., 1995).

Ocean rearing

Upon initially entry into marine waters, SONCC coho remain within nearshore coastal waters (Crozier et al., 2019; Moyle et al., 2017; National Marine Fisheries Service, 2014c). Moyle (2002) suggests this may be due to high densities of available food resources. However, once moving away from these coastal waters, the marine distribution of SONCC coho is poorly understood (Crozier et al., 2019; Moyle et al., 2017). In general, coho distribution while at sea is wide-ranging, with distributions from California to the North Pacific (Moyle et al., 2017). The majority of coho salmon caught in California marine fisheries originate from Oregon systems, not the California stocks (Moyle et al., 2017). CWT coho from this region display a more southerly marine distribution than populations to the north (Weitkamp et al., 1995). The region-specific recovery of tags by percentages for SONCC coho as reported by Weitkamp et al. (1995) is California (65–92 percent), with some recoveries in Oregon (7–34 percent) and almost none (<1 percent) in Washington or British Columbia. During NMFS summer salmon trawl surveys conducted from 2010–2014, coho captured in California waters were much more abundant in more northern California waters in the Klamath-Trinidad region than the Lost Coast region, with very few coho captured in the Gulf of Farallones region (Harding, 2015). Coho captured during these surveys occurred at lower abundance for deeper offshore trawl stations relative to shallower stations (Harding, 2015).

Return migration

Following 6–24 months at sea, SONCC coho return to their natal systems as age-3 fish (Weitkamp et al., 1995; (Moyle et al., 2017) NMFS 2014). Mature adult coho will wait in nearshore marine environments for suitable conditions to allow their upstream migration. Depending on fall rains and coastal river water levels, mature coho from this ESU typically enter coastal estuaries from October to March, with peak migration occurring from mid-November to January (Crozier et al., 2019; Moyle et al., 2017; National Marine Fisheries Service, 2014c). However, for some larger system like the Klamath River, the migration period is earlier and longer, extending from late-August to mid-January (National Marine Fisheries Service, 2014c).

C.5.2.2.2.3 Population Trends

The updated 2016 status review indicates that there has been no improvement in the status of SONCC coho salmon in the last five years (National Marine Fisheries Service, 2016h). The SONCC coho salmon ESU continues to be at risk of extinction. Twenty-four out of 31 independent populations are at high risk of extinction, six are at moderate risk of extinction, and none is at low risk of extinction. All core populations (those intended to serve as anchors for recovery) are thousands of adults short of the numbers needed for them to play their role in recovery of the entire ESU (National Marine Fisheries Service, 2016h).

C.5.2.2.2.4 Population Threats

Population threats are similar across many coho salmon ESUs. Please see the description of these threats described for the Oregon Coast Coho Salmon ESU.

C.5.2.2.3 Central California Coast ESU

C.5.2.2.3.1 Status and Management

The Central California Coast Coho Salmon ESU was listed as threatened on October 31, 1996 (61 FR 56138) and downgraded to endangered on June 28, 2005 (70 FR 37160). The ESU status was reaffirmed as endangered on April 2, 2012 (77 FR 19552) and subsequently updated on April 14, 2014 (79 FR 20802). This ESU includes naturally spawned coho salmon originating from rivers south of Punta Gorda, California to and including Aptos Creek, coho salmon originating from tributaries to San Francisco Bay, as well as coho salmon from three artificial propagation programs (79 FR 20802). NMFS has not proposed any changes to the hatchery listing status for this ESU in their 2016 proposed rule (81 FR 72759).

Critical Habitat

Critical habitat for the Central California Coast ESU of coho salmon was designated on May 5, 1999 (64 FR 24049). Critical habitat includes juvenile summer and winter rearing areas, juvenile migration corridors, areas for growth and development to adulthood, adult migration corridors, and spawning areas. The physical or biological features that characterize these sites include substrate, water quality, water quantity, water temperature, water velocity, cover/shelter, food, riparian vegetation, space, and safe passage conditions. Designated critical habitat for the Central California Coast ESU of coho salmon does not overlap spatially with the HCTT Study Area, but does occur in freshwater systems to the east of the northern portion of the California Study Area (Figure C-5).

Recovery Goals

See the 2012 recovery plan for complete down listing/delisting criteria for each of the following recovery goals for Central California Coast coho salmon (National Marine Fisheries Service, 2012b):

- Prevent extinction by protecting existing populations and their habitats,
- Maintain current distribution of coho salmon and restore their distribution to previously occupied areas essential to their recovery,
- Increase abundance of coho salmon to viable population levels, including the expression of all life history forms and strategies,
- Conserve existing genetic diversity and provide opportunities for interchange of genetic material between and within meta populations,
- Maintain and restore suitable freshwater and estuarine habitat conditions and characteristics for all life history stages so viable populations can be sustained naturally,
- Ensure all factors that led to the listing of the species have been ameliorated, and
- Develop and maintain a program of monitoring, research, and evaluation that advances understanding of the complex array of factors associated with coho salmon survival and recovery and which allows for adaptively managing our approach to recovery over time.

C.5.2.2.3.2 Habitat and Geographic Range

Geographic range of spawning systems for this ESU: Naturally spawned coho salmon originating from rivers south of Punta Gorda, California to and including Aptos Creek, as well as such coho salmon originating from tributaries to San Francisco Bay. This DPS includes steelhead from three artificial propagation programs: the Don Clausen Fish Hatchery Captive Broodstock Program, the Scott Creek/King Fisher Flats Conservation Program, and the Scott Creek Captive Broodstock Program (79 FR 20802).

Juvenile outmigrants

Central California Coast coho fry emerge from February through June, with peak emergence occurring from March to May (National Marine Fisheries Service, 2012b). Juveniles may rear in streams or estuaries for at least a year, with some fish remaining in freshwater habitats through age-3 (Crozier et al., 2019; National Marine Fisheries Service, 2012b). Typically, coho smolt outmigration begins in March, with peak outmigration occurring from April to early July (Weitkamp et al., 1995). However, several of the estuaries utilized by this ESU are blocked by sandbars during low water flows. As a result, the outmigration timing and age of Central California Coast coho smolts migrating to marine habitats can be highly variable and dependent on sufficient water levels (Crozier et al., 2019).

Ocean rearing

Detections of Central California Coast coho in the marine environment are limited. However, tagging studies elsewhere have indicated that juvenile fish likely reside in nearshore marine waters relatively close to their natal system (Crozier et al., 2019; National Marine Fisheries Service, 2012b; Pearcy & Fisher, 1988; Quinn & Myers, 2005; Shapovalov & Taft, 1954; Weitkamp & Neely, 2002). Shapovalov and Taft (1954) found that younger Central California Coast coho stayed within approximately 90 mi of the coastline. As fish from this ESU mature and become larger they display a much broader offshore marine environment and have been found off the coast of Oregon (National Marine Fisheries Service, 2012b; Weitkamp & Neely, 2002; Weitkamp et al., 1995).

In an effort to better understand the distribution of coho salmon from this region, beginning in 2003, Hayes et al. (2011), and others, tagged and monitored 579 Scott Creek Fish Hatchery coho smolts at least 200 mm in length (7.8 in.) with archival tags. Scott Creek is located approximately 60 mi south of San Francisco. Although the tagged fish were not included within the Central California Coast coho ESU, these hatchery fish originate from the same geographic area. These tags provided estuarine distribution, temperature, and even predation data, however, likely due to poor ocean conditions and large tag size relative to small smolt size, only one of these tags returned data indicating that the fish had entered marine waters. This lone tagged fish remained in the marine environment until it was recaptured approximately 15 mi. to the southeast of Scott Creek, near Santa Cruz six months after it was tagged and released.

During NMFS summer salmon trawl surveys conducted from 2010–2014, coho captured in California waters were much more abundant in more northern California waters in the Klamath-Trinidad region than the Lost Coast region, with very few coho captured in the Gulf of Farallones region (Harding, 2015). Coho captured during these surveys occurred at lower abundance for deeper offshore trawl stations relative to shallower stations (Harding, 2015).

Return migration

As Central California Coast coho need to mill outside of the mouth of the natal estuaries until water flows and conditions are suitable for upstream migration, their run-timing is later than for coho populations further north (Weitkamp et al., 1995). Adults begin entering these estuaries as early as late-September and as late as February, with a peak migration occurring in November to January (National Marine Fisheries Service, 2012b; Shapovalov & Taft, 1954; Weitkamp et al., 1995). Very little time is spent between river entry and spawning activity (Weitkamp et al., 1995).

C.5.2.2.3.3 Population Trends

Prior viability and stock assessments (Good et al., 2005; Spence & Williams, 2011; Williams et al., 2011; Williams et al., 2016b) indicated that all independent and dependent populations of this ESU were well below recovery targets and, in some cases, exceed high-risk thresholds established by Spence et al. (2008), with an area of particular concern being the downward trends in abundance of virtually all dependent populations across all diversity strata. Despite conservation efforts, conditions for Central California Coast coho salmon continue to decline (National Marine Fisheries Service, 2016a, 2023a), and the threats to recovery have, with few exceptions, remained unchanged since prior reviews (National Marine Fisheries Service, 2016a, 2023a). NMFS (2023a) determined that the viability of populations is progressively worse moving north to south within the Central California Coast Coho ESU. The best available information on the biological status of this ESU and the threats facing this ESU indicate that it continues to remain endangered.

Over recent years the distribution and abundance of coho salmon populations in California have been considerably reduced (California Department of Fish and Wildlife, 2024a). Although coho salmon are still found in most major river systems in the northern portion of the state, many spawning runs have declined substantially in size and were eliminated from many tributaries, including some streams in the Klamath and Eel River basins (California Department of Fish and Wildlife, 2024a).

Overall, from Humboldt County north to the Oregon border, coho salmon are now found in approximately two-thirds of the streams identified as historical habitat (California Department of Fish and Wildlife, 2024a). In the southern part of their range, coho salmon are now absent from all tributaries of San Francisco Bay and many streams south of the Bay; this is likely associated with adverse effects from increased urbanization and other human developments on watersheds and fish habitat (California Department of Fish and Wildlife, 2024a).

C.5.2.2.3.4 Population Threats

Population threats are similar across many coho salmon ESUs. Please see the description of these threats described for the Oregon Coast Coho Salmon ESU.

C.5.2.3 Steelhead (*Oncorhynchus mykiss*)

Five ESA-listed DPSs of steelhead have the potential to occur in the Study Area, including the Northern California DPS, California Central Valley DPS, Central California Coast DPS, South-Central California Coast and Southern California DPS.

C.5.2.3.1 Northern California DPS

C.5.2.3.1.1 Status and Management

The Northern California Steelhead DPS was listed as threatened on June 7, 2000 (65 FR 36074) and their status was updated on January 5, 2006 (71 FR 834) and April 14, 2014 (79 FR 20802). This DPS includes naturally spawned steelhead originating below natural and manmade impassable barriers in California coastal river basins from Redwood Creek to and including the Gualala River (79 FR 20802). No hatchery programs are currently included as part of this DPS.

<u>Critical Habitat</u>

NMFS designated critical habitat for Northern California DPS steelhead on September 2, 2005 (70 FR 52488). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Northern California DPS steelhead does not overlap spatially with the HCTT Study Area, however these habitats do occur in freshwaters to the north and east and the northernmost portion of the California Study Area (Figure C-6).

<u>Recovery Goals</u>

See the 2016 Recovery Plan for the Northern California Steelhead DPS for complete down-listing/delisting criteria for recovery goals for the DPS (National Marine Fisheries Service, 2016l). The recovery plan's objectives are to: (1) Reduce the present or threatened destruction, modification, or curtailment of habitat or range; (2) Ameliorate utilization for commercial, recreational, scientific, or educational purposes; (3) Abate disease and predation; (4) Establish the adequacy of existing regulatory mechanisms for protecting NC steelhead now and into the future (i.e., post-delisting); 5. Address other natural or manmade factors affecting the continued existence of NC steelhead; and 6. Ensure NC steelhead status is at a low risk of extinction based on abundance, growth rate, spatial structure and diversity.

C.5.2.3.1.2 Habitat and Geographic Range

Geographic range of spawning systems for this DPS: Naturally spawned anadromous steelhead originating below natural and manmade impassable barriers in California coastal river basins from Redwood Creek to and including the Gualala River (79 FR 20802). The Northern California steelhead DPS includes both summer- and winter-run fish (79 FR 20802), with winter-run fish being more common (Moyle et al., 2017).

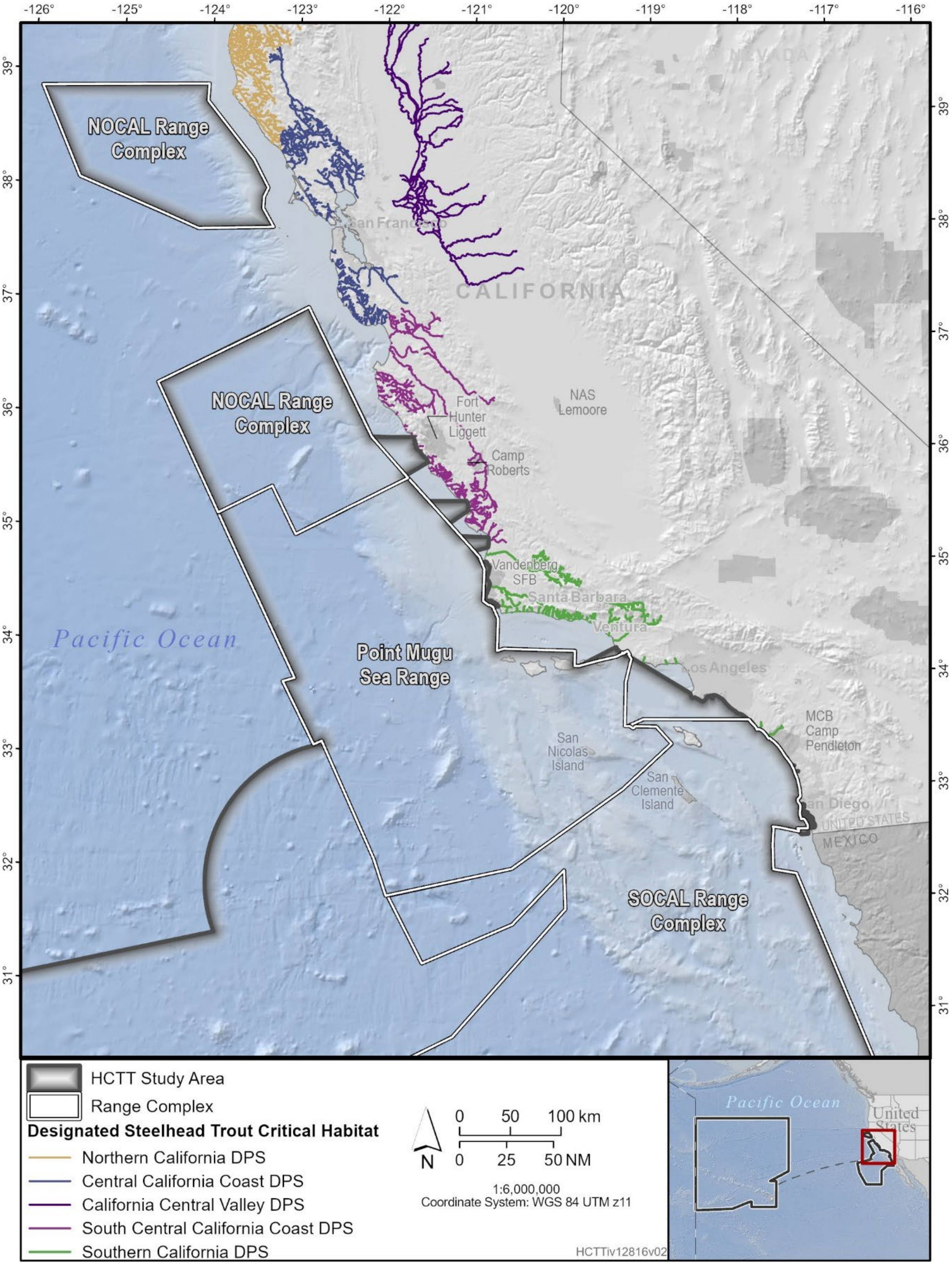

Figure C-6: Designated Steelhead Critical Habitat

Juvenile outmigrants

The designated Northern California Steelhead DPS combines numerous life history strategies into a single ESA-listed DPS (Moyle et al., 2017). In addition to being comprised of both winter-run and summer-run ecotype fish, there is considerable variability within each run designation. Much of this diversity comes numerous smaller systems that include both coastal and mountain streams, and distinct, isolated populations (Moyle et al., 2017). As Northern California steelhead spawning occurs over an expansive time period, so does fry emergence (Moyle et al., 2017; National Marine Fisheries Service, 2016i). Fry rear in river systems for months to years before migrating to estuaries as smolts. Smolts typically leave the estuaries and migrate to marine environments between March and June, though if conditions are poor, they may wait until late fall (Moyle et al., 2017). Outmigrating kelts (adult steelhead that have spawned and are migrating back downstream to the ocean) from these systems also typically leave freshwater and re-enter the marine system by late spring (Moyle et al., 2017). When these fish leave their natal systems and migrate to marine habitats, marine entry occurs to the north and east of the northernmost portion of the California Study Area (Figure C-6).

Ocean rearing

As steelhead are infrequently captured in the marine environment, DPS-specific marine distribution for steelhead is limited. Steelhead, as a species, are broadly distributed across the North Pacific, ranging from California to Alaska in the Northeast Pacific, and west towards Russia in the Northwest Pacific (Myers et al., 1998). Further, less is known about the ocean rearing phase of California-origin steelhead than of those originating from river systems further north.

Burgner et al. (1992) conducted a review of steelhead documented in nearshore coastal waters. They found that in Oregon and Washington, young, age-0 steelhead that had recently migrated to the ocean were most abundant in the nearshore marine environment in May and were detected further offshore and further north in June and July. However, in northern Californian waters, juvenile steelhead were detected in the nearshore marine environment in July, but quickly moved offshore, with no age-0 fish caught in southern Oregon or northern California by August. Harding (2015), and others, noted captures of juvenile steelhead both north and south of Cape Blanco, Oregon in August (Moyle et al., 2017).

Burgner et al. (1992) also noted that, although approximately 1.2M California-origin steelhead were CWT and released between 1980–1988, no California CWT-tagged fish were recovered at sea. However, nine disk-tagged California-origin steelhead were captured in the Gulf of Alaska (Burgner et al., 1992). These authors suggest that California-origin steelhead may have a more westerly (as opposed to northerly) distribution than steelhead originating from more northern river systems. However, Myers et al. (1996) noted that five maturing California-origin steelhead were detected in the Gulf of Alaska, with Myers (2018) stating as many as nine maturing California-origin steelhead were detected in these waters.

Harding et al., (2021) analyzed summer 20102014 salmonid trawl surveys conducted along 16 east-west transect lines with five locations surveyed along each transect; from the northernmost location at Heceta Head, Oregon, south to Pigeon Point (Bodega Bay), California. They found that both juvenile and subadult steelhead catch per unit effort (CPUE) was greatest between Klamath River and the Eel River, and rare south of Cape Mendocino. Both juvenile and subadult steelhead occurred more frequently at offshore stations than Chinook or coho salmon (Harding et al., 2021). Juvenile steelhead were detected most frequently at locations with mean water depths from 55 to 122 m but were relatively absent in the catch at the most nearshore locations (mean depth of 30 m) and furthest offshore locations (mean depth of 370 m). Subadult steelhead occurred most frequently at the furthest offshore locations and were infrequently captured at the most nearshore locations.

After the first month or two, juvenile steelhead move north and west away from the coastline and begin maturing. These older steelhead are difficult to capture as they are non-schooling, surface-oriented fish, with a very broad at-sea distribution. Relatively few older fish have been captured at-sea, though some authors reviewing capture data suggest that California-origin steelhead may be more abundant west, than north, though no comparative abundance studies were found in the literature. More recent pop-up satellite archival tags studies investigating marine occupancy of Alaska-origin kelts have shown that these fish tend to occupy ocean surface habitats directly over the shelf slope, where upwelling likely increases available food resources (Courtney et al., 2022; Seitz & Courtney, 2021).

While migrating and rearing in marine waters, steelhead are generally-surface oriented. Moore and Berejikian (2022) used acoustic tags with depth and temperature sensors to monitor the behavior of 498 steelhead smolts in the presence of a Puget Sound floating bridge. In the absence of the bridge, steelhead smolts were considered “nearly exclusively surface-oriented.” Only when encountering the bridge did these fish occasionally dive to avoid the migration barrier and fish predators that congregate there. In a radio and sonic tagging study of the marine migratory behavior of 19 adult steelhead in a British Columbia fjord, Ruggerone et al. (1990) found that these fish spent the majority of time in the top two meters of the water column. In a pop-up satellite archival tag (PSAT) study of 16 steelhead kelts from the Situk River, Alaska, Seitz and Courtney (2021) found that these fish, once exiting this southeast Alaska river and entering the North Pacific Ocean, moved west of the continental slope, typically occurring in the top five meters of the water column, though occasionally diving as deep as 20 m.

Return migration

Northern California steelhead spend from 1 to 4 years rearing in the ocean prior to returning to their natal systems (Moyle et al., 2017). The diversity of having both ocean-maturing and stream-maturing fish within this DPS results in adult entry, at the DPS level, occurring year-round, with return timing tied to both system origin and life history type (Myers, 2018). Summer-run steelhead from this DPS return to estuaries between April and June in the northernmost systems for this DPS, between April and July in the Mad River, and between March and June in the Mattole River (Moyle et al., 2017). In larger rivers (e.g., Mad and Eel Rivers), winter-run adults leave the marine environment and enter estuaries as early as September of October. In smaller systems, enter the estuaries from December to May (Busby et al., 1996).

C.5.2.3.1.3 Population Trends

Overall, the available data for winter-run populations—predominately in the North Coastal, North-Central Coastal, and Central Coastal strata—indicate that all populations are well below viability targets, most being between 5 percent and 15 percent of these goals (National Marine Fisheries Service, 2024a). There is a mix in trends regarding the longer and shorter time series. Most supporting independent populations have shown downward (but non-significant) trends (National Marine Fisheries Service, 2024a).

Summer-run populations continue to be of significant concern. While Middle Fork Eel River population is closer to its 80 percent recovery target, other populations range from as low as 18–26 percent of their recovery target, or there is a lack of data to estimate percent recovery(National Marine Fisheries Service, 2024a). In summary, the available information for winter-run and summer-run populations of Northern California steelhead do not suggest an appreciable increase or decrease in extinction risk since publication of the last status reviews and viability assessments. Although conservation efforts have reduced some threats facing this DPS, the threats have remained unchanged since the last review (National Marine Fisheries Service, 2024a). Poor ocean conditions, water withdrawals, marijuana cultivation and drought, in particular, have significant negative impacts on Northern California steelhead since the last review. In summary, the best available updated information on the biological status of and threats to Northern California steelhead DPS indicate it continues to remain a threatened species.

C.5.2.3.1.4 Population Threats

Most of the threats to steelhead occur outside the Study Area and include alteration of stream flow patterns and habitat degradation, barriers to fish passage, channel alterations, water quality problems, non-native fishes and plants, and climate change.

In addition, steelhead predators include seabirds, such as terns and cormorants, and marine mammals, such as sea lions and harbor seals (National Marine Fisheries Service, 2010g). Juveniles in freshwater feed mostly on zooplankton (small animals that drift in the water), while adults feed on aquatic and terrestrial insects, molluscs, crustaceans, fish eggs, minnows, and other small fishes, including other trout and salmon depending on whether they are inhabiting streams or the ocean (National Marine Fisheries Service, 2010g).

C.5.2.3.2 California Central Valley DPS

C.5.2.3.2.1 Status and Management

The California Central Valley Steelhead DPS was listed as threatened on March 19, 1998 (63 FR 13347) and their status was updated on January 5, 2006 (71 FR 834). This DPS includes naturally spawned steelhead originating below natural and manmade impassable barriers from the Sacramento and San Joaquin Rivers and their tributaries; excludes such fish originating from San Francisco and San Pablo Bays and their tributaries. In 2020, NMFS issued a final rule (85 FR 81822) adding the Mokelumne River Hatchery Program to this DPS which increased the total number of hatchery programs to three.

Critical Habitat

NMFS designated critical habitat for Central Valley DPS steelhead on September 2, 2005 (70 FR 52488). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Central Valley DPS steelhead does not overlap spatially with the HCTT Study Area (Figure C-6).

Recovery Goals

See the 2014 Recovery Plan for the California Central Valley Steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species (National Marine Fisheries Service, 2014b). The delisting criteria for this DPS are:

- One population in the Northwestern California Diversity Group at low risk of extinction,
- Two populations in the Basalt and Porous Lava Flow Diversity Group at low risk of extinction,
- Four populations in the Northern Sierra Diversity Group at low risk of extinction,
- Two populations in the Southern Sierra Diversity Group at low risk of extinction, and
- Maintain multiple populations at moderate risk of extinction.

C.5.2.3.2.2 Habitat and Geographic Range

Geographic range of spawning systems for this DPS: Naturally spawned anadromous steelhead originating below natural and manmade impassable barriers from the Sacramento and San Joaquin Rivers and their tributaries; excludes such fish originating from San Francisco and San Pablo Bays and their tributaries. This DPS includes steelhead from three artificial propagation programs: the Coleman National Fish Hatchery Program, the Feather River Fish Hatchery Program, and the Mokelumne River Hatchery Program(79 FR 20802; 85 FR 81822).

Juvenile outmigrants

The remaining populations of California Central Valley steelhead are winter-run fish (Moyle, 2002; National Marine Fisheries Service, 2014e). California Central Valley steelhead fry emerge from the gravel in spring. Scale analysis indicates that 70 percent of juveniles from this DPS reared in freshwater systems for 2 years prior to migrating to the ocean (McEwan, 2001). This DPS, more so than more northern populations of steelhead, appears to have a higher proportion of fish that remain resident, and forego migrating to marine habitats (Crozier et al., 2019). Downstream migration of naturally-spawned juvenile steelhead from this DPS typically occurs from December to May, with a peak outmigration in March, with another small peak in fall (McEwan, 2001). When these fish leave their natal systems and migrate to marine habitats, marine entry occurs to the east of the northernmost portion of the California Study Area (Figure C-6).

Ocean rearing

Natural-spawning steelhead from this DPS are believed to rear in the ocean for 1–2 years, whereas hatchery fish typically spend 1–3 years in the marine environment (Crozier et al., 2019). As steelhead are infrequently captured in the marine environment, DPS-specific marine distribution for steelhead is largely unavailable. Therefore, with the additional DPS-specific finding from one fish in Teo et al. (2013) below, the ocean rearing description for Northern California steelhead would also be applicable for this DPS.

Teo et al. (2013) tagged 14 Coleman National Fish Hatchery kelts (11 females and 3 males) from this DPS with acoustic and archival tags. Tagged fish ranged in size from 17 to 20 in. Of these 14 fish, only one exited estuarine waters and was detected in coastal California waters. From May to August 2008, this fish ranged from as far north as offshore of Eureka and as far south as due west of Big Sur. This fish spent most of the time within 50 NM of the coast, though on a few occasions it was detected beyond 100 NM from shore (Teo et al., 2013).

Return migration

Adult California Central Valley steelhead return through the San Francisco estuary from August through April (Myers, 2018; National Marine Fisheries Service, 2014e) and are detected in their natal streams from late September through early April (Lindley et al., 2006; National Marine Fisheries Service, 2014e).

C.5.2.3.2.3 Population Trends

Continued decline in the ratio between naturally produced juvenile steelhead to hatchery juvenile steelhead in fish monitoring efforts indicates that the wild population abundance is declining (National Marine Fisheries Service, 2016c, 2023e; Williams et al., 2016a). Hatchery releases have remained relatively constant over the past decade, yet the proportion of adipose fin-clipped hatchery smolts to unclipped naturally produced smolts has steadily increased over the past several years. One continuing strength of this DPS is its widespread distribution throughout the Central Valley. While most of the measured populations are small, steelhead can be found in most of the major rivers and streams of the Sacramento River, San Joaquin River, and eastside tributaries including the Mokelumne River and Calaveras River. Although there have been recent restoration efforts in the San Joaquin River tributaries, California Central Valley steelhead populations in the San Joaquin Basin continue to show an overall very low abundance, and fluctuating return rates. The widespread distribution of wild steelhead in the Central Valley provides the spatial structure necessary for the DPS to survive and avoid localized catastrophes. However, most wild California Central Valley populations may lack the resiliency to persist for protracted periods if subjected to additional stressors, particularly widespread stressors such as climate change and drought (National Marine Fisheries Service, 2014d).

The genetic diversity of California Central Valley steelhead has likely been impacted by low population sizes and high numbers of hatchery fish relative to wild fish (Goetz et al., 2024; Huber et al., 2024). The best chance for eventual delisting of this species is expansion of their range, as it was the creation of dams that has removed them from over 80 percent of their original spawning and rearing habitat in the Central Valley. This species has clearly benefited from the removal of Saeltzer Dam on Clear Creek, resulting in one of the strongest steelhead populations in the Central Valley. NMFS concluded that California Central Valley steelhead remain listed as threatened, as the DPS is likely to become endangered within the foreseeable future throughout all or a significant portion of its range (National Marine Fisheries Service, 2016c, 2023e; Williams et al., 2016a).

C.5.2.3.2.4 Population Threats

Population threats are similar across many steelhead populations. Please see the description of these threats described for the Northern California Steelhead DPS.

C.5.2.3.3 Central California Coast DPS

C.5.2.3.3.1 Status and Management

The Central California Coast Steelhead DPS was listed as threatened on August 18, 1997 (62 FR 43937) and their status was reaffirmed on January 5, 2006 (71 FR 834) and updated on April 14, 2014 (79 FR 20802). This DPS includes naturally spawned steelhead originating below natural and manmade impassable barriers from the Russian River to and including Aptos Creek, and all drainages of San Francisco and San Pablo Bays eastward to Chipps Island at the confluence of the Sacramento and San Joaquin Rivers. This DPS also includes steelhead from two artificial propagation programs (79 FR 20802). There are no proposed changes in hatchery programs included in this DPS (81 FR 72759).

Critical Habitat

NMFS designated critical habitat for Central California Coast DPS steelhead on September 2, 2005 (70 FR 52488). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Central California Coast DPS steelhead does not overlap spatially with the HCTT Study Area (Figure C-6).

Recovery Goals

See the 2016 recovery plan for the Central California Coast Steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species (National Marine Fisheries Service, 2016l). The recovery plan's objectives are to: 1) Reduce the present or threatened destruction, modification, or curtailment of habitat or range; 2) Ameliorate utilization for commercial, recreational, scientific, or educational purposes; 3) Abate disease and predation; 4) Establish the adequacy of existing regulatory mechanisms for protecting CCC steelhead now and into the future (i.e., post-delisting); 5) Address other natural or manmade factors affecting the continued existence of CCC steelhead; and 6) Ensure CCC steelhead status is at a low risk of extinction based on abundance, growth rate, spatial structure and diversity.

C.5.2.3.3.2 Habitat and Geographic Range

Geographic range of spawning systems for this DPS: Naturally spawned anadromous steelhead originating below natural and manmade impassable barriers from the Russian River to and including Aptos Creek, and all drainages of San Francisco and San Pablo Bays eastward to Chipps Island at the confluence of the Sacramento and San Joaquin Rivers. This DPS includes steelhead from two artificial propagation programs: the Don Clausen Fish Hatchery Program and the Kingfisher Flat Hatchery Program (Monterey Bay Salmon and Trout Project) (79 FR 20802).

Juvenile outmigrants

Central California Coast steelhead exhibit a broad range of life history patterns (Moyle et al., 2017). As indicated by Shapovalov and Taft (1954), as many as 32 different combinations of life history patterns from juvenile emergence, residency, and migration to adult spawning and migration occurs within steelhead in this geographic region. However, Central California Coast steelhead life history can be generally grouped into three categories: anadromous, freshwater resident, lagoon-anadromous (Bond, 2006; Moyle et al., 2017). All steelhead populations have the potential to produce both anadromous and freshwater resident fish. However, it's not until reaching the more southern populations of steelhead do they display the third variation to adapt to seasonal migrational barriers. As a result, juvenile steelhead, instead of maturing in marine environments, grow and mature in lower stem lagoons (Bond, 2006). These fish can then return as mature/maturing adults to upstream habitats suitable for spawning.

Following the late-spring spawning of adults, Central California Coast steelhead fry emerge from the gravel 5–7 weeks later. Like many steelhead populations, juveniles from this DPS may rear in freshwater systems for 1–3 years, though approximately 57 percent of Central California Coast steelhead remain in freshwater for two years before migrating downstream to towards the ocean (review within Crozier et al., (2019)). Larger smolts move directly to sea in the spring (Osterback et al., 2018), whereas smaller smolts rear in estuaries for an extended period (Bond, 2006). These "lagoon-anadromous life history" steelhead grow larger in freshwater lagoons and contribute to a larger proportion of adult spawners (Bond, 2006). Peak outmigration for anadromous variants occurs in April and May (Crozier et al., 2019) and peak outmigration for lagoon-anadromous occurs during winter storms (Bond, 2006). When these fish leave their natal systems and migrate to marine habitats, marine entry occurs to the east of the northernmost portion of the California Study Area (Figure C-6).

Ocean rearing

A literature review by Crozier et al. (2019) indicated that approximately 70 percent of spawning adults from this DPS spent 1–2 years maturing in the marine environment. However, as steelhead are infrequently captured in the marine environment, DPS-specific marine distribution for steelhead is largely unavailable. Therefore, with the additional DPS-specific findings from Hayes et al. (Bond et al., 2008; Hayes et al., 2011; Huber et al., 2024; Osterback et al., 2018), below, the ocean rearing description for Northern California steelhead would also be applicable for this DPS (Bond et al., 2008; Hayes et al., 2011; Huber et al., 2024; Osterback et al., 2018).

From 2003 to 2008, Hayes et al. (2011) tagged and monitored steelhead at the Scott Creek Fish Hatchery with archival tags. Scott Creek is located approximately 60 mi. south of San Francisco. Although not included within the Central California Coast steelhead DPS, these hatchery fish originate from the same geographic area. They tagged a total of 319 steelhead, of these, 69 were attached to adult female steelhead (kelts) with the remaining tags deployed on smolts. To reconstruct at-sea travel range estimates, the authors conducted a literature review and ascertained that, while at sea. Steelhead display a mean swim speed of 15 mi./day, and a maximum of 53 mi./day, resulting in a 40 mi./day estimate for modeling (two standard deviations above the mean distance). Tag return rates were low, less than 1 percent for smolts and 2 percent for steelhead kelts. Only three of these tags indicated outmigration into the ocean; one smolt and two kelts. The archival tags indicated substantial variability. In 2004, one hatchery juvenile and one wild kelt generally occupied waters west of California and southern Oregon. However, the tags attached to a 2004 wild kelt and a 2007 hatchery kelt indicated that these fish occupied the Gulf of Alaska and North Pacific. Ideal sea surface temperature was thought to play a role in the broad marine distribution of steelhead.

Return migration

Steelhead originating from more northern streams within the Central California Coast Steelhead DPS typically return to spawn as age-2+ adults, whereas fish returning to more southern systems (e.g., Russian River) return as age 3+ adults (Moyle et al., 2017). Generally, adult steelhead from this DPS leave the marine environment and migrate upstream from October to May (Busby et al., 1996; Myers, 2018; Osterback et al., 2018), with peak entry occurring between November and February (Moyle et al., 2017), and spawning occurring in late spring.

C.5.2.3.3.3 Population Trends

In the North Coastal and Interior strata, steelhead still appear to occur in the majority of watersheds, though in the Russian River basin, the ratio of hatchery fish to natural origin fish returning to spawn remain largely unknown and continues to be a source of concern (Williams et al., 2016a). New information in the Santa Cruz Mountain stratum indicates that population sizes are perhaps higher than previously thought. However, the downward trend in the Scott Creek population, which has the most robust estimates of abundance, is a source of concern. The status of populations in the two San Francisco Bay diversity strata remains highly uncertain, and it is likely that many populations where historical habitat is now inaccessible due to dams and other passage barriers are at high risk of extinction. In summary, while data availability for this DPS remains poor, there is little new evidence to suggest that the extinction risk for this DPS has changed appreciably in either direction since publication of the last status review in 2010 (Williams et al., 2016a).

C.5.2.3.3.4 Population Threats

Population threats are similar across many steelhead populations. Please see the description of these threats described for the Northern California Steelhead DPS.

C.5.2.3.4 South-Central California Coast DPS

C.5.2.3.4.1 Status and Management

The South-Central California Coast Steelhead DPS was listed as threatened on August 18, 1997 (62 FR 43937) and their status was reaffirmed on June 28, 2005 (70 FR 37160) and updated on April 14, 2014

(79 FR 20802). This DPS includes naturally spawned steelhead originating below natural and manmade impassable barriers from the Pajaro River to (but not including) the Santa Maria River (79 FR 20802). No hatchery programs are currently included as part of this DPS.

Critical Habitat

NMFS designated critical habitat for South-Central California Coast DPS steelhead on September 2, 2005 (70 FR 52488). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the South-Central California Coast DPS steelhead only includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas (70 FR 52488), not marine habitats, so it does not overlap spatially with the HCTT Study Area (Figure C-6).

Recovery Goals

See the 2013 recovery plan for the South-Central California Coast Steelhead DPS (National Marine Fisheries Service, 2013b) for complete down-listing/delisting criteria for recovery goals for the species. The Recovery Plan outlines the following coordinated actions that will be required for an effective steelhead recovery program:

- Prevent steelhead extinction by protecting existing populations and their habitats,
- Maintain current distribution of steelhead and restore distribution to some previously occupied areas,
- Increase abundance of steelhead to viable population levels, including the expression of all life-history forms and strategies,
- Conserve existing genetic diversity and provide opportunities for interchange of genetic material between and within viable populations within the DPS,
- Restore and maintain suitable habitat conditions and characteristics for all life-history strategies, thereby preserving the diversity of life-history strategies that allow for adaptation to a highly variable environment, and
- Conduct necessary research to refine recovery criteria, monitor the status and trends of individual populations, and adaptively modify recovery actions and strategies in response to new information and better understanding of the biology and habitat requirements of the species.

C.5.2.3.4.2 Habitat and Geographic Range

Geographic range of spawning systems for this DPS: Naturally spawned anadromous steelhead originating below natural and manmade impassable barriers from the Pajaro River to (but not including) the Santa Maria River (79 FR 20802).

As described for Central California Coast steelhead, South-Central California Coast and Southern California steelhead have also evolved to have anadromous, freshwater resident, lagoon-anadromous forms to ensure annual spawning occurs, even in the presence of downstream barriers to migration. To maintain occupancy within suitable habitat conditions, the two southernmost steelhead DPS’s display greater range movement within and between freshwater habitats within a given system (Moyle et al., 2017). The proportion of these fish that leave the freshwater system and mature within the marine environment is unknown.

Juvenile outmigrants

South-Central California Coast steelhead fry emerge from the gravel from late spring through early summer. Like many other steelhead, juveniles from this DPS may rear in freshwater and estuarine habitats for 1–3 years. Juveniles migrate from freshwater habitat to estuaries in spring and summer. However, within the San Luis Obispo creek watershed, Spina et al. (2005) found that in observing three years of trap data that the peak downstream migration of steelhead smolts from this DPS occurred in April, with no smolts captured after May. As with other southern steelhead populations, due to low flows, outmigrating smolts from this DPS can get locked behind sandbars that delay their migration into marine waters. As a result, these fish could have their outmigration delayed by weeks or months (Moyle et al., 2017). When these fish leave their natal systems and migrate to marine habitats, marine entry occurs to the east of the California Study Area (Figure C-6).

Ocean rearing

As with other more southern populations of steelhead, ocean migration and rearing of South-Central California Coast steelhead is poorly understood. South-Central California Coast steelhead likely rear in the marine environment for two to four years (Moyle et al., 2017). Although fish from California have been detected as far north as the Gulf of Alaska (Burgner et al., 1992), it is likely that when adults have returned to their natal estuaries to find that they are blocked due to low stream flows, these fish remain relatively close to their natal systems in coastal California waters waiting for the barrier to be breached.

Return migration

In low water years where there are physical barriers to entering the estuaries (e.g., sandbars), returning South Central California Coast steelhead adults may delay their entry by weeks to a year (Moyle et al., 2017). Moyle et al. (2017) indicated that adults from this DPS can enter estuaries between January and May, before continuing their migration upstream to spawn. In a review, Myers (2018) indicated that adults begin entering the San Lorenzo River as early as November, where adults typically don't begin entering the Carmel River until January.

C.5.2.3.4.3 Population Trends

Following the completion of their most recent status review, NMFS (2023b) determined that, based on the best available information, including the new viability assessment, the current South-Central California Coast DPS recovery priority number remains the same as it was prior to this review (3C). The systemic anthropogenic threats identified at the time of the initial listing have remained essentially unchanged, though there has been significant progress in removing fish passage barriers in a number of the smaller and mid-sized watersheds. Threats to the South-Central California Coast DPS posed by environmental variability resulting from projected climate change are likely to exacerbate the factors affecting the continued existence of the DPS.

C.5.2.3.4.4 Population Threats

Population threats are similar across many steelhead populations. Please see the description of these threats described for the Northern California Steelhead DPS.

C.5.2.3.5 Southern California DPS

C.5.2.3.5.1 Status and Management

The Southern California Steelhead DPS was listed as threatened on August 18, 1997 (62 FR 43937) and their status was reaffirmed on June 28, 2005 (70 FR 37160) and updated on April 14, 2014 (79 FR 20802). This DPS includes naturally spawned steelhead originating below natural and manmade impassable barriers from the Santa Maria River to the U.S.-Mexico Border (79 FR 20802). No hatchery programs are currently included as part of this DPS.

Critical Habitat

NMFS designated critical habitat for Southern California DPS steelhead on September 2, 2005 (70 FR 52488). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Designated critical habitat for the Southern California DPS steelhead only includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas (70 FR 52488), not marine habitats, so it does not overlap spatially with the HCTT Study Area (Figure C-6).

Recovery Goals

See the 2012 recovery plan for the Southern California Steelhead DPS (National Marine Fisheries Service, 2012f) for complete down-listing/delisting criteria for recovery goals for the species. The Recovery Plan outlines the following coordinated actions that will be required for an effective steelhead recovery program:

- Prevent steelhead extinction by protecting existing populations and their habitats,
- Maintain current distribution of steelhead and restore distribution to some previously occupied areas,
- Increase abundance of steelhead to viable population levels, including the expression of all life history forms and strategies,
- Conserve existing genetic diversity and provide opportunities for interchange of genetic material between and within viable populations, and
- Maintain and restore suitable habitat conditions and characteristics to support all life-history stages of viable populations.
- Refine and demonstrate attainment of recovery criteria through research and monitoring.

C.5.2.3.5.2 Habitat and Geographic Range

Geographic range of spawning systems for this DPS: Naturally spawned anadromous steelhead originating below natural and manmade impassable barriers from the Santa Maria River to the Tijuana River at U.S.-Mexico Border (79 FR 20802).

Juvenile outmigrants

The highly variable environmental conditions where Southern California steelhead occur requires fish from this DPS to be highly adaptable to poor conditions and display multiple life history patterns. Like the two steelhead DPS's in closest proximity to the north, Southern California steelhead display anadromous, freshwater resident, lagoon-anadromous life history strategies. Habitat occupancy and migration within streams at the southern extent of this species' range are dependent on winter rains bringing increased flows and lower temperatures. Like populations to the north, it's likely that lagoon-anadromous steelhead represent a larger proportion of the spawning fish than do the anadromous steelhead in this DPS (Moyle et al., 2017).

Due to warmer conditions in southern habitats, Southern California steelhead eggs hatch more quickly than for steelhead further north. Juvenile steelhead can remain in freshwater systems for one to three years before migrating to the ocean. Growth rates are accelerated for juvenile steelhead within some southern systems. For example, Dagit et al. (Dagit et al., 2019; Huber et al., 2024) found that some freshwater systems produce large steelhead smolts by age-1 and age-2. Due to a combination of high growth rate and relatively poor freshwater conditions, those fish within this DPS that display anadromy may enter the marine environment at a younger age than other steelhead populations (Moyle et al., 2017). When these fish leave their natal systems and migrate to marine habitats, marine entry occurs to the east of PMSR, near Camp Roberts (Figure C-6).

Ocean rearing

Though ocean migration and rearing of Southern California steelhead is poorly understood, juveniles from this DPS are believed to outmigrate as age-1 or age-2 smolts. Age at outmigration is dependent on sufficient stream flows to navigate past physical barriers to fish migration, or to human assistance past these barriers (Stillwater Sciences, 2024). Fish from California have been detected as far north as the Gulf of Alaska (Burgner et al., 1992). Age at maturation for this DPS may be broader than for other DPS's. Although Southern California steelhead are thought to mature as age-2 to age-4 fish (Crozier et al., 2019), some of the returning fish may be age-5 and age-6 fish. This may be due, in part, to annual barriers to upstream migration, with ocean-rearing fish from this DPS spending an extended period in the marine environment waiting for natal stream barriers to be breached.

Return migration

Very little is known about mature adult steelhead from the Southern California Steelhead DPS. Only 177 adult steelhead from this DPS were documented over a 25-year period (Dagit et al., 2020). Adults appear to have a patchy coastal distribution that is tied to hydrologic conditions of their natal streams (Dagit et al., 2020). There is a high degree of interannual variability in observed presence, with adult detections ranging from as few as one fish to as many as 49 (annual average of five fish per year) (Dagit et al., 2020). Provided flows are sufficient to reduce or eliminate migration barriers, adult Southern California steelhead begin leaving the marine environment and entering their natal streams from September to November (Myers, 2018) to begin spawning from January through May, with peak spawning occurring from February through April (Crozier et al., 2019; Moyle et al., 2017).

C.5.2.3.5.3 Population Trends

Following the completion of their most recent status review, NMFS (National Marine Fisheries Service, 2023c) determined that, based on the best available information, including the new viability assessment, the current Southern California DPS recovery priority number remains the same as it was prior to this review (1C). The extended drought and genetic data documenting the high level of introgression and extirpation of native steelhead stocks in the southern portion of the DPS has elevated the threats level to the already endangered populations; the drought, and the lack of comprehensive monitoring, has also limited the ability to fully assess the status of individual populations and the DPS as whole. The systemic anthropogenic threats identified at the time of the initial listing have remained essentially unchanged over the past five years, though there has been significant progress in removing fish passage barriers in several the smaller and mid-sized watersheds. Threats to the Southern California Steelhead DPS posed by environmental variability resulting from projected climate change are likely to exacerbate the factors affecting the continued existence of the DPS.

Steelhead stocks have declined substantially from their historic numbers and many now are threatened with extinction. Native lineages have been nearly extirpated from the southern region of the native range, with only a few relict populations persisting in the headwaters of the San Gabriel, Santa Ana, and San Luis Rey rivers (National Marine Fisheries Service, 2016s). Abadia-Cardoso et al. (2016) and Jacobson et al (2014) documented that the majority of steelhead sampled between southern California watersheds and Mexico were genetically related to hatchery rainbow trout. This may indicate either replacement of native steelhead or hybridization with native steelhead in southern California.

Most of the steelhead DPS, including the Southern California Coast DPS, have low abundances relative to historical levels, and there is widespread occurrence of hatchery fish in naturally spawning populations (Goetz et al., 2024; Good et al., 2005; Huber et al., 2024; National Marine Fisheries Service, 2010g, 2012c). NMFS has reported population sizes from individual DPSs, but because of these units occur together while at sea, it is difficult to estimate the marine population numbers.

C.5.2.3.5.4 Population Threats

Population threats are similar across many steelhead populations. Please see the description of these threats described for the Northern California Steelhead DPS.

C.5.2.4 Green Sturgeon (*Acipenser medirostris*)

C.5.2.4.1 Status and Management

NMFS has identified two DPS of green sturgeon; northern and southern (Israel et al. 2009). Effective June 6, 2006, NMFS determined that the Southern DPS of green sturgeon warranted listing as a threatened species under the ESA (71 FR 17757). Green sturgeon have been observed in large concentrations in the summer and autumn within coastal bays and estuaries along the west coast of the US, including the Columbia River estuary, Willapa Bay, Grays Harbor, San Francisco Bay and Monterey Bay (Huff et al. 2012; Lindley et al. 2011; Lindley et al. 2008; Moser and Lindley 2007).

Critical Habitat

NMFS designated critical habitat for this DPS on October 9, 2009 (74 FR 52300). Critical habitat includes coastal U.S. marine waters within 60 fathoms (360 ft.) depth from Monterey Bay, California north to Cape Flattery, Washington, to its U.S. boundary. Approximately 25 mi. due west of San Francisco Bay, a small portion of the northern study area boundary overlaps with designated green sturgeon critical habitat (Figure C-7). Critical habitat also includes several rivers and estuaries along the U.S. West Coast.

For coastal marine areas, the physical or biological features of critical habitat designated for green sturgeon include food resources, migratory corridors, and water quality. Corresponding species life history events include subadult growth and development, movement between estuarine and marine areas, and migration between marine areas, as well as adult sexual maturation, growth and development, movements between estuarine and marine areas, migration between marine areas, and spawning migration (74 FR 52300).

Recovery Goals

See the 2018 Recovery Plan for the de-listing criteria crucial for the recovery of Southern DPS of green sturgeon (National Marine Fisheries Service, 2018b). The criteria for recovery are listed below:

- Abundance. The adult Southern DPS of green sturgeon census population remains at or above 3,000 for 3 generations (this equates to a yearly running average of at least 813 spawners for approximately 66 years). In addition, the effective population size must be at least 500 individuals in any given year and each annual spawning run must be comprised of a combined total, from all spawning locations, of at least 500 adult fish in any given year.
- Distribution. The Southern DPS of green sturgeon spawn successfully in at least two rivers within their historical range. Successful spawning will be determined by the annual presence of larvae for at least 20 years.
- Productivity. A net positive trend in juvenile and subadult abundance is observed over the course of at least 20 years.
- The population is characterized by a broad distribution of size classes representing multiple cohorts that are stable over the long term (20 years or more).
- Diversity. There is no net loss of Southern DPS green sturgeon diversity from current levels.
- Operation guidelines and/or fish screens are applied to water diversions in mainstem Sacramento, Feather, and Yuba rivers or San Francisco Bay Delta Estuary such that early life stage entrainment is below a level that limits juvenile recruitment.
- Threat-Based Recovery Criteria:
    1. Access to spawning habitat is improved through barrier removal or modification in the Sacramento, Feather, and/or Yuba rivers such that successful spawning occurs annually in at least two rivers. Successful spawning will be determined by the annual presence of larvae for at least 20 years.
    2. Volitional passage is provided for adult green sturgeon through the Yolo and Sutter bypasses.
    3. Water temperature and flows are provided in spawning habitat such that juvenile recruitment is documented annually. Recruitment is determined by the annual presence of age-0 juveniles in the lower Sacramento River or San Francisco Bay Delta Estuary. Flow and temperature guidelines have been derived from analysis of inter-annual spawning and recruitment success and are informing this criterion.
    4. Adult contaminant levels are below levels that are identified as limiting population maintenance and growth.
    5. Take of adults and subadults through poaching and state, federal, and tribal fisheries is minimal and does not limit population persistence and growth.

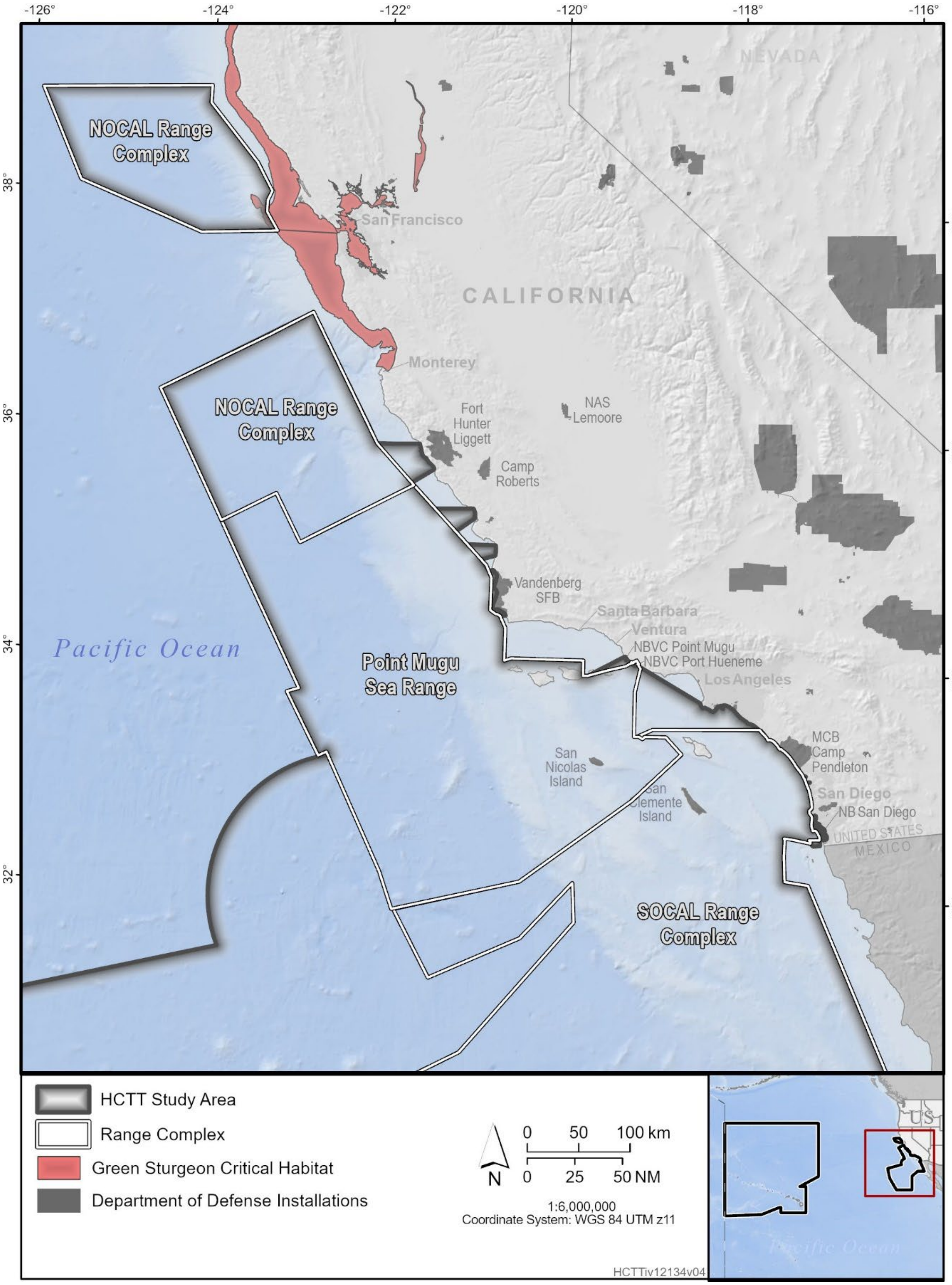

Figure C-7: Designated Critical Habitat for the Southern DPS of Green Sturgeon in the Action Area

C.5.2.4.2 Habitat and Geographic Range

Offshore portions of the northernmost portion of the California Study Area overlaps with the marine distribution of the southern DPS of green sturgeon. Subadult green sturgeon leave their Californian natal rivers and disperse widely along continental shelf waters of the west coast, typically north and shoreward of the 110 m contour (Erickson & Hightower, 2007; Moyle, 2002; National Marine Fisheries Service, 2005). Sub-adult and mature Southern DPS green sturgeon are known to exit the estuary of their natal watershed (San Francisco Bay-Delta) and enter the marine environment during late fall, migrate north along the Pacific coast within a relatively a narrow depth corridor within the 110 m contour of the continental shelf, typically occupying depths of 40–70 m (Erickson & Hightower, 2007; National Marine Fisheries Service, 2005; Payne et al., 2015). These fish then move into Oregon and Washington bays and estuaries again from mid-summer through early fall (Heironimus et al., 2022, 2023; Heironimus et al., 2024; Israel et al., 2009; Moser & Lindley, 2007). Huff et al. (2011) found that upon initially exiting the bays and estuaries again in mid-September, green sturgeon remained at relatively shallow depths of just more than 20 m. By mid-October these fish moved to deeper habitats ranging from 50–60 m in depth. Huff et al. (2011) found that no tags reported from waters greater than 115 m depth. Though not typical since green sturgeon are naturally a demersal species, green sturgeon occasionally make rapid vertical ascents to near the surface (Erickson & Hightower, 2007).

While Huff et al. (2011) found that green sturgeon along the Siletz Reef near Lincoln City, Oregon, appeared to prefer marine areas with high seafloor complexity and boulder presence, Payne et al. (2015) found that that green sturgeon further south near the Umpqua River (Reedsport, Oregon) were associated with flat, soft bottom habitats that lack high relief bottoms. Information regarding their preference for areas of high seafloor complexity and prey selection in coastal waters (benthic prey) indicate green sturgeon reside and migrate along the seafloor while in coastal waters. Neither of these studies provided information on the prevalence of the southern DPS to the northern DPS.

Huff et al. (2012) estimated the distribution of green sturgeon by modeling species-environment relationships using oceanographic and migration behavior covariates with maximum entropy modeling of species geographic distributions. The models predicted that green sturgeon presence would vary somewhat throughout the species range across seasons, with a modeled predictive range extending from Baja California, Mexico, to the Bering Sea. However, the predictive concentration of green sturgeon was modeled to occur from approximately 41 to 51.5° N and in the vicinity of San Francisco and Monterey Bays from 36 to 37° N latitude. Trawl and fisheries observer data, as well as tagging studies prior to and following this publication, support the author's prediction that green sturgeon are primarily concentrated in the coastal waters of Washington, Oregon, and Vancouver Island (Huff et al., 2012).

In general, Huff et al. (2011) suggest that green sturgeon generally occur at low densities within the Pacific coastal marine environment, with northern habitats occupied more readily than southern habitats. Within northern regions where green sturgeon are more frequently observed, Heironimus et al. (2022, 2023; 2024) found that in June and July, subadult and adult green sturgeon become more prevalent in the coastal waters near Washington estuaries, apparently staging until estuary water quality is ideal.

The coastal distribution of green sturgeon in California waters is less understood than their occupancy in more northern waters. Miller et al. (Miller et al., 2020a) placed acoustic receivers within the San Francisco estuary and immediately north of the estuary at Point Reyes to detect acoustically tagged sturgeon, but their report did not indicate that any receivers were placed south of the estuary. The Point

Reyes receivers indicated small numbers of fish moving into and out of the estuary, but whether they moved north or south along the California coastline is unknown. As noted above, the distribution model developed by Huff et al. (Huff et al., 2012) indicated that the entire coastline of California could be suitable for green sturgeon, with the Monterey Bay area being suggested as an area with a higher probability of occurrence. However, with the supplement to Huff et al. (2012), the authors reviewed green sturgeon bycatch in limited entry trawl and California halibut fisheries in 2002–2010 from Washington, Oregon, and California. Although far more fishing effort occurred in Oregon and Washington than in California, of the 269 green sturgeon that occurred as bycatch in 55,711 commercial sets, none occurred south of Santa Cruz.

Over a two-year study period, Lindley et al. (Lindley et al., 2008) acoustically tagged more than 200 green sturgeon and monitored their detection using acoustic receivers from southeast Alaska to Monterey Bay. Unfortunately, the receivers in Monterey Bay were only deployed for 5 ½ months during the study (March–July 2005), but any potential differentiation between northern and southern DPS fish was not included. The Monterey Bay receivers detected very small numbers of green sturgeon from March–May 2005. Based on the limited number of studies investigating green sturgeon habitat occupancy in coastal California waters, it is estimated that green sturgeon likely occur in very low numbers south of the San Francisco estuary, relative to waters north of the estuary. There is insufficient data to estimate whether green sturgeon might occur more frequently in any given season along the California coastline.

C.5.2.4.3 Population Trends

The recovery criteria requires that the adult Southern DPS green sturgeon population remain at or above 3,000 for 3 generations. This equates to a yearly running average of at least 813 spawners for approximately 66 years (National Marine Fisheries Service, 2021b). However, in the most recent census, Mora et al. (2018) estimated the total population of Southern DPS green sturgeon at 17,548 individuals, with an estimated 2,106 adults, therefore notably below the recovery criteria.

C.5.2.4.4 Population Threats

Threats to the green sturgeon species that contribute to their risk of extinction include the loss of spawning habitat; concentration of spawning into a single spawning river; entrainment or impingement by water project operations, dredging, the Yolo Bypass migration barrier, power plant operations, or other in-water activities; bycatch of green sturgeon in other fisheries; and poor water quality conditions. The main factor in the decline of the Southern DPS of green sturgeon is the reduction of the spawning area to a limited section of the Sacramento River. Other threats to the DPS include insufficient freshwater flow rates in spawning areas; contaminants (e.g., pesticides); bycatch of green sturgeon in other fisheries; potential poaching (for caviar); entrainment by water projects; influence of non-native species; small population size; and elevated water temperatures (National Marine Fisheries Service, 2021b).

Larval and juvenile green sturgeon are likely eaten by other species present in freshwater spawning and rearing areas. Adult green sturgeon have few known predators, although some observations suggest predation by some shark species and marine mammals may occur (Emmett et al., 1991; Huff et al., 2011).

C.5.2.5 Eulachon (*Thaleichthys pacificus*)

C.5.2.5.1 Status and Management

The Southern DPS of Pacific Eulachon was listed as threatened under the ESA on March 18, 2010 (75 FR 13012. This listing includes all subpopulations of eulachon within the states of Washington, Oregon, and California and extended from the Skeena River in British Columbia south to the Mad River in Northern California.

Critical Habitat

Critical habitat for the Southern DPS was designated on October 20, 2011 (76 FR 65324). The southern DPS of Pacific eulachon has 16 specific designated areas as critical habitat within the states of California, Oregon, and Washington. The designated areas are a combination of freshwater creeks and rivers and their associated estuaries, comprising approximately 335 mi. (539 km) of habitat (76 FR 65324). Designated critical habitat for the Southern DPS of Pacific Eulachon does not overlap spatially with the HCTT Study Area.

Recovery Goals

See (National Marine Fisheries Service, 2017) for complete down listing/delisting criteria for the Southern DPS of eulachon. The goal of this recovery plan is to: 1) Increase the abundance and productivity of eulachon; 2) Protect and enhance the genetic, life history, and spatial diversity of eulachon throughout its geographical range; and 3) Reduce existing threats to warrant delisting of the species. To accomplish these goals, the Eulachon Recovery Team identified four recovery objectives:

- Ensure subpopulation viability.
- Conserve spatial structure and temporal distribution patterns.
- Conserve existing genetic and life history diversity and provide opportunities for interchange of genetic material between and within subpopulations.
- Eliminate or sufficiently reduce the severity of threats.

C.5.2.5.2 Habitat and Geographic Range

The Southern DPS of Pacific eulachon may be present in the most northern portions of the California Study Area. Eulachon typically spends three to five years in saltwater before returning to freshwater to spawn. They have been documented to occur from Point Conception, CA to Alaska (Gustafson et al., 2016b). Eulachon are primarily found within the "echo scattering layer" of the coastal continental shelf in near-benthic habitats of open marine waters (Gustafson et al., 2016b). Eulachon appear to live near the ocean bottom, or on the continental shelf at depths most commonly of 20–200 m, though they may occur as deep as 500 m (Wilson et al., 2006). Gustafson (2016b) found that the average depth of occurrence is between 137 and 147 m, with minimum depths of 59–79 m and maximum depths of 322–466 m.

C.5.2.5.3 Population Trends

Although eulachon spawn in the lower reaches of freshwater rivers and streams, they are primarily a marine fish, spending over 95 percent of their lives in ocean waters (California Department of Fish & Game, 2010; Gustafson et al., 2010; Gustafson et al., 2016a). The main spawning population in California has historically occurred in the Klamath River with smaller runs in the Mad River and Redwood Creek (California Department of Fish & Game, 2010). This spawning population represented the southernmost population of the species. In January 2006, a mature male eulachon was caught in a juvenile salmonid monitoring rotary screw trap operation at Knights Landing in the Upper Sacramento River, indicating that this species is not locally extirpated; however, abundances are extremely low (California Department of Fish & Game, 2010).

C.5.2.5.4 Population Threats

Limited new information has become available regarding the threats to eulachon since the 2016 5-year review (National Marine Fisheries Service, 2022a). High to moderate threats to eulachon include climate change impacts on ocean conditions, dams /water diversions, eulachon by-catch, climate change impacts on freshwater habitat, predation, water quality, and destruction, modification or curtailment of eulachon habitat or range (National Marine Fisheries Service, 2022a).

Predators of adult Pacific eulachon include fish-eating birds, sturgeon, salmonids, Pacific halibut, and marine mammals (Hart, 1973).

C.5.2.6 Oceanic Whitetip Shark (*Carcharhinus longimanus*)

C.5.2.6.1 Status and Management

The oceanic whitetip shark was listed as threatened on January 30, 2018 (83 FR 4153). A comprehensive status review of the oceanic whitetip shark based on the best scientific and commercial information available was completed in 2016 (Young et al., 2016a). Young and Carlson (2020) summarized the biology and conservation information of the oceanic whitetip shark, identified information gaps, and discussed future directions for recovery of this species. In 2023, NMFS published a draft recovery plan which provides detailed information on the oceanic whitetip shark's biology, ecology, status and threats, and conservation efforts, as well as a description of site-specific management actions necessary for the conservation and survival of the species and objective, measurable criteria that, when met, will allow the species to be removed from the endangered and threatened species list (National Marine Fisheries Service, 2023d).

Critical Habitat

In 2020, NMFS concluded that critical habitat is not determinable because sufficient information is not currently available to assess the impacts of designation or regarding physical and biological features essential to the conservation of this species (85 FR 12898).

Recovery Goals

(National Marine Fisheries Service, 2024e)See (National Marine Fisheries Service, 2017, 2024e) for complete down listing/delisting criteria for the oceanic whitetip shark. The goal of this recovery plan is to increase oceanic whitetip shark viability across its range, such that the species can achieve recovery and be removed from the List of Endangered and Threatened Wildlife under the ESA (i.e., delisted). To accomplish this goal, the Recovery Team identified three recovery objectives:

- Ensure the oceanic whitetip shark maintains resiliency and geographic representation, and is a functional component of the ecosystem, by increasing overall abundance to achieve viable populations in all ocean basins.
- Increase oceanic whitetip shark resiliency by managing or eliminating significant anthropogenic threats.
- Ensure the continued viability of the oceanic whitetip shark through development and effective implementation of regulatory mechanisms for the long-term protection of the species.

C.5.2.6.2 Habitat and Geographic Range

Oceanic whitetip sharks are found worldwide in warm tropical and subtropical waters between the 30° N and 35° S latitude near the surface of the water column (Young et al., 2016a). Oceanic whitetips occur throughout the Central Pacific, including the Hawaiian Islands south to Samoa Islands and in the eastern Pacific from Southern California to Peru, including the Gulf of California. This species has a clear preference for open ocean waters, with abundances decreasing with greater proximity to continental shelves. In terms of California fish fauna, Allen and Cross (2006) categorized oceanic white tip sharks as holoepipelagic and individuals would be found mostly far from shore. Preferring warm waters near or over 20°C (68°F), and offshore areas, the oceanic whitetip shark is known to undertake seasonal movements to higher latitudes in the summer (National Oceanic and Atmospheric Administration, 2016e) and may regularly survey extreme environments (deep depths, low temperatures) as a foraging strategy (Young et al., 2016a).

Oceanic whitetip sharks could occur in deep open ocean areas in the California Current Large Marine Ecosystem. They are known to occur in Baja California and may be found in surface waters off the continental shelf (Baum et al., 2015). Oceanic whitetip sharks would be expected offshore of Hawaii, but only in pelagic waters off the continental shelf in the far western part of southern California.

C.5.2.6.3 Population Trends

While the current population size is unknown, the best available information indicates the oceanic whitetip shark has experienced significant declines in abundance throughout its range over at least the last several decades due to overutilization in commercial fisheries resulting in excessive fishing mortality (National Marine Fisheries Service, 2023d).

C.5.2.6.4 Population Threats

Threats include pelagic longline and drift net fisheries bycatch, targeted fisheries (for the shark fin trade), and destruction or modification of its habitat and range (Baum et al., 2015; Defenders of Wildlife, 2015b). Legal and illegal fishing activities have caused significant population declines for the oceanic whitetip shark. It is caught as bycatch in tuna and swordfish longlines throughout its range. Habitat degradation has occurred due to pollutants in the environment that bioaccumulate and biomagnify to high levels in their bodies due to their high position in the food chain, long life, and large size (Defenders of Wildlife, 2015b).

As an apex species in the tropical open ocean waters, the oceanic whitetip shark has few natural threats.

C.5.2.7 Scalloped Hammerhead Shark (*Sphyrna lewini*)

C.5.2.7.1 Status and Management

On July 3, 2014, four of six identified DPS of scalloped hammerhead sharks were listed as endangered or threatened (79 FR 38214). The Eastern Pacific DPS of the scalloped hammerhead population, which includes the west coast of the U.S. and the southern part of the California Study Area, is listed as endangered under the ESA. The central Pacific DPS, which includes the Hawaiian archipelago and the Hawaii Study Area, was not warranted for listing. The scalloped hammerhead shark has undergone substantial declines throughout its range (Baum et al., 2003b). There is some evidence of population increases in some areas of the southeast U.S., such as the Gulf of Mexico (Ward-Paige et al., 2012), but because many catch records do not differentiate between the hammerhead species, or shark species in general, population estimates and commercial or recreational fishing landing data are unavailable in the

Study Area. Most of the abundance data is from the Gulf of California, where it is estimated that the scalloped hammerhead population has been decreasing by 6 percent per year (INP, 2006).

Critical Habitat

In 2015, NMFS concluded that that there are no marine areas within the jurisdiction of the United States that meet the definition of critical habitat for the Eastern Pacific DPS of scalloped hammerhead shark (80 FR 71774). In addition, NMFS found that there are no identifiable physical or biological features that are essential to the conservation of the scalloped hammerhead DPSs (80 FR 71774).

Recovery Goals

There is currently no Recovery Plan for scalloped hammerhead sharks.

C.5.2.7.2 Habitat and Geographic Range

The scalloped hammerhead shark is a coastal and semi-oceanic species distributed in temperate and tropical waters (Froese & Pauly, 2016a). Distribution in the eastern Pacific Ocean extends from the coast of southern California (U.S.), including the Gulf of California, to Ecuador and possibly Peru (Compagno, 1984) and off Hawaii in the central Pacific ocean. A genetic marker study suggests that females remain close to coastal habitats, while males disperse across larger open ocean areas (Daly-Engel et al., 2012).

Juveniles rear in coastal nursery areas in the southern California portion of the Study Area (Duncan & Holland, 2006), but rarely inhabit the open ocean (Kohler & Turner, 2001). Sub adults and adults occur over shelves and adjacent deep waters close to shore and entering bays and estuaries (Compagno, 1984). In the California Current Large Marine Ecosystem, records of the presence of scalloped hammerhead sharks in this area are very rare. Sighting and landings in the Study Area are documented to have occurred in San Diego Bay in 1981, 1996, and 1997 (Shane, 2001). Results of a long-term telemetry study (2009–2020) by Hutchinson et al. (2023) found that adult males and juveniles tagged in Kaneohe Bay (Oahu) exhibit fairly restricted movements throughout the Hawaiian Archipelago, and mature males specifically exhibit strong seasonal site fidelity to Kaneohe Bay.

C.5.2.7.3 Population Trends

The scalloped hammerhead shark has undergone substantial declines throughout its range (Baum et al., 2003a). There is some evidence of population increases in some areas of the southeast (Ward-Paige et al., 2012), but because many catch records do not differentiate between the hammerhead species, or shark species in general, population estimates and commercial or recreational fishing landing data are unavailable in the Study Area.

Specific information for the Eastern Pacific DPS of scalloped hammerhead shark is unavailable as only data for overall shark population estimates are available. In its 2014 status review, NMFS used two models to estimate the overall population of scalloped hammerhead sharks to range from approximately 142,000 to 169,000 individuals in 1981 and between 24,000 and 28,000 individuals in 2005 (Miller et al., 2014). Although there are no population estimates specifically for the Eastern Pacific DPS in Southern California, estimates of historical (~3,600 to 12,000 years ago) effective population sizes for the entire eastern Pacific region range from 34,995 to 43,551. However, Nance et al. (2011) reported that the current effective population size of the Eastern Pacific DPS is significantly smaller (1–3 orders of magnitude) than the historical effective population size. In general, however, the species has higher occurrence in the eastern tropical Pacific south of Southern California (Musick & Fowler, 2007). In 2019, NMFS announced its intent to conduct a 5-year review for the four DPSs of the scalloped hammerhead shark (84 FR 46938), however this review has not yet been completed.

Historically, three species of hammerhead sharks have been reported in California waters, although all are noted as uncommon species: *Sphyrna zygaena* (smooth hammerhead shark), *S. tiburo* (bonnethead shark), and *S. lewini* (scalloped hammerhead shark) (Shane, 2001). All three species have similar eastern Pacific distributions with smooth hammerhead shark being the more frequent of the uncommon species in California waters (Allen et al., 2006).

With a lack of population data for the Southern California area and likely limited occurrence due to very warm water preference, fisheries data is one method for assessing frequency of occurrence. Even though these types of data are often biased, fishery direct catch and bycatch data are often the most reliable source of information on the presence of transient, infrequent marine species such as the Eastern Pacific DPS of scalloped hammerhead shark.

Fusaro and Anderson (1980), Seigel (1985), and Shane (2001) report on known documented scalloped hammerhead shark catches in Southern California:

- First documented catch of a scalloped hammerhead in Southern California was for a single shark caught 1 mi. (2 km) off Santa Barbara in 1977 (Fusaro & Anderson, 1980)
- Three catches were recorded from Los Angeles County in 1984, with one shark reported as a juvenile (Seigel, 1985)
- 19 juvenile sharks (9 females/10 males) were caught by commercial gillnet and scientific research gillnets in south San Diego Bay from 1996 to 1997 (Shane, 2001)[1]

The San Diego Bay scalloped hammerhead shark catches were associated with the unusually strong 1997–1998 El Niño event (National Weather Service Climate Prediction Center 2014a). Shane (2001) reported monthly mean surface sea water temperature off La Jolla from 1926 to 1994 as ranging from 14–19°C with temperatures during the 1997–1998 El Niño being elevated 3–4°C higher than normal. Long term maximum coastal sea water temperatures off San Diego typically average around 21°C (70–71°F) (National Weather Service Climate Prediction Center 2014b).

C.5.2.7.4 Population Threats

The primary threat to the scalloped hammerhead shark is direct take, especially by the foreign commercial shark fin fishery (Miller et al., 2014; National Marine Fisheries Service, 2011d). Scalloped hammerheads are a principal component of the total shark bycatch in the swordfish and tuna longline fishery and are particularly susceptible to overfishing and bycatch in gillnet fisheries because of schooling habits (Food and Agriculture Organization of the United Nations, 2013). Longline mortality for this species is estimated between 91 and 94 percent (National Marine Fisheries Service, 2011d).

Scalloped hammerhead sharks have few predators and few natural threats. However, because the this species is found over broad geographic ranges, large-scale impacts such as global climate change that affect ocean temperatures, currents, and potentially food chain dynamics, most likely to pose the greatest natural threat to this species (Miller et al., 2014).

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[1] The highly estuarine south San Diego Bay area was also influenced by a thermal plume from seawater discharge by a regional power plant. The power plant and discharge pipe was decommissioned and demolished in February 2013.

C.5.2.8 Giant Manta Ray (*Manta birostris*)

C.5.2.8.1 Status and Management

The giant manta ray was listed as a threatened species under the ESA by NMFS on January 12, 2018 (83 FR 2916). NMFS also found that that critical habitat for the giant manta ray is not determinable due to the lack of sufficient data to perform the required analyses.

Critical Habitat

In 2019, NMFS determined that a designation of critical habitat was not prudent because there are no identifiable physical or biological features that are essential to the conservation of the giant manta ray within areas under U.S. jurisdiction (84 FR 66652).

Recovery Goals

There is currently no Recovery Plan for giant manta rays.

C.5.2.8.2 Habitat and Geographic Range

Giant manta rays are visitors to productive coastlines with regular upwelling, including oceanic island shores, and offshore pinnacles and seamounts. They utilize sandy bottom habitat and seagrass beds, as well as shallow reefs, and the ocean surface both inshore and offshore. The species ranges globally and is distributed in tropical, subtropical, and temperate waters. Typically they seasonally migrate more than 1,000 km (621.4 mi.), however not likely across ocean basins (National Oceanic and Atmospheric Administration, 2016d).

Although giant manta rays are found throughout the Hawaiian Islands, due to their typically more oceanic distribution (Stewart et al., 2016a), they are far less abundant in the coastal waters of the islands than reef mantas, *Manta alfredi* (Kashiwagi et al., 2011). In a genetic connectivity study, Whitney et al., (Whitney et al., 2023) found that manta rays in the Hawaiian archipelago have small, genetically-isolated resident island populations, suggesting that female manta rays are strongly philopatric (remain near a particular area) and do not migrate between island groups. Southern California is the northern edge of the giant manta ray's distribution in the California Current Large Marine Ecosystem (Defenders of Wildlife, 2015a).

C.5.2.8.3 Population Trends

No stock assessments exist for the giant manta ray. Most estimates of subpopulations are based on anecdotal observations by divers and fishermen, with current populations throughout its range estimated between 100 and 1,500 individuals (Miller & Klimovich, 2016). In general, giant manta ray populations have declined, except in areas where they are specifically protected, such as the Hawaiian Islands (National Oceanic and Atmospheric Administration, 2016d). Giant manta rays reach maturity at age 10 and have one pup every two to three years (National Oceanic and Atmospheric Administration, 2016d).

C.5.2.8.4 Population Threats

Threats to giant manta rays include fisheries and bycatch, and destruction or modification of habitat. The international market highly values the gill plates of the giant manta ray for use in traditional medicines. They also trade their cartilage and skins and consume the manta ray meat or use it for local bait. Bycatch occurs in purse seine, gillnet, and trawl fisheries as well (National Oceanic and Atmospheric Administration, 2016d). Other potential threats include degradation of coral reefs, interaction with marine debris, marine pollution, and boat strikes (Food and Agriculture Organization of the United Nations, 2013).

Threats to giant manta rays include disease and predation. Because of their large size, mantas have few natural predators and only large sharks such as tiger sharks and some hammerhead species are thought to be capable of preying on adult mantas (Manta Trust, 2017).

C.5.3 Species Not Listed under the Endangered Species Act

C.5.3.1 Jawless Fishes-Hagfishes (Order Myxiniformes) and Lampreys (Order Petromyzontiformes)

Hagfishes and lamprey are primitive, cartilaginous, vertebrates with very limited external features often associated with fishes, such as fins and scales (Helfman et al., 2009). Both groups inhabit marine water column and soft bottom seafloor habitats in depths greater than 30 m and below 13°C in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems.

Hagfish reproduction and early development has not been observed, and captive breeding has been unsuccessful (Powell et al., 2005). Females lay leathery eggs on the seafloor and when the eggs hatch, they are essentially miniature adults. Hagfishes prey on dying fishes or feed on dead fishes. Some hagfishes have commercial fishery importance as their external "skin" is used for making "eel leather" goods.

Lampreys are anadromous and larvae are buried in the soft bottoms of river backwaters (Moyle & Cech, 2004). Juvenile lamprey filter feed on algae and detritus. Adults are parasitic and use their oral disc mouth to attach to other fishes and feed on their blood (Moyle & Cech, 2004; Nelson et al., 2004). Hagfishes and lampreys have no known predators.

C.5.3.2 Ground Sharks (Orders Carcharhiniformes), Mackerel Sharks (Order Lamniformes), Carpet Sharks (Order Orectolobiformes), and Bullhead Sharks (Order Heterodontiformes)

Ground sharks and allies (bull, dusky, hammerheads, oceanic whitetip, and tiger) are cartilaginous fishes with two dorsal fins, an anal fin, five gill slits, and eyes with nictitating membranes. Reproduction includes internal fertilization with the young born fully developed. These sharks are highly migratory. They are found in the water column and bottom/seafloor habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems and open ocean areas. These sharks are associated with hard and soft bottoms, nearshore and open ocean surface waters, and deep-sea habitats.

Mackerel sharks and allies (great white, makos, and porbeagle) are cartilaginous fishes with a large first dorsal fin that is high, erect, and angular or somewhat rounded, anal fin with a keel, and a mouth extending behind the eyes. Reproduction includes internal fertilization with young being produced by means of eggs that are hatched within the body of the female. They are found in the water column and bottom/seafloor habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems and open ocean areas. These sharks are associated with nearshore and open ocean surface water habitats. Ground and Mackerel Sharks are efficient predators on large fishes, cephalopods, and marine mammals. Some species are targeted for commercial and recreational purposes.

Carpet sharks and allies are a diverse group inhabiting coral and rocky reefs in the order Orectolobiformes. This group includes whale sharks, which are the largest shark in the group and are one of three filter feeding sharks. Many of the carpet sharks, such as whale shark, are also highly migratory. Carpet sharks all share certain characteristics, including their mouth being completely in front of the eyes, both dorsal fins without spines, five pairs of gill slits, and an anal fin being present. Nurse sharks are also in this group and are usually yellowish-tan to dark brown, average around 8–9 ft. long, and can weigh over 200 pounds. They are nocturnal, scouting the sea bottom for prey such as crustaceans, molluscs, and stingrays. They spend most of the day resting on sandy bottom or in caves or reef crevices. Whale sharks are another member of the carpet sharks group and are the largest shark in the world, growing to a length of over 40 ft.

Bullhead sharks and allies (horn shark) are cartilaginous fishes with two dorsal fins, an anal fin, five gill slits, and eyes without nictitating membranes. Reproduction includes internal fertilization with egg cases laid in crevices. They are found in the bottom/seafloor habitat in the California Current Large Marine Ecosystems and are associated with soft bottoms habitat.

C.5.3.3 Frilled and Cow Sharks (Order Hexanchiformes), Dogfish Sharks (Order Squaliformes), and Angel Sharks (Order Squatiniformes)

Frill and cow sharks (sevengill, sixgill) are cartilaginous fishes, generally characterized by lacking traits such as an anal fin and nictitating membrane; they do possess six to seven gill slits, compared to five gill slits found in all other sharks. Reproduction includes internal fertilization with young being produced by means of eggs that are hatched within the body of the female. They are associated with deep-sea habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems (Froese & Pauly, 2016b; Moyle & Cech, 2004).

Dogfish sharks are cartilaginous fishes with two dorsal fins spines and a caudal fin that’s divided into two lobes: a larger dorsal lobe and a smaller ventral lobe. Reproduction includes internal fertilization with young emerging from eggs that are hatched within the body of the female. They are associated with soft bottom and deep-sea habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems (Froese & Pauly, 2016b; Moyle & Cech, 2004).

Angel sharks (e.g., Pacific angel shark) are cartilaginous fishes with flat, batoid-like body, two small spineless dorsal fins behind pelvic fins, and anal fin absent. Reproduction includes internal fertilization with young emerging from eggs that are hatched within the body of the female. They are associated with soft bottom habitat in the California Current Large Marine Ecosystem (Froese & Pauly, 2016b; Moyle & Cech, 2004).

C.5.3.4 Stingrays and Allies (Order Myliobatiformes), Sawfishes (Order Pristiformes), Skates and Guitarfishes (Order Rajiformes), and Electric Rays (Order Torpediniformes)

Stingrays and allies (eagle ray, manta) are cartilaginous fishes, distinguished by flattened bodies, enlarged pectoral fins that are fused to the head and gill slits that are placed on their ventral surfaces. Reproduction includes internal fertilization with the young born fully developed. They are associated with reefs, nearshore open ocean, inland waters, and deep-sea water column habitat in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems (Froese & Pauly, 2016b; Moyle & Cech, 2004).

Skates and guitarfishes are cartilaginous fishes, distinguished by flattened bodies, two reduced dorsal fins, and a reduced caudal fin. Reproduction includes internal fertilization and deposition of egg sacks. They are associated with soft bottom habitat in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems (Froese & Pauly, 2016b; Moyle & Cech, 2004).

Electric rays are cartilaginous fishes, distinguished by flattened bodies, two well-developed dorsal fins and caudal fin. Two large kidney shaped organs in a disc on either side of the electric ray’s head distinguish it from others, as these organs are able to produce strong electric shock at will (Madl & Yip, 2000). Reproduction includes internal fertilization with young being produced by means of eggs that are hatched within the body of the female. Only one species, the Pacific electric ray (*Torpedo californica*), has been recorded in the Study Area.

C.5.3.5 Ratfishes (Order Chimaeriformes)

Ratfishes (chimera, rabbitfish, and ratfish) are cartilaginous fishes, with smooth skin largely covered by placoid scales, and their color can range from black to brownish gray. Reproduction includes internal fertilization and deposition of egg capsules. Fishes in this group are associated with soft bottom and deep-sea habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems (Froese & Pauly, 2016b).

C.5.3.6 Herrings (Order Clupeiformes)

Herring and allies (anchovies, herrings, sardines, and shad) are bony fishes with a silvery body with the lateral line and fin spines absent, and usually scutes along ventral profile. They are found only in the marine environment in the water column and in seafloor habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems. Herring, menhaden, sardine, and anchovy species are well-known as valuable targets of commercial fisheries. Herring account for a large portion of the total worldwide fish catch (Food and Agriculture Organization of the United Nations, 2005, 2009). Herrings and allies are broadcast spawners. They are known to form schools to help conserve energy and minimize predation (Brehmer et al., 2007), which may facilitate some level of communication during predator avoidance (Marras et al., 2012). They feed on decaying organic matter and plankton while swimming in the water column (Moyle & Cech, 2004). Herring and allies support marine food webs as a forage fish and preyed upon by fish, birds, and marine mammals.

C.5.3.7 Tarpons (Orders Elopiformes and Albuliformes)

Tarpons and allies (bonefishes, halosaurs, Hawaiian ladyfish, and machete) are bony fishes with the body encased in silvery scales, a large mouth, a single dorsal fin (most), and a somewhat tapered tail with fin spines absent. They are associated with riverine, estuarine and marine environments on the surface, water column, and seafloor/bottom habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems. Tarpon and allies are important game species but are not considered edible. Tarpons and allies are broadcast spawners. Fertilized eggs float in the water column until hatching into a leptocephalous larva (ribbon-like, with no resemblance to the adult). During the change from larvae to juvenile, the body shrinks in length. Juveniles prey upon plankton and marine invertebrates, while adults feed on mid-water fishes. Tarpon and allies are nocturnal ambush predators (Wainwright & Richard, 1995) who prey on bottom-dwelling invertebrates and small fishes. Tarpons and allies are preyed upon by larger fishes, birds, and marine mammals.

C.5.3.8 Eels (Anguilliformes, Notacanthiformes, and Saccopharyngiformes)

Eels (conger, cutthroat, duckbill, false moray, morays, sawtooth, short-tailed, spiny, gulpers, and pelican eels) are bony fishes with a very elongate body, usually scaleless with pelvic fins, and without fin spines. They are associated with riverine, estuarine and marine environments in the water column, and seafloor/bottom habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems. Eels and allies have little fishery importance. Some species are broadcast spawners, and fertilized eggs float in the water column until hatching into a leptocephalous larva. Juveniles prey upon plankton and marine invertebrates, while adults feed on small fishes. Depending on the species and its habitat, eels can be diurnal or nocturnal ambush predators and prey on bottom-dwelling invertebrates and small fishes. Eels are preyed upon mostly by larger fishes.

C.5.3.9 Argentines and Allies (Order Argentiniformes)

Argentines and allies (argentines, barreleyes, deep-sea smelts, slickheads, and tubeshoulders) are bony fishes with typically silvery, elongate bodies, adipose fin and extremely large mouths sometimes present, and pelvic fins and spines sometimes absent. They are found only in the marine environment in the water column, and seafloor habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems. Argentines and allies have little fishery importance. Argentines and allies vary in their reproduction strategy. Some deep-sea species are capable of bioluminescence and release scents that may help to attract mates. Argentines are broadcast spawners and fertilized eggs float in the water column until hatching. Argentines and allies likely have few predators, but may be preyed upon by larger fishes.

C.5.3.10 Bristlemouths (Order Stomiiformes) and Allies

Bristlemouths and allies (dragonfishes, fangjaws, hatchfishes, and lightfishes) are bony fishes with photophores and adipose fin present and chin barbels sometimes present. Bristlemouths and hatchetfishes are small in size and the most abundant fishes in many parts of the world's oceans. They are capable of eating large and small prey items and are known to engage in prey-related vertical migration patterns. Other species in this order are largely piscivorous (Moyle & Cech, 2004).

C.5.3.11 Greeneyes and Allies (Order Aulopiformes)

Greeneyes and allies (barracudinas, daggertooth, lizardfishes, pearleyes, and waryfishes) are bony fishes with an upper protrusible jaw, an adipose fin and forked tail usually present with fin spines absent. Most greeneyes and allies are small (less than 50 cm) predators capable of devouring a wide range of species, including other fishes nearly their same size and pelagic invertebrates. Fishes in this order are preyed upon by salmon, tunas, and swordfishes. Reproduction is usually external, and includes the ability to change sex (Froese & Pauly, 2016b).

C.5.3.12 Lanternfishes and Allies (Order Myctophiformes)

Lanternfishes and allies (headlight, lampfishes, and lancetfishes) are bony fishes that are usually small-sized, with an adipose fin, forked tail and photophores usually present. Lanternfishes can occur closer to the surface at night (10–100 m) and deeper during the day (300 to 1,200 m) (Froese & Pauly, 2016b), where they may become prey for marine mammals. These fishes often are an important part of the deep scattering layer (Moyle & Cech, 2004). Lanternfishes prey upon copepods and krill (Tyler & Pearcy, 1975).

C.5.3.13 Hakes and Allies (Order Gadiformes)

Hakes and allies (cods, codlings, grenadiers, and whiptails) are bony fishes with long dorsal and anal fins, no true spines in fins, although spinous rays present in dorsal fin of most species, and chin barbels are often present. Hakes and allies account for approximately half of the global commercial landings (Food and Agriculture Organization of the United Nations, 2005). Prey items for fishes in this group include small crustaceans during juvenile phases and larger crustaceans, squid and fishes as adults. Predators include striped bass, sharks, and cetaceans (Froese & Pauly, 2016b).

C.5.3.14 Brotulas and Allies (Order Ophidiiformes)

Brotulas and allies (cusk-eels) are bony fishes with pelvic absent or far forward and filamentous, dorsal and anal fins joined to caudal fin, and spines absent. These fishes exhibit a variety of reproductive strategies including external fertilization and giving live birth. Prey items for fishes in this group include small crustaceans during juvenile phases and larger crustaceans, squid and fishes as adults. Predators include striped bass, sharks, and cetaceans (Froese & Pauly, 2016b).

C.5.3.15 Toadfishes and Allies (Order Batrachoidiformes)

Toadfishes and allies (midshipman) are bony fishes with compressed bodies, large, depressed head and mouth usually with tentacles, and two dorsal fins with the first with spines. These fishes are known to build nests (Moyle & Cech, 2004).

C.5.3.16 Anglerfishes and Allies (Order Lophiiformes)

Anglerfishes and allies (footballfishes, frogfishes, goosefishes, and sea devils) are bony fishes with globulose bodies, a spine on the first dorsal fin and the pelvic fins usually absent. Anglerfish attract potential prey using their first dorsal fin (illicium) as a lure (Yasugi & Hori, 2016). Fishes in these orders are found occasionally on the surface, but most frequently in the water column and seafloor habitats. Additional adaptations include large mouths, sharp teeth, and sensitive lateral line (sensory) systems (Haedrich, 1996; Koslow, 1996; Marshall, 1996; Rex & Etter, 1998; Warrant & Locket, 2004). These fishes are mostly generalist feeders. Reproduction is not well studied, but sexes are separate and some exhibit parasitism (Moyle & Cech, 2004). Fishes in this group generally have no fishery importance.

C.5.3.17 Flyingfishes (Order Beloniformes)

Flyingfishes (halfbeaks, needlefishes, and sauries) are bony fishes with jaws extended into a beak; pelvic fins very large wing-like; spines absent. These fishes are associated with reefs, submerged aquatic vegetation, and open ocean habitat in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems and open ocean areas (Froese & Pauly, 2016b).

C.5.3.18 Killifish (Order Cyprinodontiformes)

Killifishes such as the California killifish (*Fundulus parvipinnis*) is bony fish with a protrusible upper jaw, fin spines rarely present, and a single dorsal fin. Killifishes are found in the water column of rivers and estuaries in the California Current Large Marine Ecosystem.

C.5.3.19 Silversides (Order Atheriniformes)

Silversides (grunion, jacksmelt, and topsmelt) are bony fishes with a silvery stripe on their sides, high pectoral fins, a dorsal fin, and a pelvic fin with a spine. These fishes are found on the surface and in the water column in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems.

C.5.3.20 Opahs and Allies (Order Lampriformes)

Opahs and allies (crestfishes, oarfishes, ribbonfishes, tapertails, and tube-eyes) are bony fishes with an upper protrusible jaw, and pelvic fins located forward on body, below, or just behind insertion of pectoral fins. Toadfishes (midshipman) have compressed bodies; large, depressed head and mouth usually with tentacles; and two dorsal fins, the first with spines. Fishes in this group exhibit a variety of reproductive strategies including external fertilization and parasitism. Prey items for fishes in this group include crustaceans, squid, and fishes. These fishes are found in the water column and seafloor habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems and open ocean areas.

C.5.3.21 Squirrelfishes and allies (Order Beryciformes)

Squirrelfishes and allies (bigscales, fangtooths, pricklefishes, slimeheads, and whalefishes) are bony fishes with round bodies and one dorsal fin often set far back, with pelvic fins absent and fin spines often present. Squirrelfishes (family Holocentridae) are the largest and most widely distributed family in the order, with over 60 species found throughout tropical and subtropical marine habitats (Moyle & Cech, 2004). Most species in this group occupy shallow nearshore reef and rocky areas where they hide during the day and come out at night to feed on zooplankton in the water column.

C.5.3.22 Dories and Allies (Order Zeitformes)

Dories and allies (boarfishes, oreos, and tinselfishes) are bony fishes that have deeply compressed bodies, protrusible jaws, spines in dorsal fin, and pelvic fin spines sometimes present. There are three species reported in the Study Area (Froese & Pauly, 2016b). These fishes are only found in marine habitats, and most are deep-sea species. Fishes in this order typically have large heads with distensible jaws that allow them to capture larger-sized prey, including fishes and crustaceans.

C.5.3.23 Pipefishes and Allies (Orders Syngnathiformes)

Pipefishes and allies (cornetfish, seahorses, and snipefishes) are bony fishes that exhibit unique body shapes with a tube-like snout, small mouth, and scales that are often modified bony plates. These fishes are associated with hard and soft bottom, submerged aquatic vegetation, reefs, and deep-sea habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems (Froese & Pauly, 2016b; Paxton & Eshmeyer, 1998). Some pipefishes and allies exhibit a high level of parental care by brooding pouches (male seahorses), which results in relatively few young being produced (Helfman et al., 2009). Most fishes in this group are diurnal ambush predators and prey on zooplankton, marine invertebrates, and small fishes. Pipefishes and allies are preyed upon by larger fishes and birds.

C.5.3.24 Sticklebacks (Order Gasterosteiformes)

Sticklebacks are small fishes comprised of only seven species that live in freshwater, saltwater, or brackish water (Helfman et al., 2009; Moyle & Cech, 2004). Species in this group are easily recognized by the presence of three to 16 isolated spines on their back in front of the dorsal fin, large eyes, and small upturned mouths. Most species in this group possess a row of bony plates on each side. Some sticklebacks display parental care through nest building. Fishes in this group are found in littoral marine waters and freshwater habitats in the Study Area.

C.5.3.25 Scorpionfishes (Order Scorpaeniformes)

Scorpionfishes and allies (poachers, rockfishes, snailfishes, and sculpins) are bony fishes with usually strong spines on head and dorsal fin, cheeks with bony struts, and rounded pectoral fins. These fishes are associated with hard and soft bottom, reefs, and deep-sea habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems and open ocean areas (Froese & Pauly, 2016b; Paxton & Eshmeyer, 1998). Some scorpionfishes have commercial and recreation fishery importance (Moyle & Cech, 2004). Reproduction methods vary widely between species and include external fertilization and egg deposition (sculpins) and internal fertilization and bearing live young (rockfishes). Most fishes in this group are diurnal ambush predators and prey on bottom-dwelling invertebrates and small fishes. Scorpionfishes are allies are preyed upon by larger fishes, birds, and marine mammals.

C.5.3.26 Mullets (Order Mugiliformes)

Mullets (blue spot, flathead grey, kanga, striped) are bony fishes with a streamline body, forked tail, hard angled mouth, large scales, high pectoral fins, and pelvic fins with one spine. Striped mullet is an important commercial fishery (Froese & Pauly, 2016b). These fishes are associated with soft bottom, reefs, and nearshore open ocean habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems (Froese & Pauly, 2016b; Moyle & Cech, 2004). Mullet are catadromous; they spawn in saltwater but spend most of their lives in freshwater environments. Mullet farming is also an ancient

Hawaiian tradition that pre-dates European contact with the islands (Costa-Pierce, 2012). Fishponds in ancient Hawaii were developed in upland areas to cultivate taro and simultaneously grow a limited range of euryhaline and freshwater fish, such as mullet.

C.5.3.27 Order Perciformes

Perciforms are the largest order of vertebrates, with over 7,800 species. They are extremely diverse, but most species are adapted for life as predators in the shallow or surface waters of the ocean. Some of the characteristics include fin spines present, dorsal fins either double or made up of two distinct parts with the lead spiny, adipose fin absent, pelvic fins thoracic or jugular in position or absent, pectoral fins on side of body, ctenoid scales, and closed swim bladder. Nearly half of all species belong to four families: gobies, wrasses, seabasses, or blennies(Moyle & Cech, 2004). Fish groupings in this section generally follow the classification in Nelson (2016).

C.5.3.27.1 Perches and Allies

Perches and allies (angelfishes, cardinal fishes, damselfishes, drums, grunts, jacks, remoras, sea basses, snappers, striped bass, and surfperches) are bony fishes with deep to moderately elongate bodies, one to two dorsal fins, with large mouth and eyes and thoracic pelvic fins. The Deep 7 (six species of snapper and one grouper) are the most culturally important and highly valued of the deep-water bottomfish species in Hawaii. Perches and allies are associated with hard and soft bottom, reefs, submerged aquatic vegetation, open ocean, and deep-sea habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems and open ocean areas (Froese & Pauly, 2016b; Moyle & Cech, 2004).

C.5.3.27.2 Wrasses and Allies

Wrasses and allies (hogfishes, parrotfishes, wrasses, and damselfishes) are bony fishes with a compressed body, large scales, well-developed teeth, and usually colorful coloring. Some wrasses and allies have recreational fishery and aquarium trade importance. Most of these fishes are associated with depths less than 30 m hard and soft bottom and reef habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems (Froese & Pauly, 2016b; Moyle & Cech, 2004). Wrasses and allies can change sex, usually female-to-male, and exhibit broadcast spawning; the fertilized eggs float in the water column or attach to substrate until hatching into larvae. Most are diurnal opportunistic predators (Wainwright & Richard, 1995). Prey items include zooplankton, invertebrates, and small fishes. Predators of wrasses and allies include larger fishes and marine mammals.

C.5.3.27.3 Eelpouts and Allies

Eelpouts and allies (gunnels, ocean pout, pricklebacks, and wolfeels) are bony fishes with an eel-like body, long dorsal and anal fins, and pelvic fins usually absent. These fishes are associated with soft bottom and deep-sea habitats in the California Current Large Marine Ecosystem (Froese & Pauly, 2016b; Moyle & Cech, 2004). Eelpouts have been found to occur near deep-sea vents in the Atlantic Ocean's Mid-Atlantic Ridge (National Geographic, 2016) and in deep water areas off Oahu and the Northern Hawaiian Islands (Yeh, 2008).

C.5.3.27.4 Stargazers

Stargazers are bony fishes with an elongated body and eyes on top of their head and big oblique mouths and are associated with soft bottom and deep-sea habitats in the California Current Large Marine Ecosystem (Froese & Pauly, 2016b). This group of fishes ambush their prey from the sand.

C.5.3.27.5 Blennies, Gobies, and Allies

Blennies, gobies, and allies (blackeye goby, cheekspot goby, mussel blenny) are bony fishes with an eel-like to sculpin-like body, and pelvic fins reduced or fused. They are associated with hard and soft bottoms, reefs, and deep-sea habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems (Froese & Pauly, 2016b).

C.5.3.27.6 Surgeonfishes

Surgeonfish (blue tang, moorish idol) are bony fishes with bodies that are deeply compressed laterally, small mouth, small scales, and pelvic fins with spines. They are associated with reef habitats in the Insular Pacific-Hawaiian Large Marine Ecosystem (Froese & Pauly, 2016b). These fishes scrape algae from coral reefs with small, elongated mouths. These grazers provide an important function to the reef system by controlling the growth of algae on the reef (Goatley & Bellwood, 2009).

C.5.3.27.7 Tunas and Allies

The tuna and allies (barracudas, billfishes, swordfishes, and tunas) have a large mouth, keels usually present, pelvic fins often absent or reduced, and are fast swimmers. These fishes are associated with reefs, nearshore and offshore open ocean habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems (Froese & Pauly, 2016b; Moyle & Cech, 2004). Most species have commercial and recreational importance. Tuna and allies are voracious open ocean predators (Estrada et al., 2003). They exhibit broadcast spawning and fertilized eggs float in the water column until hatching into larvae. Many feed nocturnally (Goatley & Bellwood, 2009) and in low-light conditions of twilight (Rickel & Genin, 2005). Many species in this group make large-scale migrations that allow for feeding in highly productive areas, which vary by season (Pitcher, 1995). Prey items include zooplankton for larvae and juvenile stages, while fishes and squid are consumed by subadults and adults. Predators of tuna and allies include other tuna species, billfishes, toothed whales, and some open ocean shark species. The Pacific bluefin tuna is a candidate species for listing under ESA.

C.5.3.27.8 Butterfishes

Butterfishes (ariommas, driftfishes, and medusafishes) are bony fishes with a blunt and thick snout, teeth small, and a maxilla mostly covered by bone. They are associated with soft bottom and deep-sea habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems (Froese & Pauly, 2016b). Butterfishes form large schools over the continental shelf, except during winter months when it may descend to deeper waters. Juveniles are associated with jellies and floating vegetation. Adults feed mainly on jellies, squids, and crustaceans. Some species of butterfishes are also commercially harvested (Froese & Pauly, 2016b).

C.5.3.28 Flatfishes (Order Pleuronectiformes)

Flatfish (flounders, halibut, sand dabs, soles, and tonguefish) are bony fishes with a flattened body and eyes on one side of body. These fishes occur on soft bottom habitat in inland waters, as well as in deep-sea habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems, and are an important part of commercial fisheries in the Study Area. The California halibut (*Paralichthys californicus*) is a representative of this group and is a recreationally fished species. Flatfishes are broadcast spawners. They are ambush predators, and prey on other fishes and bottom-dwelling invertebrates. Some species in this group have been affected by overfishing (Drazen & Seibel, 2007; Froese & Pauly, 2010).

C.5.3.29 Pufferfishes (Order Tetraodontiformes)

Pufferfishes (boxfishes, filefishes, ocean sunfishes and triggerfishes) are bony fishes with thick or rough skin, sometimes with spines or scaly plates, pelvic fins absent or reduced, and a small mouth with strong teeth coalesced into a biting plate. They are associated with hard and soft bottom, reef, submerged aquatic vegetation, nearshore and offshore open-ocean, and deep-sea habitats in the California Current and Insular Pacific-Hawaiian Large Marine Ecosystems. Pufferfishes are broadcast spawners. Predators vary by species, but due to spiny and rough exterior of this group, it is likely few are successful. Prey vary by species, but includes jellies, crustaceans, detritus, molluscs, and other bottom dwelling marine invertebrates (Froese & Pauly, 2016b).

C.6 Marine Mammals

C.6.1 General Background

Marine mammals are a diverse group of approximately 132 species (Committee on Taxonomy, 2022). Four main types of marine mammals are recognized worldwide: cetaceans (whales, dolphins, and porpoises), pinnipeds (seals, sea lions, and walruses), sirenians (manatees, dugongs, and sea cows), and other marine carnivores (sea otters and polar bears) (Jefferson et al., 2008; Rice, 1998). However, only 45 species are known to occur in the Study Area, and these include only species of cetaceans, seals and sea lions, and the southern sea otter.

Cetaceans can be divided into two smaller groups, odontocetes (i.e., toothed whales, dolphins, and porpoises) and mysticetes (i.e., baleen whales), based on their feeding strategy and anatomy. The different feeding strategies affect their distribution and occurrence patterns (Goldbogen et al., 2015). Odontocetes use teeth to capture and consume prey. Baleen whales use a fibrous structure called a baleen formed along the upper jaw and made of keratin, a type of protein similar to that found in human fingernails, which enables the whales to filter or extract prey from the water. Baleen whales are batch feeders that use baleen to engulf, suck, or skim large numbers of small prey, such as small schooling fishes, shrimp, or tiny animals called zooplankton from the water or out of ocean floor sediments (Heithaus & Dill, 2008).

Most pinnipeds can be divided into two families: phocids (true seals) and the otariids (fur seals and sea lions). Pinnipeds forage at sea on variety of prey, but unlike cetaceans, pinnipeds also spend time on land, or haul-out, for a variety of reasons including breeding, pupping, resting, and predator avoidance (Jefferson et al., 2015; Rice, 1998).

All marine mammals in the U.S. waters are protected under the Marine Mammal Protection Act (MMPA), while select species also receive additional protection under the ESA. Within the framework of the MMPA, a marine mammal “stock” is defined as “a group of marine mammals of the same species or smaller taxon (subspecies) in a common spatial arrangement that interbreed when mature” (16 U.S.C. section 1362). Per NMFS guidance, “for purposes of management under the MMPA, a stock is recognized as being a management unit that identifies a demographically independent biological population” (National Marine Fisheries Service, 2016o). However, in practice, recognized management stocks may fall short of this ideal because of a lack of information or, in some cases, stocks may even include multiple species in a management unit.

The ESA provides for listing species, subspecies, or DPSs of species, all of which are referred to as “species” under the ESA. The Interagency Policy Regarding the Recognition of Distinct Vertebrate Population Segments under the ESA (61 *FR* 4722, February 7, 1996) defines a DPS as, “any subspecies of fish or wildlife or plants, and any DPS of any species of vertebrate fish or wildlife which interbreeds when mature." If a population meets the criteria to be identified as a DPS, it is eligible for listing under the ESA as a separate species (National Marine Fisheries Service, 2016o). However, MMPA stocks do not necessarily coincide with DPS under the ESA (81 *FR* 62660–62320, September 8, 2016).

For a comprehensive summary of the general biology and ecology of marine mammals beyond the scope of this section, see Rice (1998), Twiss and Reeves (1999), Hoelzel (2002), Berta et al. (2006), Jefferson et al. (2015), and Perrin et al. (2009b). Additional species profiles and information on the biology, life history, distribution, and conservation of marine mammals can also be found through the following organizations:

- NMFS Office of Protected Resources (includes species distribution maps)
- Ocean Biogeographic Information System-Spatial Ecological Analysis of Megavertebrate Populations (known as OBIS-SEAMAP) species profiles
- National Oceanic Atmospheric Administration (NOAA) Cetacean Density and Distribution Mapping Working Group
- International Whaling Commission
- International Union for Conservation of Nature, Cetacean Specialist Group
- Marine Mammal Commission
- Society for Marine Mammalogy

C.6.1.1 Group Size

Many species of marine mammals, particularly odontocetes, are highly social animals that spend much of their lives in groups called "pods." The sizes and structures of these pods are dynamic and, based on the species, can range from several to several thousand individuals. Similarly, aggregations of mysticete whales may form during particular breeding or foraging seasons, although they do not persist through time as a social unit. Marine mammals that live or travel in groups are more likely to be detected by observers, and group size characteristics are incorporated into the acoustic effects modeling to represent a more realistic patchy distribution of species density. The behavior exhibited by many species of aggregating into pods is also important for the purposes of mitigation and monitoring, since animals that occur in larger groups have an increased probability of being detected. Information on species group size was reviewed and updated as part of updating density estimates for the Study Area.

C.6.1.2 Habitat Use

Marine mammals occur in every marine environment in the Study Area, from coastal and inshore waters to the open ocean. Their distributions are influenced by multiple factors, but primarily migration driven by breeding behaviors and prey distribution. Major ocean currents, bottom relief, water temperature, water depth, and salinity, can all affect prey distribution and are therefore often factors in understanding marine mammal distributions. The continuous movement of water from the ocean bottom to the surface in persistent upwelling zones around the world creates nutrient-rich, highly productive environments for marine mammal prey and are areas where marine mammal densities are typically higher (Jefferson et al., 2015; U.S. Department of the Navy, 2024). While these factors generally affect the distribution of most marine mammal species, their degree of influence varies by species.

For most cetaceans the distribution, abundance, and quality of prey largely determine where cetaceans occur at any specific time (Heithaus & Dill, 2008). Most of the baleen whales are migratory, but many of the toothed whales do not migrate in the strictest sense. Instead, they undergo seasonal dispersal or shifts in density distribution and occupy habitats preferable for feeding, breeding, and other important behaviors. Pinnipeds occur mostly in coastal habitats close to haulouts and forage over the continental shelf; however, some species disperse widely into the open ocean during the non-breeding, foraging season and may not haul out until returning to island or coastal breeding colonies.

In 2011, the Cetacean Density and Distribution Mapping Working Group identified and categorized areas of importance for cetaceans for reproduction, feeding, migration behaviors and areas in which small or resident populations are concentrated. Areas identified through this process were termed Biologically Important Areas (BIAs) (Ferguson et al., 2015; Van Parijs, 2015). The BIAs located in the Main Hawaiian Islands have since been reviewed and revised based on new data and information collected since the original BIAs were defined on how species use these areas (Kratofil et al., 2023). Similar revisions and additions to the original BIAs located off California were also identified (Calambokidis et al., 2024).

The BIAs were not meant to define exclusionary zones or serve as sanctuaries or marine protected areas and have no direct or immediate regulatory consequences. The identification of BIAs is intended to be a "living" process based on the best available science at the time (Ferguson et al., 2015; Harrison et al., 2023). As new empirical data are gathered, BIAs can be redefined to more closely correspond to the data describing a species' habitat use. Changes in habitat use may indicate that a previously defined BIA is no longer important to the essential life function for which it was defined, or may show that a broader expanse of habitat is used by the species for an essential behavior and that the BIA should be larger than previously defined. Also, a BIA may shift as a species' migratory routes change in response to environmental changes (e.g., warming sea surface temperatures).

The initial BIA delineations were published in March 2015, and included BIAs in Hawaii and off the U.S. West Coast (Aquatic Mammals, 2015a, 2015b; Baird et al., 2015b; Calambokidis et al., 2015c; Ferguson et al., 2015). In Hawaii, 21 BIAs for 12 cetacean species were identified. These included 20 small resident population areas and 1 non-contiguous humpback whale reproductive area (Baird et al., 2015b). In 2023, Kratofil et al. (2023) presented revisions to BIAs in Hawaii based on the implementation of a standardized scoring system. Each new and existing BIA was scored based on four criteria: Intensity of Use, Data Support, Importance, and Spatiotemporal Variability (Harrison et al., 2023). Using the scoring system and a hierarchical approach to identify core use areas within a broader area, 35 BIAs were identified in Hawaii for 12 cetacean species; these included 33 small resident BIAs for 11 odontocetes and 2 reproductive BIAs for humpback whales in the Main Hawaiian Islands (Kratofil et al., 2023). Hierarchical BIAs were defined for 9 of the 12 species yielding between one and three child BIAs for each of the 9 parent BIAs, depending on the species. Ten non-hierarchical BIAs were defined for six species. Table 4 in Kratofil et al. (2023) summarizes the characteristics and scores for each BIA. BIAs in the Hawaii Study Area are depicted in the descriptions of habitat and geographic range below for those species with BIAs defined in the Hawaiian Islands.

Twenty eight BIAs were identified for four species off the U.S. West Coast in 2015 (Calambokidis et al., 2015c), with five of those areas located within or overlapping the California portion of the 2018 HSTT Study Area. The BIAs included four feeding areas for blue whales and a migration area for gray whales (Calambokidis et al., 2015c). NMFS recently updated the BIAs for cetaceans on the U.S West Coast using the standardized scoring system described above for the BIAs in Hawaii (Calambokidis et al., 2024). BIAs were identified for two additional species, fin whales and Southern Resident killer whales, that were not delineated in the original effort. With the exception of the BIAs for harbor porpoise, which did not change, the BIAs for all other species included larger overall areas (parent BIAs), and smaller core use areas (child BIAs). BIAs in the California Study Area are depicted in the descriptions of habitat and geographic range below for those species with BIAs defined in the West Coast region.

It is important to note that BIAs are not a complete list of areas where species engage in important behavioral activities, do not represent a species' range or comprehensive habitat, and likely represent only a fraction of a species' overall distribution and habitat use (Ferguson et al., 2015).

C.6.1.3 Dive Behavior

Most marine mammals spend a considerable portion of their lives underwater while traveling or foraging. Some species of marine mammals have developed specialized adaptations to allow them to make deep dives lasting over an hour, primarily to forage for deep-water prey such as squid. Other species spend the majority of their lives closer to the surface and make relatively shallow, short duration dives. The diving behavior of a particular species or individual has implications for an observer's ability to detect them for purposes of mitigation and monitoring. In addition, their relative distribution through the water column is an important consideration when conducting acoustic exposure and direct strike analyses. Information and data on diving behavior for each marine mammal species were compiled and summarized in a technical report (U.S. Department of the Navy, 2017b) that provides estimates of time at depth based on available research. The dive data compiled in this technical report were incorporated into the Navy acoustic effects modeling.

C.6.1.4 Hearing and Vocalization

Refer to Appendix D (Acoustic and Explosive Impacts Supporting Information) for a summary and details regarding the hearing and vocalization of marine mammals.

C.6.1.5 General Threats

The growth and productivity of marine mammal populations, stocks, and individuals can be negatively affected by various natural factors and human activities. Direct effects, such as those resulting from disease or activities such as hunting and whale watching, can combine with indirect effects, such as a reduction in prey availability resulting from a large-scale change in the environment (e.g., an El Niño event) or lowered reproductive success of individuals, to impact populations. Nelms et al. (2021) provide a general discussion of current and emerging threats faced by marine mammals and approaches to conservation. General threats to marine mammal populations are summarized for each species in the population threats sections below. A more detailed discussion of general threats to marine mammals was provided in the 2018 HSTT and 2022 PMSR EIS/OEISs. New research published since 2022 on threats to marine mammals is consistent with information in the 2018 HSTT and 2022 PMSR EIS/OEISs and does not change the qualitative assessment of general threats on marine mammals described in those two EIS/OEISs. Therefore, with two exceptions, no updates to the descriptions of general threats provided in the 2018 HSTT EIS/OEIS are made in this section. The two exceptions are for threats from vessel strike and climate change. Information on vessel strikes as a general threat to marine mammals is relevant to the Proposed Action given the potential for vessel strikes from Navy vessels in the Study Area, and updates on climate change are relevant because of the potential for large-scale impacts on most marine mammal species and their distributions and the comprehensive new research published since the 2018 HSTT EIS/OEIS (Frankel et al., 2022; Gulland et al., 2022; Nelms et al., 2021; Sanderson & Alexander, 2020; Santora et al., 2020).

C.6.1.5.1 Vessel Strike

Vessel strikes are a growing issue for most large marine mammals. Although mortality may be a more significant concern for ESA-listed species, especially those that occupy areas with high levels of vessel traffic, the likelihood of encounter is greater for non-listed species with higher abundances (Currie et al., 2017a; Van der Hoop et al., 2013; Van der Hoop et al., 2015).

The reasons that whales are struck by vessels are likely several fold, including vessels approaching at high speeds making avoidance difficult, acoustic shadowing of an approaching vessel (i.e., the vessel body interferes with sound from the propulsion unit propagating forward) reducing alert time, and an increase in the number of vessels over the years transiting the oceans (Winkler et al., 2020). An animal at the surface could be struck directly by a vessel, a surfacing animal could hit the bottom of a vessel, or an animal just below the surface could be cut by a vessel's propeller. The severity of injuries typically depends on the size and speed of the vessel (Knowlton & Kraus, 2001; Laist et al., 2001; Vanderlaan & Taggart, 2007). Species that spend more time at or near the surface are at greater risk.

An examination of all known ship strikes from all shipping sources (civilian and military) indicates vessel speed is a principal factor in whether a strike results in death (Jensen & Silber, 2003; Knowlton & Kraus, 2001; Laist et al., 2001; Vanderlaan & Taggart, 2007(Rockwood et al., 2021; Winkler et al., 2020). In assessing records in which vessel speed was known, Laist et al., (2001) found a direct relationship between the occurrence of a whale strike and the speed of the vessel involved in the collision. The authors concluded that most deaths occurred when a vessel was traveling more than 13 knots. Jensen and Silber (2003) detailed 292 records of known or probable ship strikes of all large whale species from 1975 to 2002. Of these, vessel speed at the time of collision was reported for 58 cases. Of these cases, 39 (or 67 percent) resulted in serious injury or death. Operating speeds of vessels that struck various species of large whales ranged from 2 to 51 knots. The majority (79 percent) of these strikes occurred at speeds of 13 knots or greater. The average speed that resulted in serious injury or death was 18.6 knots. Pace and Silber (2005) found that the probability of death or serious injury increased rapidly with increasing vessel speed. Specifically, the predicted probability of serious injury or death increased from 45 percent to 75 percent as vessel speed increased from 10 to 14 knots and exceeded 90 percent at 17 knots. Higher speeds during collisions result in greater force of impact, but higher speeds also appear to increase the chance of severe injuries or death by pulling whales toward the vessel. Computer simulation modeling showed that hydrodynamic forces pulling whales toward the vessel hull increase with increasing speed (Clyne & Kennedy, 1999; Knowlton et al., 1995; Silber et al., 2010).

C.6.1.5.2 Climate Change

The global climate is warming and is having impacts on some populations of marine mammals (Garcia-Aguilar et al., 2018; Jefferson & Schulman-Janiger, 2018; National Oceanic and Atmospheric Administration, 2015c, 2018a; Peterson et al., 2006; Salvadeo et al., 2010; Shirasago-Germán et al., 2015; Silber et al., 2017; Simmonds & Eliott, 2009; Tulloch et al., 2018). Climate change can affect marine mammal species directly by causing shifts in distribution to match physiological tolerance under changing environmental conditions (Doney et al., 2012; National Marine Fisheries Service, 2018c; Peterson et al., 2006; Silber et al., 2017), which may or may not result in net habitat loss (some can experience habitat gains). Climate change can also affect marine mammals indirectly via impacts on prey, changing prey distributions and locations, and changes in water temperature (Giorli & Au, 2017; National Marine Fisheries Service, 2020a; Peterson et al., 2006; Rockwood et al., 2020; Santora et al., 2020). Sanford et al. (2019) have noted that severe marine heatwaves in California in 2014–2016 triggered marine mammal mortality events, harmful algal blooms, and declines in subtidal kelp beds. According to the Office of National Marine Sanctuaries (2019), climate drivers are currently the most concerning aspect of a decline in water quality and ecosystem health for giant kelp, mussels, and deep-sea corals across the Southern California Bight.

Changes in prey can impact marine mammal foraging success, which in turn affects reproduction success and survival. Starting in January 2013, an elevated number of strandings of California sea lion pups were observed in five Southern California counties. Additional California counties experiencing elevated California sea lion strandings include Santa Barbara County, Ventura County, Los Angeles County, and Orange County. This unusual number of strandings, continuing into 2016, were declared an Unusual Mortality Event (UME) by NMFS (National Oceanic and Atmospheric Administration, 2017a, 2018a). Although this UME was still considered as "ongoing" through 2017, the number of strandings recorded in 2017 were at or below average (National Oceanic and Atmospheric Administration, 2017a). This is the sixth UME involving California sea lions that has occurred in California since 1991. For this 2013–2015 event, NMFS biologists indicated that warmer ocean temperatures have shifted the location of prey species that are no longer adjacent to the rookeries, which thereby impacted the female sea lions' ability to find food and supply milk to their pups (National Oceanic and Atmospheric Administration, 2017a). As a result, this confluence of natural events causes the pups to be undernourished, and many are subsequently found stranded dead or emaciated due to starvation. In 2015, an UME was declared for Guadalupe fur seals along the entire California coast because of an eight-fold increase over the average historical number of strandings (approximately 12 per year) (National Marine Fisheries Service, 2019a; National Oceanic and Atmospheric Administration, 2018a). This event continued into 2017, although the number of animals involved declined in 2017; in April 2017 an additional seven Guadalupe fur seals stranded associated with this UME, with these latest strandings still being investigated. The initial assumption was that the cause for the increase in strandings was a change in the prey base due to warming conditions, but to date there has been no subsequent cause or other information in that regard provided by NMFS (National Oceanic and Atmospheric Administration, 2015c, 2018a). In a similar occurrence for gray whales and since January 2019, an elevated number of gray whale strandings has occurred along the west coast of North America from Mexico through Alaska resulting in NMFS declaring a UME for this species (National Marine Fisheries Service, 2019b; National Oceanic and Atmospheric Administration, 2020). This is similar to a previous UME for gray whales that occurred in 1999–2000.

Likely also due to changing prey distributions, data tagging efforts in July 2016 focusing on blue and fin whales had to be shifted north to central California waters when the majority of blue, fin, and humpback whales encountered in southern California waters were found to be too thin or otherwise in poor body condition to allow for them to be tagged (Oregon State University, 2017). In central California waters, the researchers identified good numbers of blue, fin, and humpback whales in better condition and indicative of a good feeding area that was likely to be sustained that season (Oregon State University, 2017).

Harmful algal blooms may become more prevalent in warmer ocean temperatures with increased salinity levels such that blooms will begin earlier, last longer, and cover a larger geographical range (Edwards, 2013; Moore et al., 2008). Warming ocean waters have been linked to the spread of harmful algal blooms into the North Pacific where waters had previously been too cold for most of these algae to thrive. The spread of the algae and associated blooms has led to mortality in marine mammals in locations where algae-caused biotoxicity had not been previously known (Lefebvre et al., 2016).

Climate change may indirectly influence marine mammals through changes in human behavior, such as increased shipping and oil and gas extraction, which benefit from sea ice loss (Alter et al., 2010). Ultimately impacts from global climate change may result in an intensification of current and ongoing threats to marine mammals (Edwards, 2013). In addition, the ability of marine mammals to alter behaviors may serve as a buffer against measurable climate change–induced impacts and could delay or mask any adverse effects until critical thresholds are reached (Baker et al., 2016b).

Marine mammals are influenced by climate-related phenomena, including storms and other extreme weather patterns such as the 2015–2016 El Niño in the ocean off the U.S. West Coast (see for example, Santora et al. (2020). Generally, not much is known about how large storms and other weather patterns affect marine mammals, other than that mass strandings (when two or more marine mammals become beached or stuck in shallow water) sometimes coincide with hurricanes, typhoons, and other tropical storms (Bradshaw et al., 2006; Marsh, 1989; Rosel & Watts, 2008) or other oceanographic conditions. There have also been correlations in time and space between strandings and the occurrence of earthquakes. However, there has been no scientific investigation demonstrating evidence for or against a relationship between earthquakes and the occurrence of marine mammal strandings. Indirect impacts may include altered water chemistry in estuaries (low DO or increased nutrient loading), causing massive fish kills (Burkholder et al., 2004) and thereby changing prey distribution and availability for cetaceans (Stevens et al., 2006). Human responses to extreme weather events may indirectly affect behavior and reproductive rates of marine mammals. For example, Miller et al. (2010) reported an increase in reproductive rates in bottlenose dolphins in the Mississippi Sound after Hurricane Katrina, presumably resulting from an increase in fish abundance due to a reduction in fisheries landings, a decrease in recreational and commercial boat activities (National Marine Fisheries Service, 2007c), and an increase in the number of reproductively active females available during the breeding seasons following the storm. Smith et al. (2013) supplemented the findings from this study and documented a marked increase in foraging activity in newly identified foraging areas that were observed during the two-year study period after the storm.

Habitat deterioration and loss is a major factor for almost all coastal and inshore species of marine mammals, with effects ranging from depleting a habitat’s prey base and the complete loss of habitat (Ayres et al., 2012; Kemp, 1996; Pine et al., 2016; Rolland et al., 2012; Smith et al., 2009; Veirs et al., 2015; Williams et al., 2014a). Many researchers predict that if oceanic temperatures continue to rise with an associated effect on marine habitat and prey availability, then either changes in foraging or life history strategies, including poleward shifts in many marine mammal species distributions, should be anticipated (Alter et al., 2010; Fleming et al., 2016; Ramp et al., 2015; Salvadeo et al., 2015; Silber et al., 2017; Sydeman & Allen, 1999). Poloczanska et al. (2016) analyzed climate change impact data that integrated multiple climate influenced changes in ocean conditions (e.g., temperature, acidification, DO, and rainfall) to assess anticipated changes to a number of key ocean fauna across representative areas. In relation to the Study Area, Poloczanska et al. (2016) included the California Current Ecosystem in their assessment. Their results predict a northward expansion in the distribution of zooplankton, fish, and squid, all of which are prey for many marine mammal species. Examples from the 2018 NMFS California Current Ecosystem survey consistent with that hypothesis were notable northern shifts in the summer/fall distribution and abundance of short-beaked common dolphins and blue whales from their 1996 to 2014 multi-year average in response to changing ocean conditions (Becker et al., 2020).

Concerns over climate change modifying the U.S. West Coast upwelling patterns, increasing levels of hypoxia, and ocean acidification have generated targeted research and monitoring efforts at selected “Sentinel Sites” (Lott et al., 2011); the Channel Islands National Marine Sanctuary is one of these monitored sites. There remains scientific uncertainty about how or if such changes will affect marine mammals and their prey, but acidification of the ocean could potentially impact the mobility, growth, and reproduction of calcium carbonate-forming organisms such as crustaceans and plankton, which are the direct prey of some marine mammals, as well as an important part of the overall food chain in the ocean, and can alter the propagation of sound underwater (Lynch et al., 2018; Rossi et al., 2016).

C.6.2 Endangered Species Act-Listed Species

There are 11 marine mammal species and applicable stocks or DPSs listed under the ESA that occur within in the Study Area.

C.6.2.1 Blue Whale (*Balaenoptera musculus*)

C.6.2.1.1 Status and Management

The world's population of blue whales can be separated into five subspecies, based on geographic location and some morphological differences (National Marine Fisheries Service, 2020c). In the HCTT Study Area the subspecies *Balaenoptera musculus* is present. In the North Pacific, NMFS divided *B. musculus* into two stocks, the Eastern North Pacific and Central North Pacific stocks (Carretta et al., 2023b). Both stocks of blue whales are listed as endangered under the ESA.

Recovery Goals

In response to the current threats facing the species, in 1998 NMFS developed goals to recover blue whale populations. The November 2020 Revised Recovery Plan (National Marine Fisheries Service, 2020c) for the blue whale provides the complete down-listing/delisting criteria to accomplish the recovery goal. The goal of the Revised Recovery Plan is to promote recovery of blue whales to a level at which it becomes appropriate to downlist the species from endangered to threatened status, and ultimately to delist, or remove the species from the list of Endangered and Threatened Wildlife and Plants, under the provisions of the ESA. The plan outlines the recovery actions to accomplish this goal as follows:

- Coordinate federal and international measures to maintain international regulation of whaling for blue whales.
- Determine blue whale taxonomy, population structure, occurrence, distribution, and range.
- Estimate population size and monitor trends in abundance.
- Identify, characterize, protect, and monitor habitat important to blue whale populations.
- Investigate human-caused potential threats and, should they be determined to be limiting blue whale recovery, take steps to minimize their occurrence and severity.
- Maximize efforts to acquire scientific information from dead, stranded, and entangled or entrapped blue whales.

C.6.2.1.2 Habitat and Geographic Range

Blue whales inhabit all oceans and typically occur near the coast, over the continental shelf, though they are also found in oceanic waters having been sighted, acoustically recorded, and satellite tagged in the eastern tropical Pacific (Ferguson, 2005; Stafford et al., 2004).

Blue whales from the Central North Pacific stock are found in the Hawaii portion of the Study Area, but the sighting frequency is low and the peak abundance is seasonal, generally occurring in the winter (Bradford et al., 2013). During three systematic ship surveys of the Hawaiian Islands EEZ in 2002, 2010, and 2017 (all in summer and fall), there was only one blue whale sighting in 2010 (Bradford et al., 2021). Whales feeding along the Aleutian Islands and in the Gulf of Alaska likely migrate to Hawaii in winter (Stafford et al., 2001). In the winter of 2014–2015 (December to January), passive acoustic detections of blue whales were recorded intermittently over the 3-week period of the survey (Klinck et al., 2015).

The Eastern North Pacific Stock of blue whales includes animals found in the eastern north Pacific from the northern Gulf of Alaska to the eastern tropical Pacific (Carretta et al., 2023b). Based on habitat models derived from line-transect survey data collected between 1991 and 2018 off the U.S. west coast, relatively high densities of blue whales are predicted off southern California during the summer and fall (Barlow et al., 2009b; Becker et al., 2010; Becker et al., 2016; Forney et al., 2012). Data from year-round surveys conducted off southern California from 2004 to 2013 show that the majority of blue whales were sighted in summer (62 sightings) and fall (9 sightings), with only single sightings in winter and spring (Campbell et al., 2015). In the Southern California Bight in summer and fall, the highest densities of blue whales occurred along the 200 m isobath in waters with high surface chlorophyll concentrations (Redfern et al., 2013). Campbell et al. (2015) documented blue whale sightings along both the southern California shelf, and over deep ocean water (>2,000 m).

This species has also frequently been heard on passive acoustic recording devices in the southern portion of the California Study Area (Širović et al., 2015a). Based on approximately 3 million detections in the waters of the Southern California Bight between 2006 and 2012, Širović et al. (2015a) found that blue whale vocalizations were more common at coastal sites and near the northern Channel Islands and generally heard between June and January with a peak in September. There was large variation among blue whales tagged in the southern portion of the California Study Area with the distance to shore ranging from less than 1 km and up to 884.8 km and blue whale movement along the Pacific coastline extending south to just 7.4°N latitude (just north of the equator and north to 50°N latitude just off British Colombia, Canada (Mate et al., 2015a). Data from a number of years and sources (Calambokidis & Barlow, 2013; Calambokidis et al., 2009a; Douglas et al., 2014b; Irvine et al., 2014; Mate et al., 2016a) consistently indicate large interannual variability in blue whale presence in small specific areas. Annual density predictions based on a habitat model also showed localized interannual variabilty in blue whale distribution patterns between 1996 and 2018 (Becker et al., 2020).

Abrahms et al. (2019) documented higher blue whale occurrence north of the SOCAL Range Complex and with critical areas located along shipping routes within the Santa Barbara Channel that provide access to the Ports of Los Angeles and Long Beach. Szesciorka et al. (2020) investigated the timing of blue whale migrations in association with environmental conditions and prey concentrations off southern California over a 10-year period. Their findings showed that blue whales were arriving up to one month earlier off southern California at the end of the study than they had been 10 years prior. However, the whales did not depart any earlier, leading to longer residency times in the Southern California Bight. Based on acoustic call detections, blue whales arrived in May and depart in November, remaining at the feeding grounds an average of 8.4 months. Blue whales demonstrated a flexible response to prey availability on an interannual basis based strongly on sea surface temperatures which are also correlated with krill biomass.

Szesciorka et al. (2019) analyzed passive acoustic detections and satellite tagging data to show that blue whales appeared to have little or no behavioral responses to close vessel passages. This is similar to lack of blue whale response to vessel traffic reported by McKenna et al. (2015). Palacios et al. (2019) showed how blue whale foraging behavior was influenced by modeled oceanographic variables likely associated with concentrating krill prey. The northward movement of blue whale foraging during marine heatwaves was also noted. Calambokidis et al. (2019) documented differences in blue whale day-night behaviors with more blue whale transit movements at night and at shallower depths.

BIAs were redefined for blue whale feeding behavior off the U.S. West coast by Calambokidis et al. (2024) and incorporated tagging and additional line-transect survey data not previously considered (Calambokidis et al., 2015c). The parent BIAs encompass 173,000 km^2 equivalent to 21 percent of the

U.S. West Coast EEZ and include coastal, shelf beak, and offshore waters (Figure C-8). The child BIA is 30 percent of the parent BIAs but still larger than the previous blue whale feeding BIAs defined in 2015.

C.6.2.1.3 Population Trends

Widespread whaling over the last century is believed to have decreased the global blue whale population to approximately 1 percent of its pre-whaling population size (Branch, 2007; Monnahan, 2013; Monnahan et al., 2014; Rocha et al., 2014; Širović et al., 2004). Off the Pacific Coast of North America, there was a documented increase in the blue whale population size between 1979–80 and 1991 (Barlow, 1994a) and between 1991 and 1996 (Barlow, 1997). Based on subsequent line-transect surveys conducted off the Pacific Coast between 2001 and 2005, the abundance estimates of blue whales appeared to decline in those waters over the survey period (Barlow & Forney, 2007). However, this apparent decline was likely due to variability in the distribution patterns of blue whales off the coast of North America rather than a true population decline (Barlow, 2010; Calambokidis et al., 2009a). Calambokidis et al. (2009a) suggested that when feeding conditions off California are not optimal, blue whales may move to other regions to feed, including waters farther north. A comparison of survey data from the 1990s to 2008 indicates that there has been a northward shift in blue whale distribution within waters off California, Oregon, and Washington (Barlow, 2010; Širović et al., 2015b). Consistent with the earlier suggested variability in the distribution patterns, Carretta et al. (2013) report that blue whales from the U.S. west coast have been increasingly found feeding to the north and south of the U.S. west coast during summer and fall. A mark-recapture study reported by Calambokidis et al. (2009b) indicated, ”a significant upward trend in abundance of blue whales” at a rate of increase just under 3 percent per year for the U.S. west coast blue whale population (Calambokidis & Barlow, 2013).

Recent information suggests that the population in the HCTT Study Area may have recovered and has been at a stable level following the cessation of commercial whaling in 1971 despite the impacts of ship strikes, interactions with fishing gear, and increased levels of ambient sound in the Pacific Ocean (Campbell et al., 2015; Carretta et al., 2015; Monnahan, 2013; Monnahan et al., 2015; Monnahan et al., 2014; Širović et al., 2015b). Based on a relative comparison of past sightings, dating as far back as the 1950s, with survey results from 2012, Smultea and Jefferson (2014) ranked blue whales as the sixth-most frequently sighted species in the Southern California Bight, noting that, “[t]his represents a clear relative increase from historical records.” Sixth (out of 16 species) is the highest rank for blue whales, which were ranked 12th based on 1981 survey results. Despite the evidence of a potential increase in the population size of blue whales off the U.S. west coast, a formal trend analysis has not been conducted so the current trend is unknown (Carretta et al., 2023b).

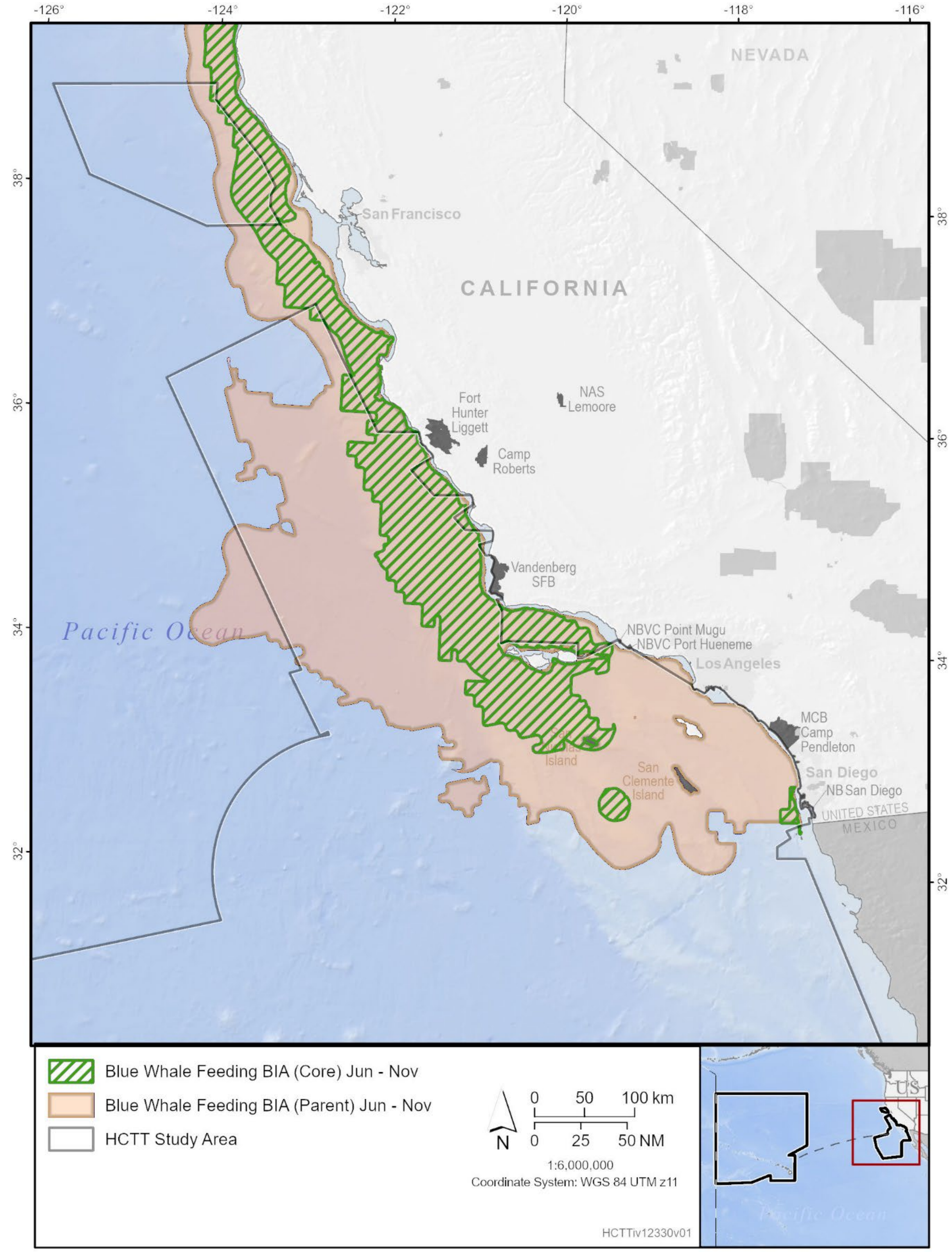

Figure C-8: Blue Whale Feeding BIAs in the California Study Area

C.6.2.1.4 Population Threats

Blue whales are susceptible to vessel strikes and entanglement in fishing gear. The most recent NMFS data from 2017 through 2021 reported 3 cases of blue whale injury or mortality attributed to vessel strikes off the U.S. West Coast (Carretta et al., 2023a). There has been approximately 14 recorded vessel strikes resulting in injury or mortality to blue whales in the region since 2007.

Furthermore, blue whales have been observed entangled in pot and trap fisheries, as well as unidentified fishing debris. Data from 2017 through 2021 indicate that that the mean annual entanglement rate of blue whales along the U.S. West Coast is 0.61 whales annually (Carretta et al., 2023a). Data specific to the California Dungeness crab pot fishery from the same 5-year period indicate that the average interaction that resulted in mortality or serious injury was 0.15 blue whales annually (Carretta et al., 2023a). Other anthropogenic threats to blue whales include pollution exposure, increased noise (e.g., vessel noise), and ingestion or other interactions with marine debris (National Marine Fisheries Service, 2016k).

Climate change has also increasingly become a threat to marine mammals, including baleen whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021). Its effects on oceanographic conditions that drive the distribution of planktonic prey are a potential threat to blue whale populations (Palacios et al., 2019; Szesciorka et al., 2020).

C.6.2.2 Fin Whale (*Balaenoptera physalus*)

C.6.2.2.1 Status and Management

The fin whale is listed as endangered under the ESA and depleted under the MMPA. In the North Pacific, NMFS recognizes three fin whale stocks: (1) a Northeast Pacific stock in Alaska; (2) a California, Oregon, and Washington stock; and (3) a Hawaii stock (Carretta et al., 2023b; Young, 2023). Although some fin whales migrate seasonally (Falcone et al., 2011; Mate et al., 2016b; Mate et al., 2015c), NMFS does not recognize fin whales from the Northeast Pacific stock as occurring in either Hawaii or off California.

Recovery Goals

In response to the threats facing the species, NMFS developed goals to recover fin whale populations. The 2010 Final Recovery Plan (National Marine Fisheries Service, 2010b) for the fin whale provides complete downlisting/delisting criteria for both of the following recovery objectives 1) Achieve sufficient and viable population in all ocean basins, and 2) Ensure significant threats are addressed. To accomplish these objectives, the plan outlines the recovery actions as follows:

- Coordinate state, federal, and international actions to implement recovery actions and maintain international regulation of whaling for fin whales.
- Determine population discreteness and population structure of fin whales.
- Develop and apply methods to estimate population size and monitor trends in abundance.
- Conduct risk analyses.
- Identify, characterize, protect, and monitor habitat important to fin whale populations in U.S. waters and elsewhere.
- Investigate causes and reduce the frequency and severity of human-caused injury and mortality.
- Determine and minimize any detrimental effects of anthropogenic noise in the oceans.

- Maximize efforts to acquire scientific information from dead, stranded, and entangled or entrapped fin whales.
- Develop post-delisting monitoring plan.

C.6.2.2.2 Habitat and Geographic Range

The fin whale is found in all the world's oceans and is the second largest species of whale (Jefferson et al., 2015). Fin whales prefer temperate and polar waters and are scarcely seen in warm, tropical waters (Reeves et al., 2002).

Fin whales are found in Hawaiian waters, but this species is not commonly detected in this portion of the Study Area (Carretta et al., 2023b). There was a total of nine fin whale sightings during systematic line-transect surveys of the Hawaiian Islands EEZ in 2002, 2010, and 2017 (Bradford et al., 2021). These data allowed for the derivation of an updated design-based abundance estimate for the Hawaiian Islands fin whale stock of 203 whales; however, the uncertainty associated with this estimate was quite high, resulting in a 95 percent confidence interval of 40 to 1,028 animals (Bradford et al., 2021). Based on sighting data and acoustic recordings, fin whales are likely to occur in Hawaiian waters mainly in fall and winter (Barlow et al., 2006; Barlow et al., 2008, Barlow, 2004 #2610; Klinck et al., 2015). In summer, fin whales are likely absent from Hawaiian waters, and during three separate line-transect surveys of waters within the Hawaiian Islands EEZ during summer and fall, fin whales were only seen during the fall months (Barlow, 2006; Bradford et al., 2017). Fin whales were not detected during the summer months of any year from 2011 to 2017 from passive acoustic recordings on an array of 14 hydrophones at the U.S. Navy Pacific Missile Range Facility (PMRF) off Kauai, Hawaii (Guazzo et al., 2021; Helble et al., 2020).

Fin whales calls have frequently been recorded in waters within the California portion of the Study Area (Barlow & Forney, 2007; Campbell et al., 2015; Jefferson et al., 2014; Mate et al., 2016b, 2017; Mizroch et al., 2009; Širović et al., 2016; Širović et al., 2004; Širović et al., 2015b; Smultea & Jefferson, 2014). As demonstrated by satellite tags and discovery tags[2], fin whales make long-range movements along the entire U.S. west coast (Falcone et al., 2011; Mate et al., 2015c; Mizroch et al., 2009). However, photo-identification studies of fin whales off the U.S. west coast suggest that not all fin whales undergo long seasonal migrations, but instead make short seasonal trips in spring and fall (Falcone et al., 2011; Falcone & Schorr, 2011).

The movements of six fin whales with satellite tags were shown to be highly variable, ranging from less than 1 to over 200 km from the California coast and as far north as the Oregon border and as far south as the central Baja California Peninsula, Mexico (Mate et al., 2015c). Fin whales frequently congregate in areas with high primary productivity, often in upwelling zones, which may indicate high concentrations of prey. Fin whales are known to be highly adaptable to varying habitat and tend to following prey, which frequently aggregates off the continental shelf (Azzellino et al., 2008; Panigada et al., 2008).

Based on predictive habitat-based density models derived from line-transect survey data collected between 1991 and 2018 off the U.S. west coast, relatively high densities of fin whales are predicted off southern California during the summer and fall with fewer occurring in winter and spring (Barlow et al., 2009a; Becker et al., 2020; Becker et al., 2012a; Becker et al., 2022a; Calambokidis et al., 2024; Forney et al., 2012). Aggregations of fin whales are present year-round in southern and central California

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[2] As a means of data collection starting in the 1930s, discovery tags having a serial number and return address were shot into the blubber of the whale by scientists and if that whale was later harvested by the whaling industry and the tag "discovered" during flensing, it could be sent back to the researchers providing data on the movement of individual whales.

(Campbell et al., 2015; Douglas et al., 2014b; Forney & Barlow, 1998a; Forney et al., 1995; Jefferson et al., 2014), although their distribution shows seasonal shifts. Sightings from year-round surveys off southern California from 2004 to 2013 show fin whales farther offshore in summer and fall and closer to shore in winter and spring (Campbell et al., 2015; Douglas et al., 2014b).

During the first phase of BIA development, the best available science was not sufficient to define BIAs for fin whale behavior off California (Calambokidis et al., 2015b). A combination of sightings, satellite tagging data, and habitat-based distribution models has since enabled researchers to define fin whale feeding BIAs along the west coast (Calambokidis et al., 2024). The parent BIA encompasses 315,000 km^2 and 38 percent of the U.S. West Coast EEZ and is the largest BIA designated off the West Coast (Figure C-9). The child BIA is 49 percent of the parent BIA (approximately 154,350 km^2), which is still a large area that does not effectively identify important core feeding areas within the child BIA. Additional distribution data are needed to more precisely delineate key feeding areas (Calambokidis et al., 2024).

A fin whale reproductive BIA was considered off Hawaii but was not developed due to a lack occurrence and distribution data, particularly in winter when the whales are most likely to occur in Hawaiian waters (Kratofil et al., 2023).

C.6.2.2.3 Population Trends

No data are available on the current population trends for fin whales in Hawaii (Carretta et al., 2023b). The available abundance estimates for the Hawaii stock of fin whales are based on sighting data collected in summer and fall (Bradford et al., 2021), when fin whales are not likely present in large numbers. Further, as noted previously the uncertainty associated with these abundance estimates is quite high (i.e., CVs close to 1.0), thus prohibiting a robust assessment of trends.

For waters off California, Moore and Barlow (2011) predicted continued increases in fin whale abundance into the present decade, and suggested that fin whale densities are reaching “current ecosystem limits.” Based on a comparison of sighting records from the 1950s to 2012, Smultea and Jefferson (2014) also showed an increase in the relative abundance of fin whales inhabiting the Study Area. Širović et al. (2015b) used passive acoustic monitoring of fin whale calls to estimate the spatial and seasonal distribution of fin whales in the Southern California Bight. An increase in the number of calls detected between 2006 and 2012 suggested that the population of fin whales off the U.S. west coast has been increasing. Based on 18 aerial surveys conducted between 2008 and 2013, fin whales were one of the most common large whales sighted in the Study Area (Jefferson et al., 2014). These findings all indicate the abundance of the California/Oregon/Washington stock has been increasing for decades and that a recovery of the stock dates back to the 1970s and 1980s (Barlow, 1994b). However, there is strong evidence that fin whale abundance increased in the California Current between 1991 and 2018 based on estimates from three methods, analysis of line transect survey data (Barlow, 2016), habitat-based species distribution models (Becker et al., 2020), and a Bayesian population trend analysis (Nadeem et al., 2016). Furthermore, the trend analysis reported by Nadeem et al. (2016) estimated that mean annual abundance of fin whales increased 7.5 percent annually from 1991 to 2014.

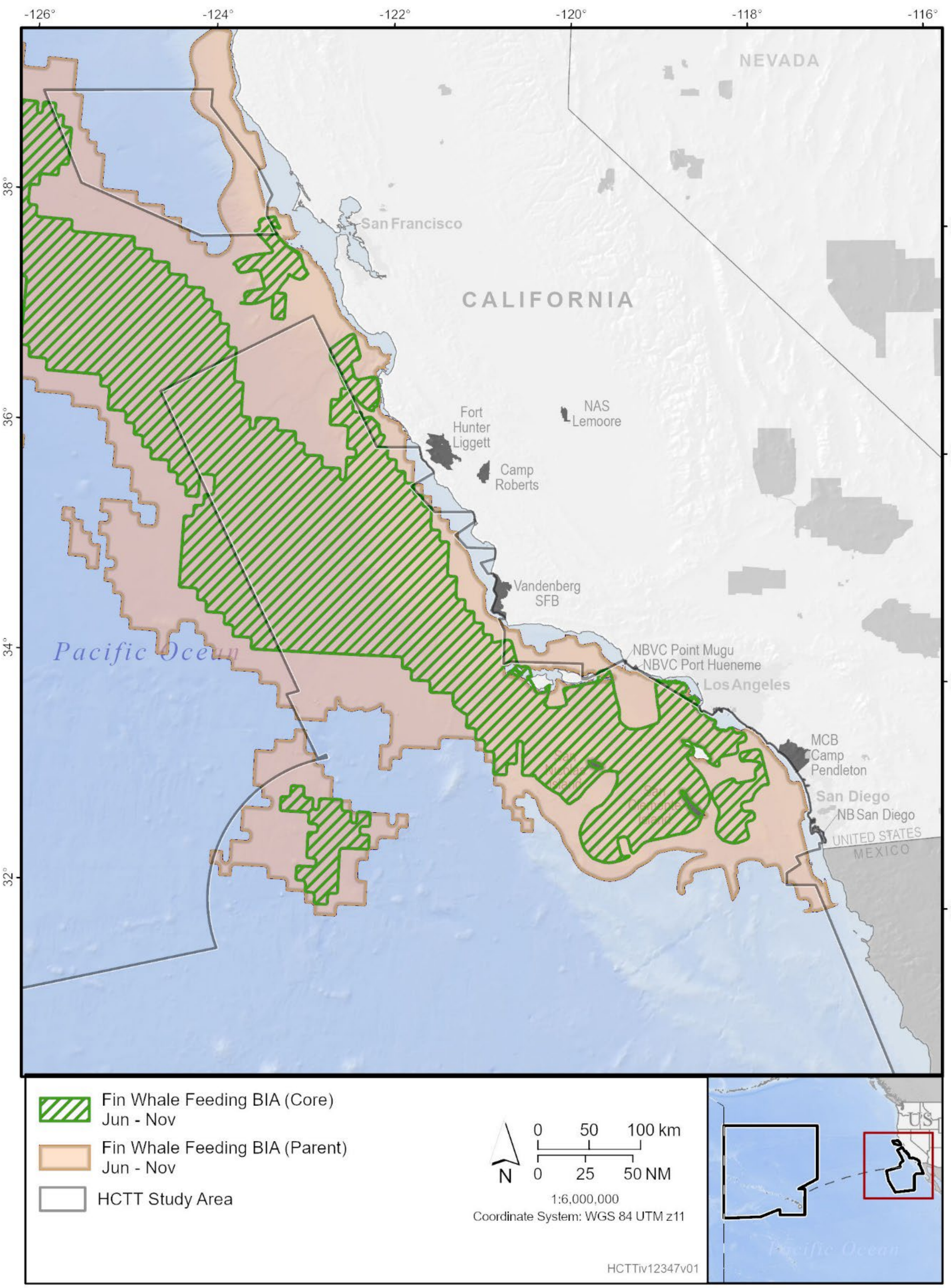


Figure C-9: Fin Whale Feeding BIAs in the California Study Area

C.6.2.2.4 Population Threats

Fin whales are susceptible to both ship strikes and entanglement in fishing gear. The most recent NMFS data from 2017 through 2021 indicate that 8 ship strikes of fin whales along the U.S. West Coast resulted in mortality (Carretta et al., 2023a). There are likely several other unobserved vessel strikes that resulted in serious injury or mortality due to some species not stranding or not appearing to be visibly injured during the time of impact (Carretta et al., 2024).

In the HSTT Study Area, for the 10-year period from 2007 to 2016, there were two Navy ship strikes with large whales and both involved fin whales off southern California and occurred in 2009. In 2021, there were three ship strikes involving large whales, which may have been fin whales due to the offshore location and time of the strikes. Additionally, an Australian Navy vessel was positioning to participate in a U.S. Navy-led exercise in Southern California and struck 2 fin whales during a single incident in 2021. As a result, the Navy has reinitiated the HSTT biological opinion to request additional takes of large whales for the remaining years of the MMPA permit (i.e., mid-2023 through the end of 2025).

The most recent data from 2017 through 2021 indicate that there were two recorded entanglements of the California/Oregon/Washington stock of fin whales in unidentified fisheries, and two entanglements involving unidentified whale species (Carretta et al., 2023a). As a result, mean annual rate of entanglement of fin whales in fisheries along the U.S. West Coast is 0.41 whales annually.

Climate change has increasingly become a threat to marine mammals, including baleen whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.2.3 Gray Whale (*Eschrichtius robustus*; Western North Pacific Stock)

C.6.2.3.1 Status and Management

NMFS recognizes two stocks of gray whales in the North Pacific: the Eastern North Pacific stock and the Western North Pacific stock (Weller et al., 2013). Both stocks could be present in the California portion of the Study Area during their northward and southward migrations (Mate et al., 2015b; Sumich & Show, 2011). The Western subpopulation has previously been known as the Korean-Okhotsk population (Carretta et al., 2023b). This stock is critically endangered, shows no apparent signs of recovery, and should be very rare in the California Study Area given their low abundance. The two stocks are genetically distinguished but the population may be changing due to evolving migratory patterns of both stocks (Carretta et al., 2023b).

Recovery Goals

Recovery goals have not been established for the Western North Pacific stock of gray whales.

C.6.2.3.2 Habitat and Geographic Range

Gray whales are not expected to occur regularly in the Hawaii Study Area or in the transit corridor. There were two sightings of what appeared to be a juvenile gray whale off the island of Hawaii in February 2022, but this is the first recorded sighting of this species in the central Tropical Pacific and it is considered very unusual (Baird et al., 2022).

Earlier sighting data suggested that the western gray whale population had a limited range extending between the Sea of Okhotsk, off the coast of Sakhalin Island (Russia), and the South China Sea (Weller et al., 2002). However, subsequent long-term studies of tagged whales, improved photographic identification, and genetic studies have since indicated that the coastal waters of eastern Russia, the Korean Peninsula, and Japan are a segment of the Western North Pacific stock's migration route, and that "Sakhalin" whales have been detected along the North American coast from British Columbia, Canada as far south as the Baja California Peninsula, Mexico (Mate et al., 2015b; Muir et al., 2016; Weller et al., 2013; Weller et al., 2002; Weller et al., 2012a; Weller et al., 2012b). A total of 43 whales initially photographed and identified in summer feeding grounds off Sakhalin Island were later identified in breeding areas off Mexico, representing 14 percent of the Sakhalin gray whale sub-population (Carretta et al., 2023b). The migratory route taken by these gray whales would have led them through nearshore waters of the HCTT Study Area. The vast majority of gray whales occurring off the West Coast and in the California Study Area are from the larger Eastern North Pacific stock (Carretta et al., 2023b). For purposes of this analysis, the Navy assumed that a very small percentage of gray whales migrating off California could be individuals from the endangered Western North Pacific stock and that the majority would be from the unlisted Eastern North Pacific stock.

C.6.2.3.3 Population Trends

Based on photo identification data collected off Sakhalin and Kamchatka in 2016, the Western North Pacific stock of gray whales was estimated at 290 whales, and given the combined Sakhalin and Kamchatka populations, there was an approximate 2 to 5 percent annual increase between 2005 and 2016 (Cooke, 2018). There have been 18 western gray whales identified in waters far enough south to have passed through the HCTT Study Area (National Marine Fisheries Service, 2014f). In a comparison of Western North Pacific and Eastern North Pacific gray whale photo ID catalogs, 12 Western North Pacific whales identified off Russia have been identified along the U.S. west coast, with an approximately equal number of both sexes (Weller et al., 2012b).

C.6.2.3.4 Population Threats

Gray whales have historically been harvested by subsistence hunters in Alaska and Russia. The International Whaling Commission sets catch limits on the annual subsistence harvest for these areas. For example, the Chukotka indigenous hunters (located on the Chukchi Peninsula) took a total of 127 gray whales in 2013 (Ilyashenko & Zharikov, 2014). In 2010, a gray whale discovered dead onshore in Humboldt, California had two embedded harpoons in its flesh; one of these harpoons had 10 m of rope attached (Carretta et al., 2016a).

Gray whales are also susceptible to entanglement in fishing gear and ship strikes, particularly incidental catches in coastal fisheries (Carretta et al., 2021b). Based on photographic data of western gray whales on their feeding ground off Sakhalin Island (Russia), approximately 19 percent of whales in the sample had detectable anthropogenic scarring resulting from fishing gear entanglement (Bradford et al., 2009).

Natural population threats to gray whales include disease, parasites, reduced prey availability, and predator attacks. The gray whale is preyed on particularly by killer whales. Many individuals exhibit attack scars indicating not all attacks are fatal, however fatalities are known. Killer whale predation of gray whales has been documented in California waters off Monterey (PHYSORG, 2017) as well as the targeting of calves during the spring migration into colder northern waters (Jones & Swartz, 2009).

C.6.2.4 Humpback Whale (*Megaptera novaeangliae;* Mexico and Central America Distinct Population Segments)

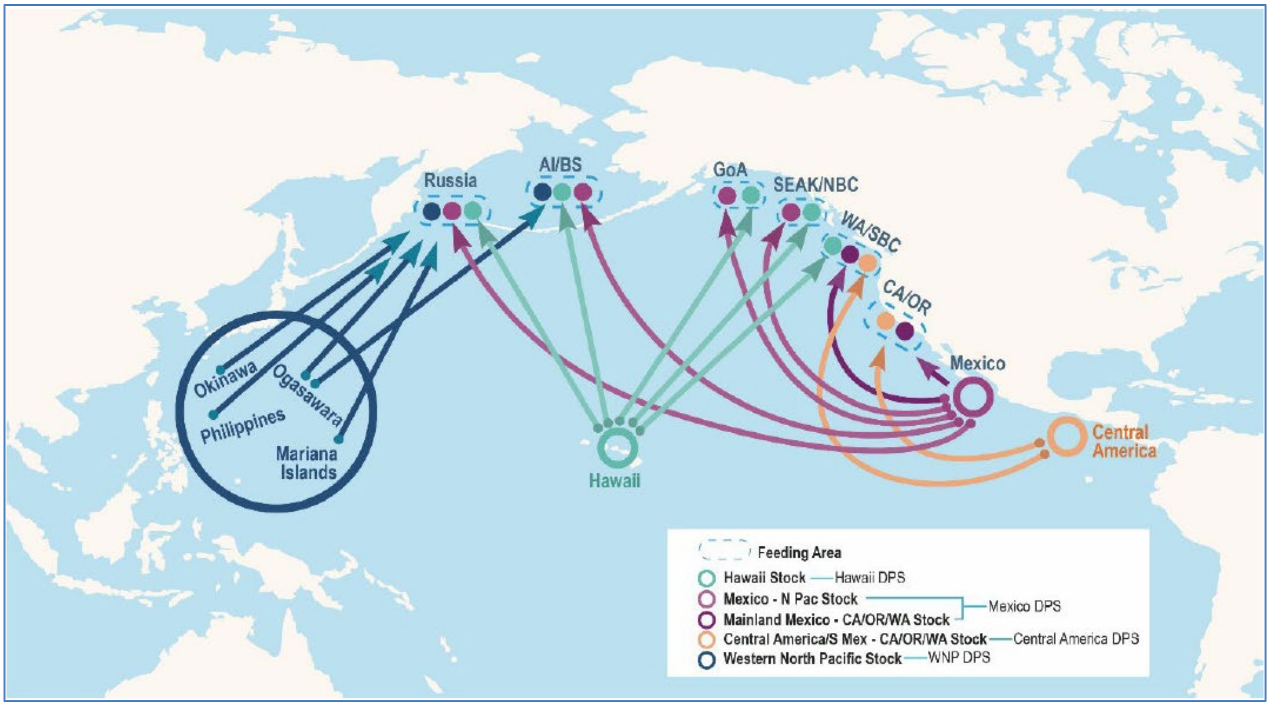
C.6.2.4.1 Status and Management

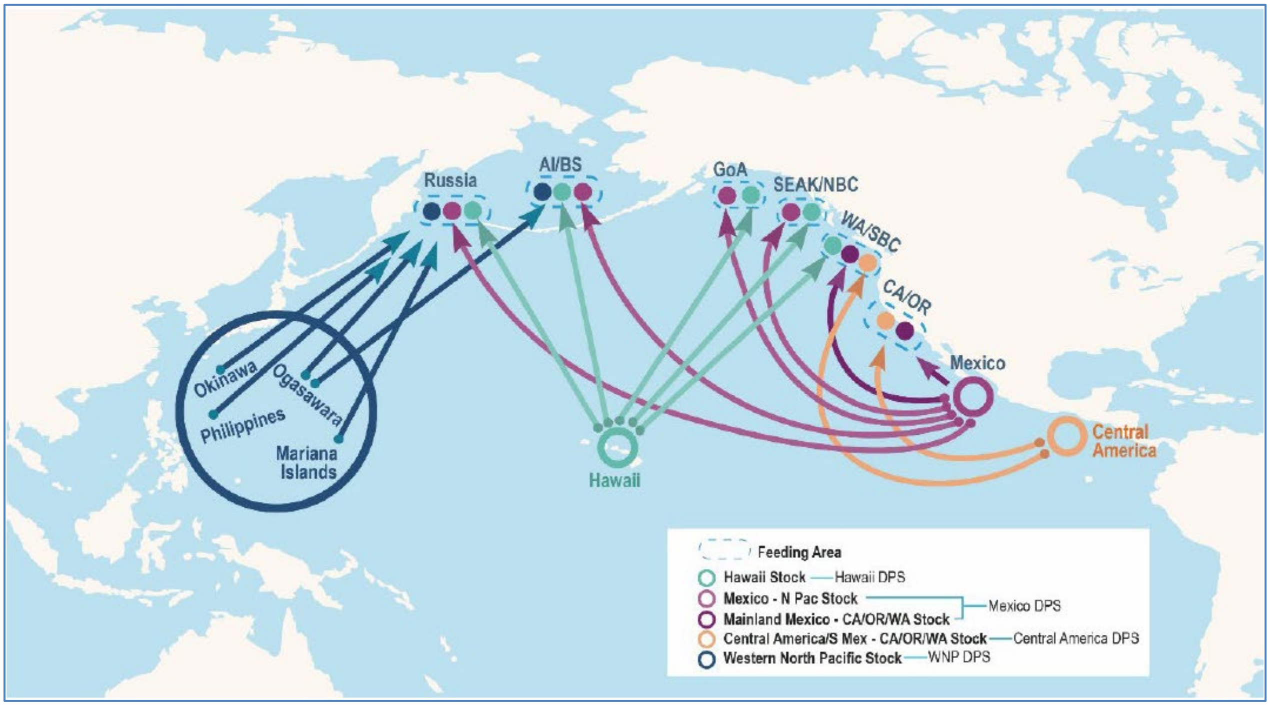
NMFS has identified 14 DPSs of humpback whales worldwide, with 4 DPSs occurring in the North Pacific (Carretta et al., 2023b). Humpback whales that occur seasonally in the HCTT Study Area are from three of the four DPSs identified by low-latitude wintering habitats: Hawaii DPS, Mexico DPS, and Central America DPS (Bettridge et al., 2015a; Carretta et al., 2023b; National Marine Fisheries Service, 2016q; Young, 2023). The three previously defined stocks of North Pacific humpback whales did not align with the DPS structure, so NMFS reevaluated the stock structure to incorporate both the locations of foraging and overwintering areas and population demographics. As a result, NMFS defined five stocks in the North Pacific:

1. Central America/Southern Mexico-California-Oregon-Washington stock
2. Mainland Mexico-California-Oregon-Washington stock
3. Mexico-North Pacific stock
4. Hawaii stock
5. Western North Pacific stock

Humpback whales from the first four stocks listed above occur seasonally in the HCTT Study Area; humpbacks from the Western North Pacific stock do not occur in the Study Area. Humpback whales wintering in Hawaii are identified as the Hawaii DPS and comprise the Hawaii stock. Humpback whales from the Hawaii DPS/stock forage across the North Pacific (Figure C-10). Humpback whales from the Mexico DPS migrate to summer foraging habitat from California northward along the U.S. West Coast, Canada, Alaska, into the Bering Sea, and off the coast of Russia and are divided into the Mexico-North Pacific stock and the Mainland Mexico-California-Oregon-Washington stock. Humpback whales from the Central America DPS forage is waters off California and the Pacific Northwest and make up the Central America/Southern Mexico-California-Oregon-Washington stock (Carretta et al., 2023b). For additional information on the revised stock and DPS structure and population dynamics refer to Martien et al. (2023), Wade (2021), Martien et al. (2021) and Martien et al. (2019).

Humpback whales in the Hawaii DPS are not listed under the ESA, because the population is believed to have fully recovered to its pre-whaling abundance (Barlow et al., 2011; Bettridge et al., 2015a; Muto et al., 2017; National Marine Fisheries Service, 2016j; Wade et al., 2016). Humpback whales from the Mexico DPS are listed as threatened and those from the Central America DPS are listed as endangered under the ESA (National Marine Fisheries Service, 2016j).


Russia
AI/BS
GoA
SEAK/NBC
WA/SBC
CA/OR
Mexico
Central America
Hawaii
Okinawa
Ogasawara
Philippines
Mariana Islands
Feeding Area
○ Hawaii Stock —— Hawaii DPS
○ Mexico - N Pac Stock
○ Mainland Mexico - CA/OR/WA Stock
Mexico DPS
○ Central America/S Mex - CA/OR/WA Stock —— Central America DPS
○ Western North Pacific Stock —— WNP DPS


Russia
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Mariana Islands
Feeding Area
○ Hawaii Stock —— Hawaii DPS
○ Mexico - N Pac Stock
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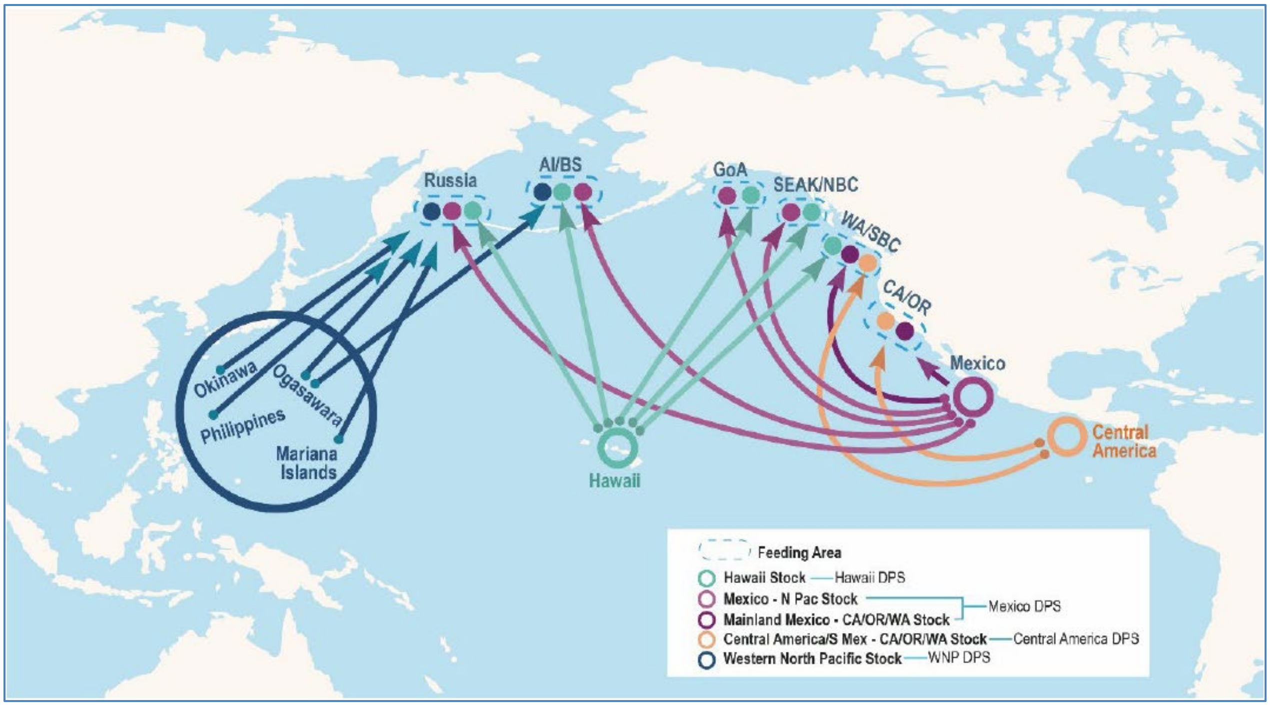
○ Western North Pacific Stock —— WNP DPS



AI/BS = Aleutian Islands/Bering Sea, GoA = Gulf of Alaska, SEAK/NBC = Southeast Alaska/Northern British Columbia, WA/SBC = Washington/Southern British Columbia, CA/OR = California Oregon. Source: Carretta et al. (2023b)

Figure C-10: Humpback Whale Stocks and DPSs Defined in the North Pacific. Whales From the Hawaii, Mexico, and Central America DPSs Occur Seasonally in the Study Area.

Critical Habitat

NMFS designated critical habitat for the Mexico and Central America DPSs of humpback whale on April 21, 2021 (86 FR 21082). The wintering areas for the Mexico DPS are the waters and islands off Mexico, and, for the Central America DPS, wintering areas are waters from southern Mexico and south along the coast of Central America (Calambokidis et al., 2008). Critical habitat for both DPSs extends beyond the Study Area but includes the California Current Ecosystem, which overlaps with nearshore potions of the Study Area but does not include all of the Southern California Bight (Figure C-11). The critical habitat is defined by water depth, with the nearshore limit set at 15 m, 30 m, or 50 m and the offshore limit at 2,000 m, 3,000 m, or 3,700 m for different segments along the California coastline. Prey species (primarily euphausiids and small schooling fishes) were identified by NMFS as the essential feature of the designated critical habitat.

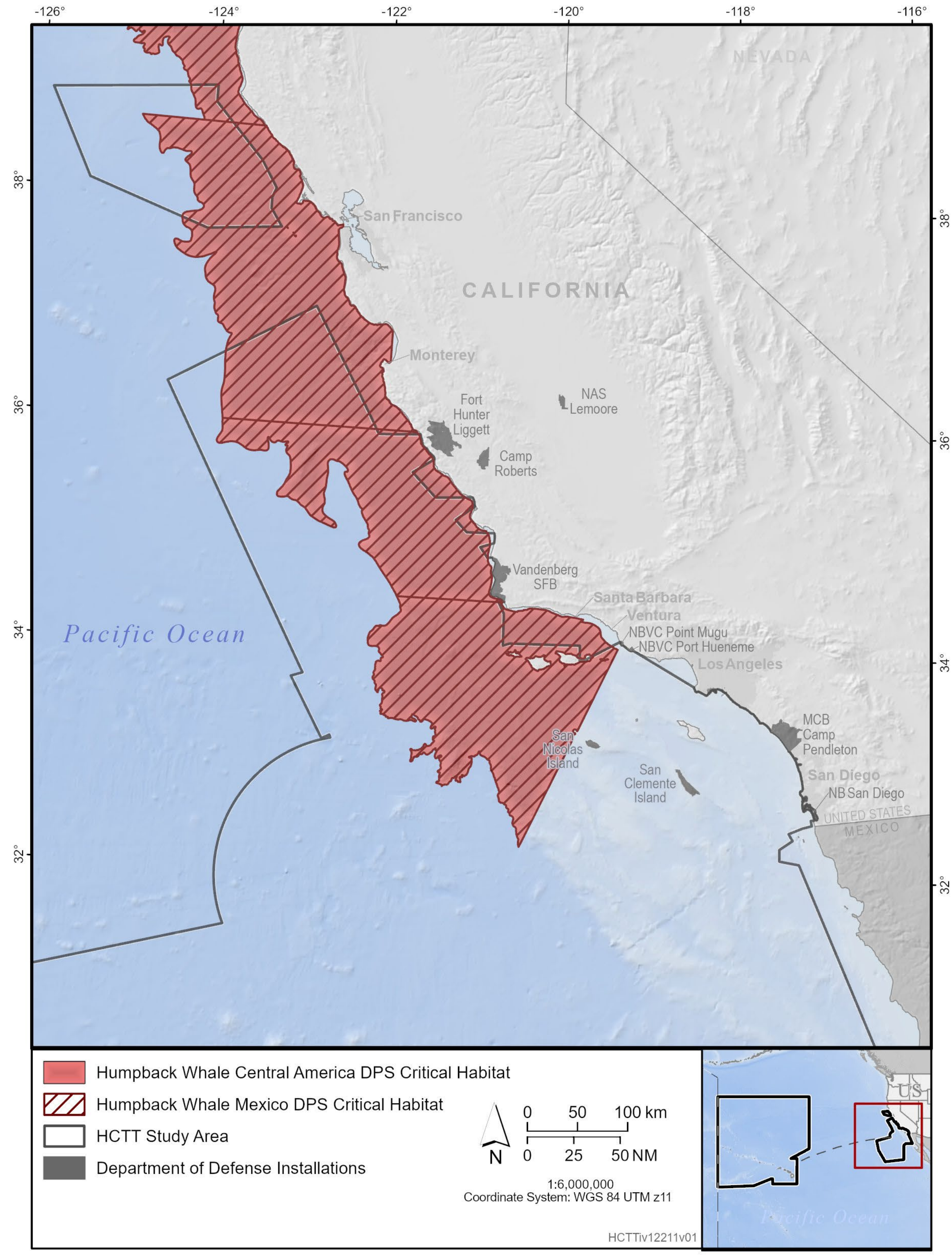


Figure C-11: Humpback Whale Critical Habitat in the HCTT Study Area

Recovery Goals

In response to the threats facing the species, NMFS developed goals to recover humpback whale populations. The 1991 Final Recovery Plan (National Marine Fisheries Service, 1991) for the humpback whale provides the complete down listing/delisting criteria for each of the four following recovery goals (these apply to the Mexico DPS and the Central America DPS of humpback whale):

1. Maintain and enhance habitats used by humpback whales currently or historically.
2. Identify and reduce direct human-related injury and mortality.
3. Measure and monitor key population parameters.
4. Improve administration and coordination of recovery program for humpback whales.

C.6.2.4.2 Habitat and Geographic Range

Humpback whales are distributed worldwide in all major oceans and most seas (Bettridge et al., 2015a; National Marine Fisheries Service, 2016j). They typically are found during the summer in high-latitude feeding grounds, including Alaska and British Colombia, and during the winter migrate to breeding areas off Hawaii, Mexico, Central America, and Okinawa where breeding and calving occurs. As a result, humpback migrations are complex and cover great distances (Bettridge et al., 2015a; Calambokidis et al., 2009b; Calambokidis et al., 2008). Whales migrating from Hawaii to summer feeding habitat in the Gulf of Alaska will cover 2,600 NM over several weeks. Mate et al. (1998), used satellite tags to track three migrating whales using independent routes to Alaskan waters, and the fastest averaged 93 NM per day. At that rate, the whale would have reached waters off the coast of Alaska in about 39 days.

Off the U.S. west coast, humpback whales are more abundant in shelf and slope waters (<2,000 m deep), and are often associated with areas of high productivity (Becker et al., 2020; Becker et al., 2010; Becker et al., 2012b; Becker et al., 2014; Forney et al., 2012; Redfern et al., 2013). While most humpback whale sightings are in nearshore and continental shelf waters, humpback whales frequently travel through deep oceanic waters during migration (Calambokidis et al., 2001; Clapham, 2000; Clapham & Mattila, 1990; Mate et al., 1997). Humpback whales migrating from breeding grounds in Central America to feeding grounds at higher latitudes may cross the SOCAL portion of the HCTT Study Area including the Transit Corridor located farther offshore. While most humpback whales migrate, data from surveys conducted between 2004 and 2013 show that humpback whales occur year-round off southern California (Campbell et al., 2015). Peak occurrence during migration occurs in the California Study Area from December through June (Calambokidis et al., 2015a). During late summer, more humpback whales are sighted north of the Channel Islands, and limited occurrence is expected south of the northern Channel Islands (San Miguel, Santa Rosa, Santa Cruz) (Carretta et al., 2010). Based on aerial survey data collected between 2008 and 2012 in the California Study Area, Smultea and Jefferson (2014) determined that humpback whales ranked eighth in relative occurrence and concluded that this species has clearly increased their representation in the Navy's SOCAL Range Complex over the last several decades.

Two BIAs (parent and core) for humpback whale feeding behavior have been identified in the California Study Area (Figure C-12) (Calambokidis et al., 2024). The BIAs are only in effect from March through November when foraging humpback whales are expected to be present.

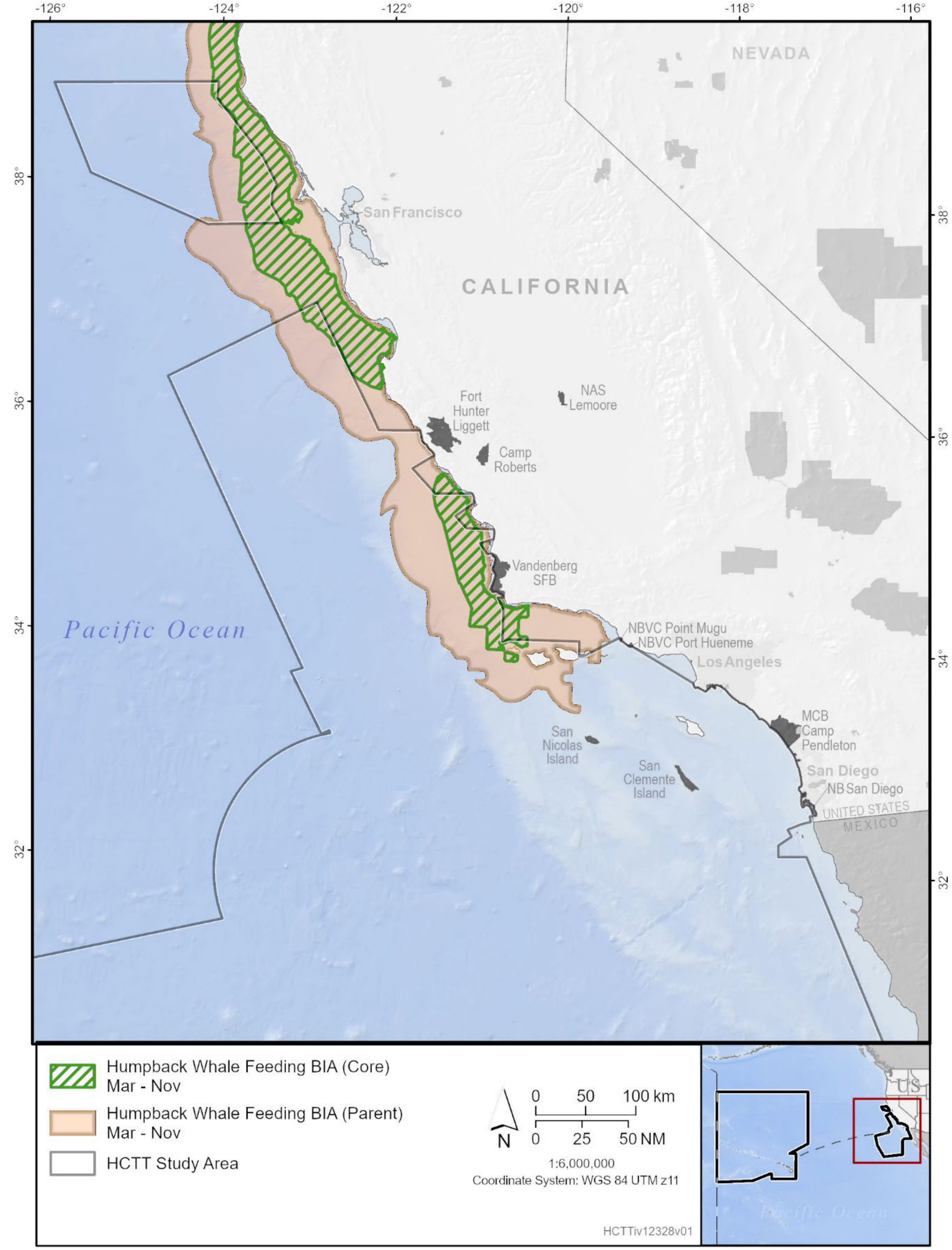


Figure C-12: Humpback Whale Feeding BIAs in the California Study Area

C.6.2.4.3 Population Trends

Humpback whale abundance off the U.S. west coast has appeared to increase at a rate of approximately 8 percent per year since 1989 (Calambokidis & Barlow, 2020). However, since multiple humpback whale stocks occur in these waters, this overall trend cannot be assumed for each of the individual stocks. Based on capture-recapture analyses of photographic identification data collected between 2019 and 2021, there appears to be an annual growth rate of 1.6 percent for the Central America/Southern Mexico – CA/OR/WA stock of humpback whales (Curtis et al., 2022). Stock-specific trend data are not available for the Mainland Mexico – CA/OR/WA stock of humpback whales (Carretta et al., 2023b). Since some of the individuals from the Mexico-North Pacific stock of humpback whales feed in Alaska, and there have been recent declines in the numbers of humpback whale in these feeding grounds, it is unknown if there is an increasing or decreasing population trend for this stock (Young, 2023).

C.6.2.4.4 Population Threats

Humpback whales are susceptible to entanglement in fishing gear and ship strikes. Pot and trap fishery entanglements are the most-common source of injury to humpback whales along the U.S. west coast (Carretta et al., 2022). Between 2015 through 2019, there have been 81 observations of humpback whale interactions with pot and trap fisheries (Carretta et al., 2022). Of these interactions, serious injuries and mortalities were calculated as 51.75 humpback whales during this time period (Carretta et al., 2022). From 2015 to 2019, there was 79 humpback whale interactions with gillnet and unidentified fisheries (Carretta et al., 2022). Gillnet related serious injuries and mortalities totaled to 54.75 for this time period (Carretta et al., 2022). Humpback whales have also been reported seriously injured and killed from entanglement in fishing gear while in their Alaskan feeding grounds (Helker et al., 2017). Humpback whales from Mexico and Central America have been identified feeding in Alaska (Bettridge et al., 2015b; Calambokidis et al., 2008). Some proportion of these entanglements could be to be whales from the Mexico DPS and from the Central America DPS. An overall minimum estimate of mortality and serious injury due to fisheries in Alaska is 14 humpback whales annually (Muto et al., 2017).

Available data from NMFS indicate that there have been 14 reports of humpback whales struck by ships from 2016 through 2020, totaling to 2.6 humpback whale serious injuries or mortalities per year (Carretta et al., 2022). The mean vessel collision mortality and serious injury rate in Alaska is 4.3 humpback whales annually (Muto et al., 2017).

Humpback whales are also potentially affected by underwater noise, jet skis and similar fast waterborne tourist-related traffic, and pollutants (Muto et al., 2017).

C.6.2.5 Sei Whale (*Balaenoptera borealis*)

C.6.2.5.1 Status and Management

The sei whale is listed as endangered under the ESA and as depleted under the MMPA, but there is no designated critical habitat for this species. Sei whales in Hawaii are assigned to the Hawaii stock. Along the U.S. west coast, the Eastern North Pacific stock is recognized within the U.S. EEZ including the California Study Area (Carretta et al., 2023b).

Recovery Goals

In response to the threats facing the species, NMFS developed goals to recover sei whale populations. The 2011 Final Sei Whale Recovery Plan (National Marine Fisheries Service, 2011a) provides the complete downlisting/delisting criteria for both of the following objectives, 1) Achieve sufficient and viable populations in all ocean basins, and 2) Ensure significant threats are addressed. The plan outlines the following recovery actions to accomplish these objectives:

- Coordinate state, federal, and international actions to implement recovery actions and maintain international regulation of whaling for sei whales.
- Develop and apply methods to collect sei whale data.
- Support existing studies to investigate population discreteness and population structure of sei whales using genetic analyses.
- Continue to collect data on “unknown” threats to sei whales.
- Maximize efforts to acquire scientific information from dead, stranded, and entangled sei whales.
- Estimate population size and monitor trends in abundance.
- Initiate new studies to determine population discreteness and population structure of sei whales.

C.6.2.5.2 Habitat and Geographic Range

Sei whales have a worldwide distribution and are found primarily in cold temperate to subpolar latitudes. During the winter, sei whales are found in warm tropical waters like Hawaii. Sei whales are also encountered during the summer off California and the North America coast from approximately the latitude of the Mexican border to as far north as Vancouver Island, Canada (Horwood, 2009; Masaki, 1976, 1977; Smultea et al., 2010). Although sei whales have been observed south of 20° N in the winter (Fulling et al., 2011; Horwood, 2009; Horwood, 1987), they are considered absent or at very low densities in most equatorial areas. Sei whales are typically found in the open ocean and are rarely observed near the coast (Horwood, 2008). Whaling data provide some evidence of differential migration patterns by reproductive class, with females arriving at and departing from feeding areas earlier than males (Horwood, 1987; Perry et al., 1999). Although there appears to be some seasonal movement between higher latitude summer feeding areas and lower latitude winter calving areas, these movements are not as extensive as those of many other baleen whale species (Jefferson et al., 2015).

Sei whales are not commonly detected in waters of the Hawaiian Islands. Sei whales were not sighted during aerial surveys conducted within 25 NM of the main Hawaiian Islands from 1993 to 1998 (Mobley et al., 2000). The first verified sei whale sighting was made during a NMFS survey of the Hawaiian Island EEZ in 2002 (Barlow, 2006). A subsequent sighting that included subadults made nearshore of the main Hawaiian Islands occurred in 2007 and was cited as evidence suggesting that the area north of the main Hawaiian Islands may be part of a reproductive area for North Pacific sei whales (Smultea et al., 2010). On March 18, 2011 off Maui, the Hawaiian Islands Entanglement Response Network found a subadult sei whale entangled in rope and fishing gear (Bradford & Lyman, 2015c; National Marine Fisheries Service, 2011c). An attempt to disentangle the whale was unsuccessful although a telemetry buoy attached to the entangled gear was reported to be tracking the whale over 21 days as it moved north and over 250 NM from the Hawaiian Islands. During three systematic ship surveys of the Hawaiian Islands EEZ between 2002 and 2017, there were a total of eight sei whale sightings, allowing for the derivation of an updated design-based abundance estimate for the Hawaiian Islands fin whale stock of 401 (CV = 0.84) whales (Bradford et al., 2021). In the summer, sei whales are likely absent from low productivity tropical waters (Jefferson et al., 2015), and during two separate line-transect surveys of the Hawaiian Islands EEZ during summer and fall, sei whales were only seen during the fall months (Barlow, 2006; Bradford et al., 2017).

Sei whales are distributed in offshore waters in the California Study Area (Carretta et al., 2023b). A total of 25 sei whale sightings were made during systematic ship surveys conducted off the U.S. west coast in summer and fall between 1991 and 2014 (Barlow, 2016), and an additional 4 sightings were made during a 2018 survey (Henry et al., 2020). Sei whales were not seen in the Southern California Study Area (or the larger Southern California Bight) during 15 aerial surveys conducted from 2008 through 2012 (Smultea et al., 2014) or during any systematic ship surveys conducted by NMFS (Barlow, 2010, 2016).

Sei whales are likely present in the Transit Corridor portion of the Study Area, and are seen at least as far south as 20° N into the North Pacific Gyre (Horwood, 2009; Horwood, 1987).

C.6.2.5.3 Population Trends

No data are available on current population trends for the Hawaii stock of sei whales in the HCTT Study Area (Carretta et al., 2023b). There has not been a formal trend analysis conducted for the Eastern North Pacific stock of sei whales, but design-based abundance estimates made from 1991 to 2014 show an increasing population trend for sei whales off the U.S. West Coast, with the 2014 estimate being the highest to date (Barlow, 2016). This increase could be due in part to this population's recovery from commercial whaling, or may also represent distribution shifts in these waters (Carretta et al., 2023b).

C.6.2.5.4 Population Threats

Sei whales off the U.S. West Coast are most likely to interact with the California swordfish drift gillnet fishery; however, there were not any observed entanglements in monitored fishing sets from 1990 through 2021 (Carretta, 2022a; Carretta et al., 2023a). Additionally, data from 2012 through 2016 indicate that there has only been one documented vessel strike of this species off the U.S. West Coast (Carretta et al., 2018; Carretta et al., 2023b). Most recent NMFS data from 2017 through 2021 indicate that there were no documented vessel strikes of sei whales during the five-year period (Carretta et al., 2023a).

Climate change has increasingly become a threat to marine mammals, including baleen whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.2.6 Sperm Whale (*Physeter macrocephalus*)

C.6.2.6.1 Status and Management

The sperm whale has been listed as endangered since 1970 under the precursor to the ESA (National Marine Fisheries Service, 2009b), and is depleted under the MMPA, but there is no designated critical habitat for this species in the North Pacific. Sperm whales are divided into three stocks in the Pacific; two (Hawaii and California/Oregon/Washington) occur within the Study Area (Carretta et al., 2023b). Based on genetic analyses, Mesnick et al. (2011) found that sperm whales in the California Current are demographically independent from animals in Hawaii and the eastern tropical Pacific.

Recovery Goals

In response to the threats facing the species, NMFS developed goals to recover sperm whale populations. The 2010 Final Sperm Whale Recovery Plan (National Marine Fisheries Service, 2010c) provides the complete downlisting/delisting criteria. The two main objectives for sperm whale recovery are to 1) achieve sufficient and viable populations in all ocean basins, and 2) ensure significant threats are addressed. The plan outlines the following recovery actions to accomplish these objectives:

- Coordinate state, federal, and international actions to implement recovery actions and maintain international regulation of whaling for sperm whales.
- Develop and apply methods to estimate population size and monitor trends in
- abundance.
- Determine population discreteness and population structure of sperm whales.
- Conduct Risk Analyses.
- Identify, characterize, protect, and monitor habitat important to sperm whale populations in U.S. waters and elsewhere.
- Investigate causes of and reduce the frequency and severity of human-caused injury and mortality.
- Determine and minimize any detrimental effects of anthropogenic noise in the oceans.
- Maximize efforts to acquire scientific information from dead, stranded, and entangled sperm whales.
- Develop post-delisting monitoring plan.

C.6.2.6.2 Habitat and Geographic Range

The sperm whale is one of the most widely distributed cetacean species and occurs throughout the entire Study Area. Primarily, this species is typically found in the temperate and tropical waters of the Pacific (Rice, 1989), with a secondary range that includes areas at latitudes higher than 50 degrees in the North Pacific (Jefferson et al., 2015; Whitehead, 2009; Whitehead et al., 2008; Whitehead & Weilgart, 2000). This species appears to have a preference for deep waters (Baird et al., 2013c; Jefferson et al., 2015). Typically, sperm whale concentrations correlate with areas of high productivity. These areas are generally near drop offs and areas with strong currents and steep topography (Gannier & Praca, 2007; Jefferson et al., 2015).

Sperm whales are somewhat migratory. General shifts occur during summer months for feeding and breeding, while in some tropical areas, sperm whales appear to be largely resident (Rice, 1989; Whitehead, 2003, 2009; Whitehead et al., 2008). Pods of females with calves remain on breeding grounds throughout the year, between 40° N and 45° N (Rice, 1989; Whitehead, 2003), while males migrate between low-latitude breeding areas and higher-latitude feeding grounds (Pierce et al., 2007). In the northern hemisphere, “bachelor” groups (males typically 15 to 21 years old and bulls [males] not taking part in reproduction) generally leave warm waters at the beginning of summer and migrate to feeding grounds that may extend as far north as the perimeter of the arctic zone. In fall and winter, most return south, although some may remain in the colder northern waters during most of the year (Pierce et al., 2007). Sperm whales occur in Hawaii waters year-round and are one of the more abundant large whales found in that region (Baird et al., 2003b; Barlow, 2006; Bradford et al., 2017; Mobley et al., 2000). A total of 109 sperm whale sightings were made during three summer/fall systematic shipboard surveys of waters within the U.S. EEZ of the Hawaiian Islands (Barlow, 2006; Bradford et al., 2021). Based on predictive habitat-based density models derived from line-transect survey data collected between 1997 and 2012 within the central North Pacific, relatively high densities of sperm whales were predicted within the U.S. EEZ of the Hawaiian Islands during the summer and fall, particularly in the northwest (Forney et al., 2015). Habitat-based density models derived from line-transect survey data collected between 2002 and 2017 within the U.S. EEZ of the Hawaiian Islands also predicted highest sperm whale densities in the northwest portion of the study area (Becker et al., 2021). Predictions from seasonal habitat-based density models derived from line-transect survey data collected between 2000 and 2020 within the offshore waters of the Main Hawaiian Islands indicated that sperm whale abundance estimates in winter were almost double that of non-winter, although the difference was not found to be statistically significant (Becker et al., 2022a).

Sperm whales are found year-round in California waters, but their abundance is temporally variable, most likely due to the availability of prey species (Barlow, 1995; Barlow & Forney, 2007; Forney & Barlow, 1993; Smultea & Jefferson, 2014). Their distribution is typically associated with waters over the continental shelf break, over the continental slope, and into deeper waters (Carretta et al., 2017b; Rice, 1989; Whitehead, 2003; Whitehead et al., 2008). Based on habitat models derived from line-transect survey data collected between 1991 and 2018 off the U.S. west coast, sperm whales show a strong preference for deep waters (Barlow et al., 2009b; Becker et al., 2020; Becker et al., 2012a; Becker et al., 2010; Forney et al., 2012). During quarterly ship surveys conducted off southern California between 2004 and 2008, there were a total of 20 sperm whale sightings, the majority (12) occurring in summer in waters greater than 2,000 m deep (Douglas et al., 2014b). Only one sperm whale group was observed during 18 aerial surveys conducted in the Southern California Bight from 2008 through 2012 (Smultea et al., 2014).

C.6.2.6.3 Population Trends

Current data are not sufficient to assess a population trend for the Hawaii stock of sperm whales (Carretta et al., 2023b).

Moore and Barlow (2014) used a Bayesian hierarchical approach to examine sperm whale population abundance and trends based on line-transect surveys conducted off the U.S. west coast from 1991 to 2008. Although an estimate of trends was not conclusive, they found that the abundance of adult male sperm whales has increased (Moore & Barlow, 2014). Moore and Barlow (2017b) updated their sperm whale assessment using new data from a NMFS 2014 U.S. west coast survey. While they reported little evidence of increasing trends in overall sperm whale abundance, the new analysis supports prior evidence for an increasing number of sperm whales that occur in small groups (presumed to be adult or near-adult males). However, given high uncertainty associated with the estimated growth rates of this population, a definite trend could not be confirmed (Moore & Barlow, 2017a).

C.6.2.6.4 Population Threats

Sperm whales are susceptible to entanglement in fishing gear and ship strikes. Off the U.S. West Coast, the sperm whales are most likely to interact with the thresher shark/swordfish drift gillnet fishery (Beeson, 1998; Carretta et al., 2019a; Carretta et al., 2019b). Based on California swordfish drift gillnet fishery observer data from 1990 through 2021, NMFS has estimated that the overall probability of a bycatch event resulting in mortality or serious injury to sperm whales from this fishery in 2017 through 2021 was 0.32 whales annually (Carretta, 2022a). Additionally, most recent NMFS data from the same 5-year period indicate that one sperm whale was recorded entangled in unidentified fishing gear (Carretta et al., 2023a). As a result, the annual mean serious injury and mortality of sperm whales resulting from interactions with California drift gillnet fisheries from 2017 through 2021 is 0.52 whales annually. Bradford and Lyman (2015b) recorded one observed interaction between a sperm whale and longline fishing in Hawaiian waters.

Although vessel strikes are considered a potential population threat to sperm whales, most recent data from 2017 through 2021 indicate that there were no observed sperm whale vessel strike deaths on the U.S. West Coast (Carretta et al., 2023a; Carretta et al., 2023b).

Climate change has increasingly become a threat to marine mammals, including sperm whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.2.7 False Killer Whale (*Pseudorca crassidens*; Main Hawaiian Islands Insular stock)

C.6.2.7.1 Status and Management

NMFS currently recognizes three stocks of false killer whale in Hawaiian waters: the Hawaii pelagic stock, the Northwestern Hawaiian Islands stock, and the Main Hawaiian Islands insular stock (Bradford et al., 2015; Carretta et al., 2015; Carretta et al., 2023b; Forney et al., 2010; National Oceanic and Atmospheric Administration, 2012; Oleson et al., 2010a). All stocks of false killer whale are protected under the MMPA. The Hawaii Pelagic stock and the Northwestern Hawaiian Islands stock of false killer whales are not listed as threatened or endangered under the ESA. The Main Hawaiian Islands insular stock is listed as endangered under the ESA as a DPS (National Oceanic and Atmospheric Administration, 2012).

<u>Critical Habitat</u>

In July 2016, a complaint was filed in the U.S. District Court for the District of Columbia (Case 1:16-cv-01442; Filed 07/13/16) by the Natural Resources Defense Council against NMFS claiming that NMFS "failed to timely designate" critical habitat for Main Hawaiian Islands insular false killer whales. On July 24 2018, NOAA Fisheries published a final rule to designate critical habitat for the main Hawaiian Islands insular false killer whale DPS by designating waters from the 45 m depth contour to the 3,200 m depth contour around the main Hawaiian Islands from Niihau east to Hawaii (83 FR 35062, July 24, 2018) (Figure C-13). The critical habitat is described by the following four characteristics or essential features, which enable the whales to forage, communicate, and move freely around and among the waters surrounding the main Hawaiian Islands.

1. Adequate space for movement and use within shelf and slope habitat
2. Prey species of sufficient quantity, quality, and availability to support individual growth, reproduction, and development, as well as overall population growth
3. Waters free of pollutants of a type and amount harmful to main Hawaiian Islands insular false killer whales
4. Sound levels that would not significantly impair false killer whales' use or occupancy.

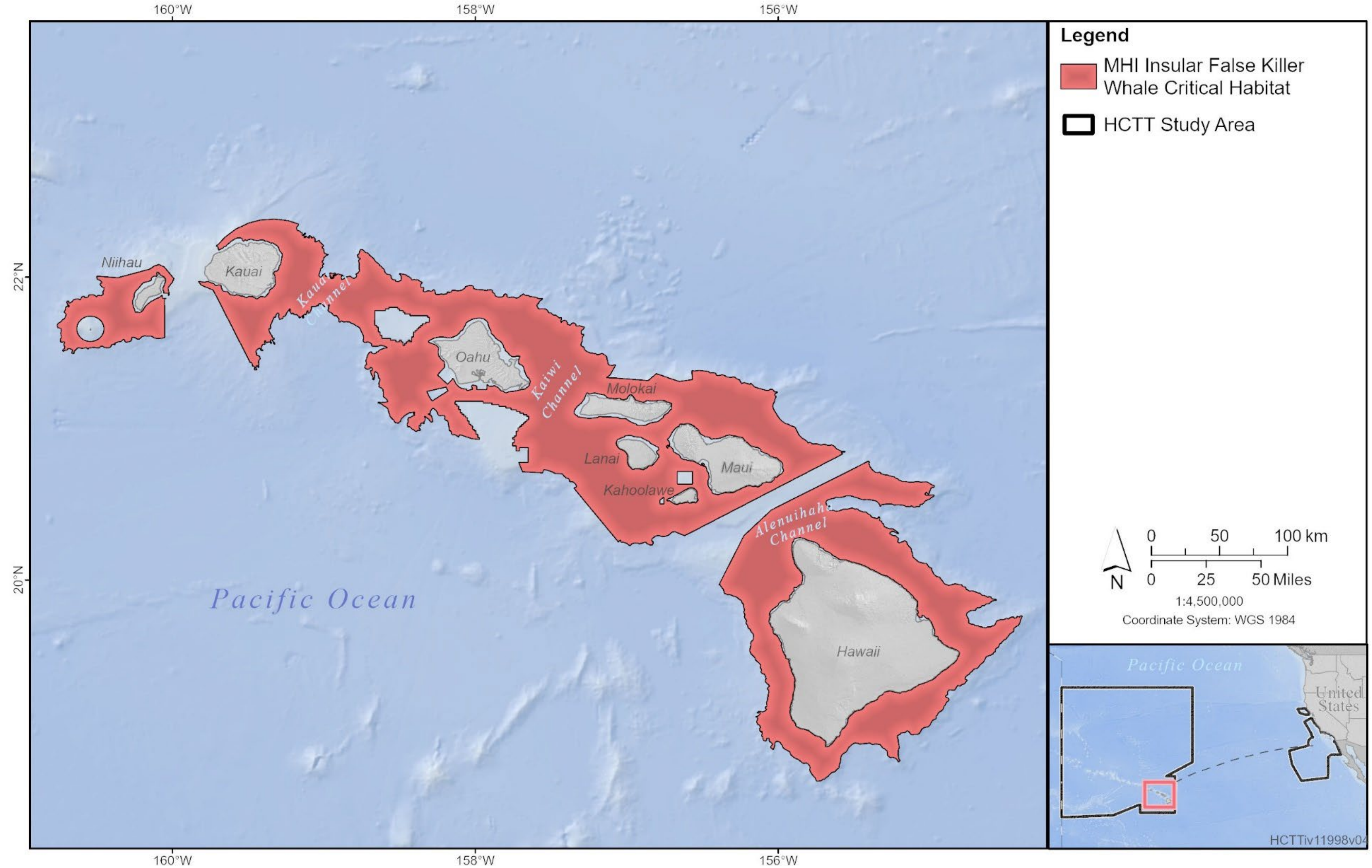


Figure C-13: Critical Habitat for MHI Insular False Killer Whale in the HCTT Study Area

Recovery Goals

In response to the threats facing the species, NMFS developed goals to recover the Main Hawaiian Islands Insular False Killer Whale DPS. The 2021 Final Main Hawaiian Islands Insular False Killer Whale Recovery Plan (National Marine Fisheries Service, 2021a) provides the complete downlisting/delisting criteria. The recovery goal is subdivided into discrete component objectives that, collectively, describe the conditions necessary to achieve recovery. NMFS (2021) identified seven recovery objectives for the Main Hawaiian Islands Insular False Killer Whale that address demographic concerns and threats abatement. The plan then outlines specific recovery actions to accomplish these objectives. The seven objectives are as follows:

- Ensure productivity and social connectedness of the Main Hawaiian Islands Insular False Killer Whale (trend, abundance, and social clusters) have met or exceeded target levels.
- Address threats from fisheries including incidental take and competition for prey.
- Address threats from environmental contaminants and biotoxins.
- Address threats from anthropogenic noise.
- Better understand the effects of climate change and manage accordingly.
- Ensure that regulatory mechanisms, including state and federal management and post-delisting monitoring, are in place prior to delisting.
- Ensure secondary threats and synergies among threats are not limiting recovery of the population.

The species is not typically expected to be present in the California Study Area, and false killer whales are not included by NMFS as a managed species in California waters (Carretta et al., 2017b; Carretta et al., 2023b). Strandings and sightings of false killer whales have been recorded in Southern California and north, but these have generally been considered extralimital. During the unusually warm oceanographic conditions in 2014, whale watching boats photographed false killer whales in Southern California waters, but there were none sighted during the SWFSC systematic survey that year (Barlow, 2016). False killer whales do occur in waters off the Baja California Peninsula, Mexico within the California Study Area (Hamilton et al., 2009).

C.6.2.7.2 Habitat and Geographic Range

This species is known to occur in deep oceanic waters off Hawaii, and elsewhere in the Pacific. False killer whales have been detected in acoustic surveys and are commonly observed in the eastern tropical Pacific, including waters off the Baja California Peninsula, Mexico within the Study Area (Carretta et al., 2015; Carretta et al., 2023b; Miyashita et al., 1996; Oswald et al., 2003; Wade & Gerrodette, 1993; Wang et al., 2001). False killer whale are also regularly found within Hawaiian waters and have been reported in groups of up to 100 over a wide range of depths and distance from shore (Baird et al., 2003b; Baird et al., 2013a; Bradford et al., 2018; Bradford et al., 2014; Bradford et al., 2015; Oleson et al., 2013; Shallenberger, 1981).

The ranges and stock boundary descriptions for false killer whales in the Hawaiian Islands are complex and overlapping. For example, all three stocks are known to overlap in the vicinity of Kauai and Niihau, which is where the Navy's underwater instrumented range has been in use since the 1980s. All significant information regarding the range of the three stocks was presented in Bradford et al. (2015), and later updated for the pelagic stock (Bradford et al., 2020). A summary of the data used to delineate the stock boundaries, and the research supporting those data are provided in the Final 2022 Pacific

Stock Assessment Report (Carretta et al., 2023b) that is synthesized in the next few paragraphs for the stocks in the Hawaiian Islands.

The Main Hawaiian Islands insular stock is considered resident to the main Hawaiian Islands consisting of Kauai, Oahu, Molokai, Lanai, Kahoolawe, Maui, and Hawaii, although they have been satellite tracked as far as 115 km from the main Hawaiian Islands (Bradford et al., 2020; Bradford et al., 2012; Bradford et al., 2015; Carretta et al., 2015; Forney et al., 2010; National Oceanic and Atmospheric Administration, 2012; Oleson et al., 2010a). The Main Hawaiian Islands insular stock boundary is a 72 km radius extending around the main Hawaiian Islands, with the offshore extent of the radii connected on the leeward sides of Hawaii Island and Niihau to encompass the offshore movements of Main Hawaiian Islands insular stock animals within that region.

False killer whales in the Northwestern Hawaiian Islands stock have been seen as far as 93 km from the Northwestern Hawaiian Islands and near shore around Kauai and Oahu (Baird et al., 2012; Bradford et al., 2015). The Northwestern Hawaiian Islands stock boundary is defined by a 93 km radius around Kauai, Niihau, and the Northwestern Hawaiian Islands, with the boundary around the Northwestern Hawaiian Islands expanded latitudinally at the eastern end to encompass animal movements observed outside the 93 km radius.

Given new telemetry data that indicated that pelagic stock animals occurred within 5.6 km of the main Hawaiian Islands and throughout the Northwestern Hawaiian Islands, the previous inner pelagic stock boundary at 11 km from shore around each of the main Hawaiian Islands was removed (Bradford et al., 2020). The pelagic stock now has no inner or outer boundary within the Hawaiian Islands EEZ. There is now an overlap zone between the entirety of the Main Hawaiian Islands insular stock area and the pelagic stock area. There is also now an overlap zone between the entirety of the Northwestern Hawaiian Islands stock area and the pelagic stock area. All three stock boundaries overlap out to the Main Hawaiian Islands insular stock boundary between Kauai and Niihau and the Northwestern Hawaiian Islands stock boundary between Kauai and Oahu (Carretta et al., 2023b).

Two year-round Small and Resident Population BIAs (a parent and a child BIA) have been delineated in the main Hawaiian Islands for the insular stock of false killer whales (Kratofil et al., 2023) (Figure C-14). The BIAs were updated from the original BIAs published by Baird et al. (2015d) based on additional analysis of photo-identification, satellite tracking, and genetic studies. The parent BIA encompasses 94,217 km^2 and the child BIA encompasses 7,775 km^2, the latter representing the core high-use areas in the Main Hawaiian Islands (Kratofil et al., 2023). In addition, a year-round non-hierarchical Small and Resident Population BIA was delineated for the Northwestern Hawaiian Islands and encompasses 138,001 km^2 (Kratofil et al., 2023) (Figure C-14).

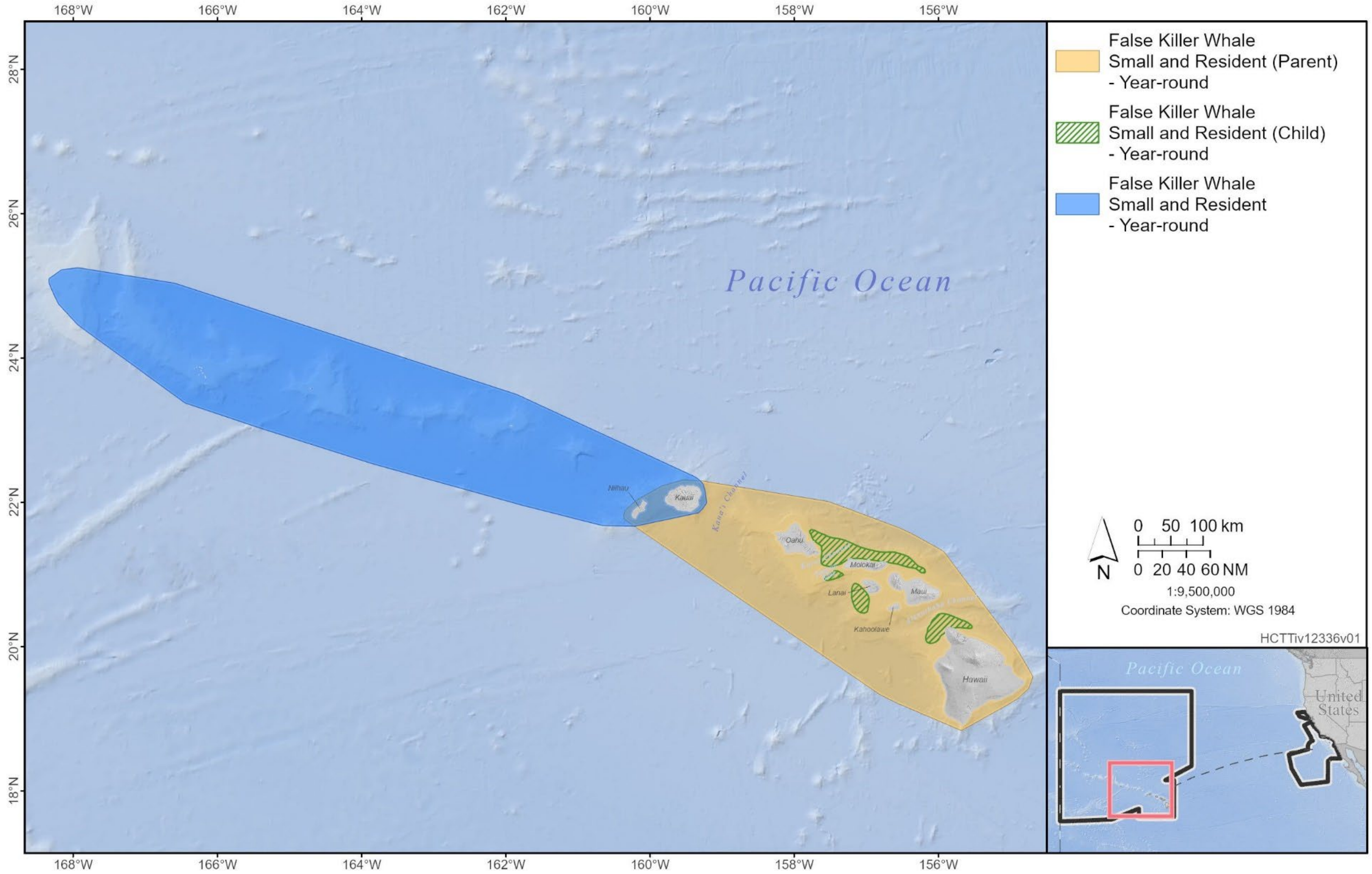


Figure C-14: False Killer Whale Small and Resident BIAs in the Hawaii Study Area

As noted previously, false killer whales are not usually expected to be present in the California Study Area. Older records document only a handful of sightings from areas such as Monterey Bay, Santa Catalina, and the Channel Islands (Baird, 2009a; Jefferson et al., 2008; Miller & Scheffer, 1986). False killer whales were not detected during the 15 aerial surveys conducted in the Southern California Range Complex from 2008 through 2012 (Smultea & Jefferson, 2014). A nearshore marine mammal survey off San Diego Bay in March 2014 detected a false killer whale pod that was assumed to be the same pod that had been seen 6 days before off Dana Point (Graham & Saunders, 2015). Two years later in April–March 2016, a whale watch vessel out of Dana Point again sighted a pod of false killer whales in the same area (Ritchie, 2016). This species normally prefers warmer tropical waters found outside of southern California and the presence of this species to the north of its usual habitat was likely due to the warmer than normal water temperatures associated with a known El Niño event. Such documented strandings and sightings of false killer whales in California waters have thus been considered extralimital. False killer whales do occur in warmer waters off the Baja California Peninsula, Mexico within the California Study Area (Hamilton et al., 2009).

C.6.2.7.3 Population Trends

Reeves et al. (2009) suggested that the Main Hawaiian Islands Insular stock of false killer whales declined between 1989 and 2009. A review of sighting rates of false killer whales from aerial surveys conducted around the main Hawaiian Islands between 1994 and 2004 also indicated a decline in this population (Baird, 2009b). A Status Review of the Main Hawaiian Islands Insular stock of false killer whales was consistent with these earlier studies, indicating that the population had declined at a rate of 9 percent per year since 1989, although alternative models showed a lower rate of decline (Oleson et al., 2010b). (Baird, 2009a; Baird et al., 2015f; Bradford & Forney, 2016, 2017; Carretta et al., 2017b; Carretta et al., 2023b; Cascadia Research Collective, 2010; Oleson et al., 2010b; Reeves et al., 2009; West, 2016)Population Threats

(Cascadia Research Collective, 2010) In Hawaiian waters, false killer whales are particularly susceptible to fishery interactions and entanglements (Baird et al., 2015f; Bradford & Forney, 2016). A historic decline in the Main Hawaiian Islands insular population has been the result of various non-Navy factors that include the small population size of this stock and incidental take by commercial fisheries (Bradford & Forney, 2016; Oleson et al., 2010b; Reeves et al., 2009). There were four strandings of Main Hawaiian Island Insular false killer whales in the Hawaiian Islands in the 7-year period between the start of 2010 and the end of 2016 (West, 2016). Two of these stranded animals had fishing gear (fishhooks, leaders, line) found within the stomach contents examined during necropsy (West, 2016). Most recent data from 2017 through 2021 indicates that there were no entanglements or hooks of false killer whales in the shallow-set longline fishery; however, there were 54 reported entanglements or hooks of this species in the deep-set longline fishery during the same time period (Carretta et al., 2023b). Of the deep-set longline fishery interactions, 18 were encountered in the outside of the Hawaii EEZ, where 14 were considered seriously injured and 2 were found dead. Of the remaining 36 encounters in the Hawaii EEZ, 23 whales were considered seriously injured and 5 were found dead (Bradford, 2018a; Bradford, 2018b; Bradford, 2021; Bradford et al., 2017; Bradford et al., 2021; Carretta et al., 2023b; McCracken, 2019).

Because false killer whales feed on large prey at the top of the food chain (e.g., squid, tunas) they may also be impacted by competition with fisheries (Cascadia Research Collective, 2010).

Like many marine mammals, false killer whales also accumulate high levels of toxins in their blubber over the course of their long lives, but the consequence of that bioaccumulation remains unknown.

Climate change has increasingly become a threat to marine mammals. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.2.8 Killer Whale (*Orcinus orca ater*; Eastern North Pacific Southern Resident stock)

Different geographic forms of killer whale are distinguished by distinct social and foraging behaviors and other ecological traits. In the North Pacific, these recognizable geographic forms are variously known as ‘‘residents,’’ ‘‘transients,” and “offshore” ecotypes (Hoelzel et al., 2007; Morin et al., 2024). Recently, new analyses suggest that species status may be warranted for two of the different eastern North Pacific ecotypes, the Bigg’s or transient killer whales and the resident killer whales, including the Eastern North Pacific Southern Resident stock (Morin et al., 2024). However, the proposal before the Society for Marine Mammalogy Committee on Taxonomy failed to receive the 2/3 majority needed to pass, so killer whales will remain as a single species, but Bigg’s and resident killer whales will now be recognized as separate subspecies. *Orcinus orca rectipinnus* and *Orcinus orca ater*, respectively. In the HCTT Study Area, both the transient and offshore ecotypes are known to occur, and Southern Resident killer whales may seasonally occur along the northern portion of the California Study area north of Monterey Bay (Hanson et al., 2018);National Marine Fisheries Service, 2021 #15025;Millman, 2019 #3115}.

C.6.2.8.1 Status and Management

Eight killer whale stocks are recognized within the North Pacific U.S. EEZ, with only the Hawaiian stock occurring in Hawaii and three stocks occurring in the California Study Area consisting of the West Coast Transient stock, the Eastern North Pacific Offshore stock, and the Eastern North Pacific Southern Resident stock (Carretta et al., 2023b). Three separate pods comprise the Southern Resident stock, identified as the J, K, and L pods (Ford et al., 2000). Killer whales are protected under the MMPA and the Southern Resident killer whale stock or DPS is listed as endangered under the ESA listed. The other two stocks are not listed under the ESA.

Critical Habitat

NMFS designated critical habitat for the Southern Resident killer whale DPS on November 29, 2006, in the inland waters of Washington State (71 FR 69054) and revised the designation to expand critical habitat into offshore waters of Washington, Oregon, and California on August 2, 2021 (86 FR 41668). The critical habitat along the West Coast that extends into the Study Area is defined as marine waters between the 6.1 and 200 m depth contours (86 FR 41668). As shown in Figure C-15, critical habitat occurs within a small, nearshore portion of the northern portion of the California Study Area and shoreward of the PMSR to just south of Monterey.

The essential features defining critical habitat for the Southern Resident killer whale are: (1) Water quality to support growth and development; (2) Prey species of sufficient quantity, quality, and availability to support individual growth, reproduction, and development, as well as overall population growth; and (3) Passage conditions to allow for migration, resting, and foraging. Calambokidis et al. (2024) defined a small and resident BIA (parent and core) off the U.S. West Coast extending from Washington State south to Point Sur, California. Only the parent BIA is located within or adjacent to the Study Area (Figure C-16); the core BIA is north of the Study Area off the coast of Washington State. The BIA is the same spatial extend as the designated critical habitat for Southern Resident killer whales.

Figure C-15: Critical Habitat for Southern Resident Killer Whale in the California Study Area

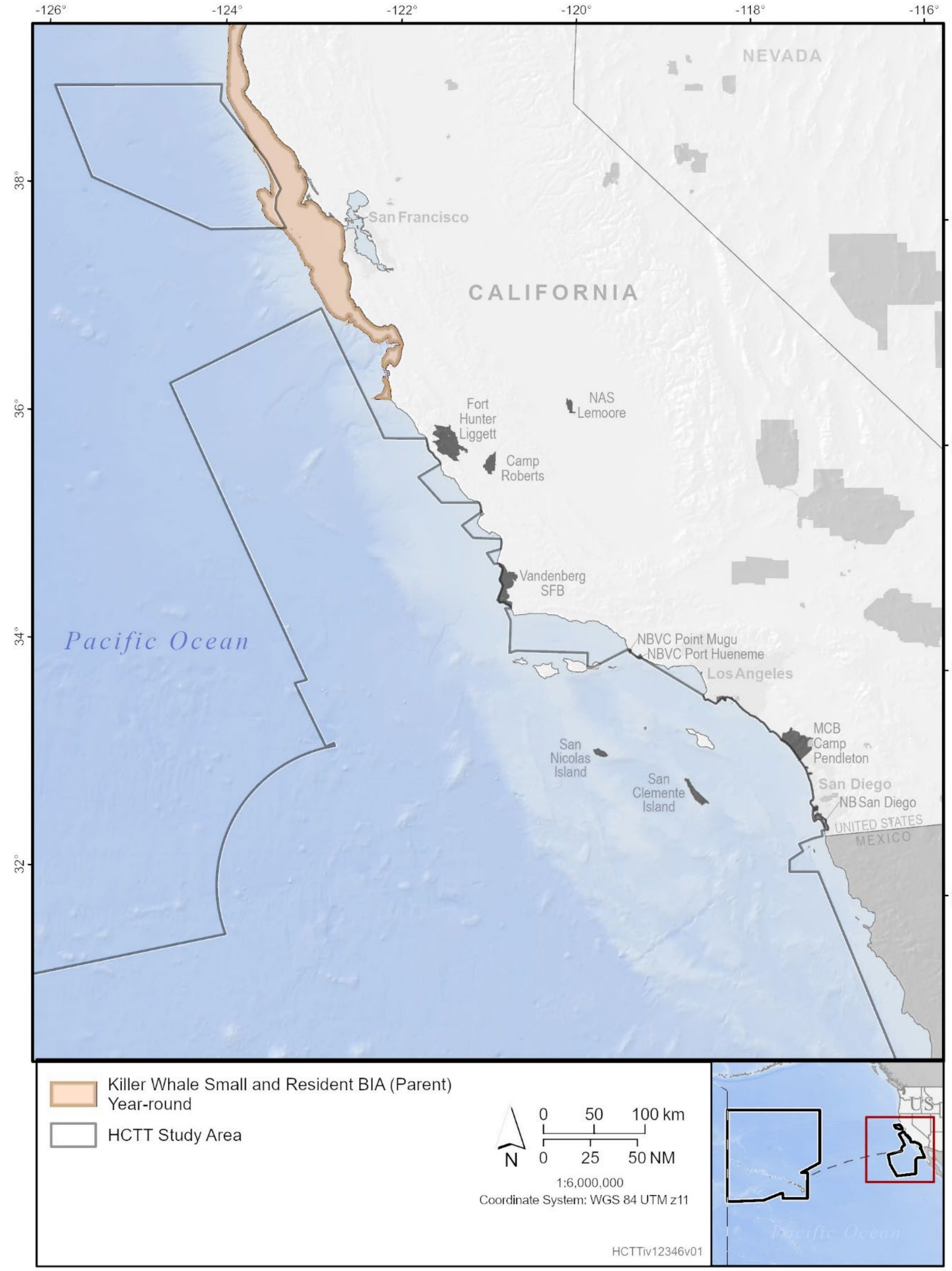


Figure C-16: Southern Resident Killer Whale Small and Resident BIA in the California Study Area

Recovery Goals

In response to the threats facing the species, NMFS developed goals to recover the Southern Resident killer whale. The 2008 Southern Resident Killer Whale Recovery Plan (National Marine Fisheries Service, 2008) provides the complete downlisting/delisting criteria. The recovery action outline includes the following management measures and research and monitoring actions:

- Protect the Southern Resident killer whale population from factors that may be contributing to its decline or reducing its ability to recover.
- Protect Southern Resident killer whales from additional threats that may cause disturbance, injury, or mortality, or impact habitat.
- Develop public information and education programs.
- Respond to killer whales that are stranded, sick, injured, isolated, pose a threat to the public, or exhibit nuisance behaviors.
- Transboundary and interagency coordination and cooperation.
- Monitor status and trends of the Southern Resident killer whale population.
- Conduct research to facilitate and enhance recovery efforts for Southern Resident killer whales.

C.6.2.8.2 Habitat and Geographic Range

Killer whales are found in all marine habitats from the coastal zone (including most bays and inshore channels) to deep oceanic basins and from equatorial regions to the polar pack ice zones of both hemispheres. Although killer whales are also found in tropical waters and the open ocean, they are most numerous in coastal waters and at higher latitudes (Dahlheim & Heyning, 1999). Forney and Wade (2006) found that killer whale densities increased by 1–2 orders of magnitude from the tropics to the poles.

All three ecotypes of killer whale are known to occur along the west coast of North America, from the entire Alaskan coast, in British Columbia and Washington inland waterways, and along the outer coasts of Washington, Oregon, and California, but the endangered resident ecotype’s range typically does not extend south of Monterey California (Calambokidis & Barlow, 2004; Carretta et al., 2017b; Dahlheim et al., 2008; Ford & Ellis, 1999; Forney et al., 1995; Hanson et al., 2018).

Southern Resident killer whales have seasonal shifts in distribution from the inland waters of the Salish Sea and Puget Sound in Washington and British Columbia to locations that can be up to hundreds of miles from Washington waters both north (as far as Southeast Alaska) or south as far as central California (Carretta et al., 2023b; National Marine Fisheries Service, 2021c). Of the three pods, the K and L pods appear to have a more extensive and seasonally variable offshore coastal distribution, with rare sightings as far south as Monterey Bay and central California in recent years; and the L pod has been observed in in Chatham Strait, Southeast Alaska (Hanson et al., 2017; National Marine Fisheries Service, 2021c).

To better predict the pattern of distribution of the endangered Southern Resident killer whales off the Washington, Oregon, and Northern California coasts, researchers integrated visual sightings, location data obtained between 2012 and 2016 from satellite-tagged Southern Resident killer whales, and acoustic detections from underwater hydrophones obtained from 6 to 13 recorders deployed from 2011 to 2015 off the Washington, Oregon, and California coast (Hanson et al., 2018; U.S. Department of the Navy, 2018). Along the Pacific coast, the distribution of satellite-tag locations confirms that Southern

Resident killer whales generally inhabit nearshore waters over the continental shelf in waters less than 200 m deep and typically within 34 km from shore (Hanson et al., 2017). Over multiple years the data revealed that the killer whales spent the greatest amount of time near the mouth of the Columbia River and off Westport, Washington, north of the California Study Area (Hanson et al., 2018; Hanson et al., 2017; U.S. Department of the Navy, 2018). Based on the Hanson et al. (2018) analyses, members of the K and L pods may occur within the northern coastal portion of the HCTT study area from January to May.

Their seasonal range and preference for nearshore habitat reduces the likelihood that Southern Resident killer whales would occur in the Study Area, and, furthermore, that their occurrence would be limited to the easternmost portion of the northern portion of the California Study Area and inshore of the PMSR in winter and early spring.

C.6.2.8.3 Population Trends

Based on the Draft 2023 U.S. Pacific Marine Mammal Stock Assessment Report (Carretta et al., 2024), the most recent estimate of the total population of Southern Resident killer whales based on data collected from 1 July 2021 through 1 July 2022 was 73 whales. In 1995, abundance of the stock peaked at 99 and has declined an average of 1 percent per year (Carretta et al., 2023b). Further declines are predicted if the population remains genetically isolated (Kardos et al., In Press).

C.6.2.8.4 Population Threats

The reduced availability of salmon, specifically Chinook salmon, their preferred prey, is considered one of the main threats to the southern resident killer whales (Couture et al., 2022). Many salmon populations that were once abundant historically have declined to the point where they have been listed as endangered or threatened with extinction (see Section 3.6, Fishes). The reduced availability of salmon has been linked to overfishing, poor artificial propagation practices, and degradation of freshwater and estuarine habitats through urbanization, dam building, and forestry, agricultural, and mining practices (Hilborn et al., 2012; National Marine Fisheries Service, 2008, 2016t; National Oceanic and Atmospheric Administration Fisheries, 2014; Office of the Washington Governor, 2018).

Since the 1970s commercial shipping, whale watching, ferry operations, and recreational boat traffic in Puget Sound and the coastal islands of southern British Columbia when southern residents occur most frequently has increased (Bassett et al., 2012; Bassett et al., 2010; Erbe, 2002; Holt et al., 2017; Holt et al., 2011; Williams et al., 2014b; Williams et al., 2019). In addition to the disturbance associated with the presence of vessels, the noise associated with vessel traffic affects the acoustic ecology of Southern Resident killer whales, which would affect their social ecology. Foote et al. (2004) compared recordings of Southern Resident killer whales that were made in the presence or absence of boat noise in Puget Sound during three time periods between 1977 and 2003. The authors determined that the duration of primary calls in the presence of boats had increased by about 15 percent during the last of the three time periods (2001–2003). Holt et al. (2008) reported that Southern Resident killer whales in Haro Strait off the San Juan Islands in Puget Sound increased the amplitude of their social calls in the face of increased levels of background noise. Although the costs of these vocal adjustments remains unknown, Foote et al. (2004) suggested that the amount of boat noise may have reached a threshold above which the killer whales need to increase the duration of their vocalization to overcome masking effects.

Exposure to contaminants may also harm killer whales. The presence of high levels of persistent organic pollutants, such as PCB, DDT, and flame-retardants, has been documented in southern resident killer whales (Krahn et al., 2007). Although the consequences of these pollutants on the fitness of individual killer whales and the population itself remain unknown, in other species these pollutants have been reported to suppress immune responses (Wright et al., 2007), impair reproduction, and exacerbate the energetic consequences of physiological stress responses when they interact with other compounds in an animal's tissues (Wright et al., 2007). Because of their long-life span, position at the top of the food chain, and their blubber stores, killer whales would be capable of accumulating high concentrations of contaminants.

Although Southern Resident killer whales are also susceptible to interactions with fisheries, there has only been one recorded entanglement of this species in 1977. Most recent data indicate that there were no recorded entanglements of Southern Resident killer whales in California gillnet fisheries from 2017 through 2021(Carretta et al., 2023a).

Climate change has increasingly become a threat to marine mammals, including killer whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.2.9 Hawaiian Monk Seal (*Neomonachus schauinslandi*)

C.6.2.9.1 Status and Management

The Hawaiian monk seal was listed as endangered under the ESA in 1976 (National Marine Fisheries Service, 1976) and is listed as depleted under the MMPA. The species is considered a high priority for recovery, based on the high magnitude of threats, the high recovery potential, and the potential for economic conflicts while implementing recovery actions (National Marine Fisheries Service, 2007b, 2011b, 2016p). The approximate area encompassed by the northwestern Hawaiian Islands was designated as the Papahanaumokuakea National Marine Monument in 2006, in part to protect the habitat of the Hawaiian monk seal. Hawaiian monk seals are managed as a single stock. There are six main reproductive subpopulations at: French Frigate Shoals, Laysan Island, Lisianski Island, Pearl and Hermes Reef, Midway Island, and Kure Atoll in the northwestern Hawaiian Islands.

Critical Habitat

Critical habitat for Hawaiian monk seals was designated August 21, 2015 (National Oceanic and Atmospheric Administration, 2015a) (Figure C-17). The essential features of the critical habitat were identified as: (1) adjacent terrestrial and aquatic areas with characteristics preferred by monk seals for pupping and nursing; (2) shallow, sheltered aquatic areas adjacent to coastal locations preferred by monk seals for pupping and nursing; (3) marine areas from 0 to 500 m in depth preferred by juvenile and adult monk seals for foraging; (4) areas with low levels of anthropogenic disturbance; (5) marine areas with adequate prey quantity and quality; and (6) significant areas used by monk seals for hauling out, resting, or molting (National Oceanic and Atmospheric Administration, 2015a).

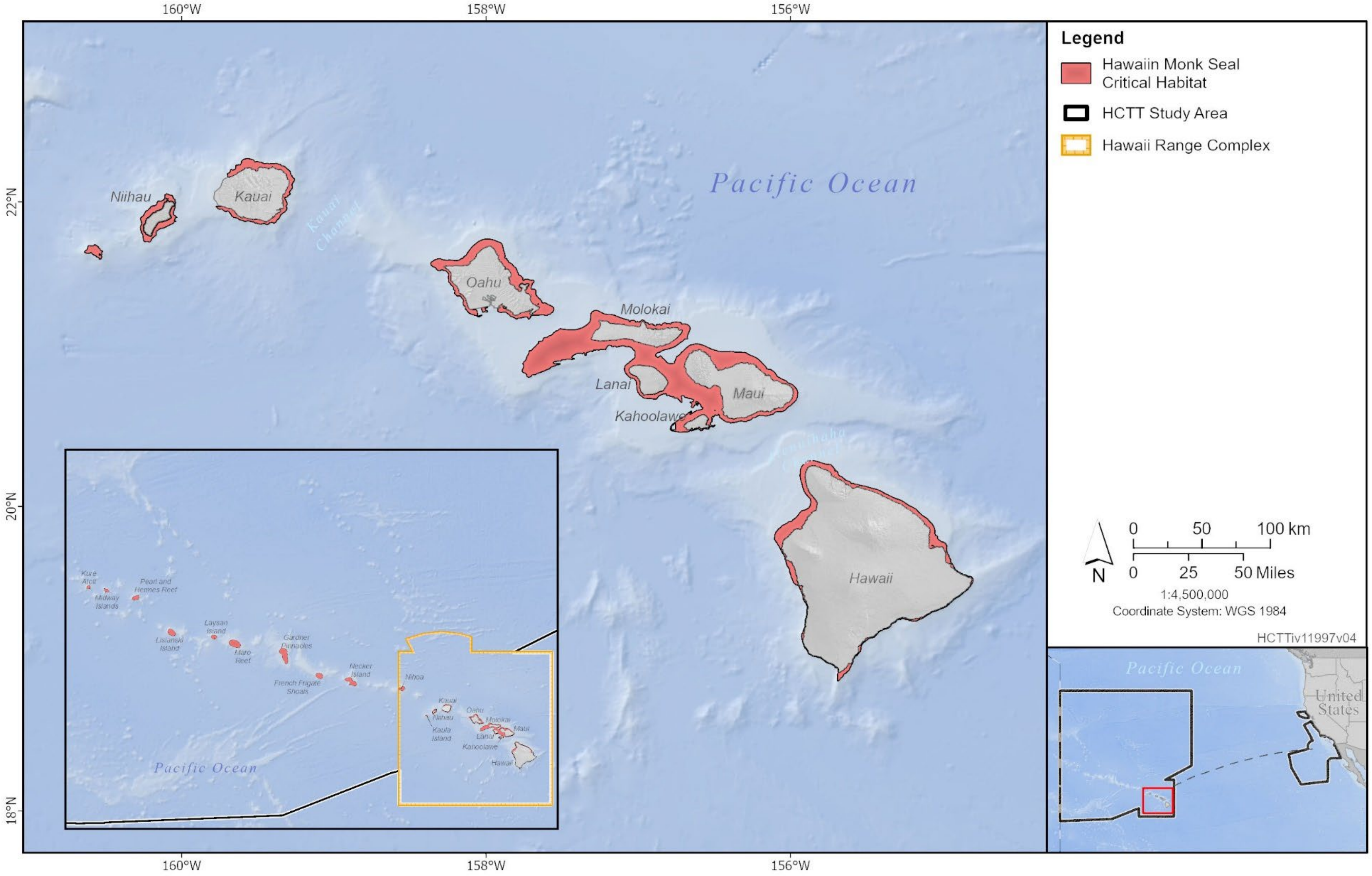


Figure C-17: Critical Habitat for Hawaiian Monk Seal in the HCTT Study Area

Recovery Goals

A recovery plan for the Hawaiian monk seal was completed in 1983 and was revised in 2007 (National Marine Fisheries Service, 2007b, 2011b, 2016p). The 2007 Hawaiian Monk Seal Recovery Plan for the complete downlisting/delisting criteria. The recovery action outline includes the following fourteen short-term and long-term actions:

- Investigate and mitigate factors affecting food limitation.
- Prevent entanglements of monk seals.
- Reduce shark predation on monk seals.
- Minimize exposure and spread of infectious disease.
- Conserve Hawaiian monk seal habitat.
- Reduce Hawaiian monk seal interactions with fisheries.
- Reduce male aggression toward pups/immature seals and adult females.
- Reduce the likelihood and impact of human disturbance.
- Investigate and develop response to biotoxin impacts.
- Reduce impacts from compromised and grounded vessels.
- Reduce the impact of contaminants.
- Continue population monitoring and research.
- Create a Main Hawaiian Islands Hawaiian Monk Seal Management Plan.
- Implement the Recovery Program for the Hawaiian monk seal.

C.6.2.9.2 Habitat and Geographic Range

Hawaiian monk seals occur in the main Hawaiian Islands and Northwestern Hawaiian Islands, but sightings have been reported at Johnston Atoll, Wake Island, and Palmyra Atoll (south of the Hawaiian Island chain; (Carretta et al., 2010; Gilmartin & Forcada, 2009; Jefferson et al., 2015; National Marine Fisheries Service, 2009a, 2010d). The six main breeding sites are in the northwestern Hawaiian Islands: Kure Atoll, Midway Islands, Pearl and Hermes Reef, Lisianski Island, Laysan Island, and French Frigate Shoals. Smaller breeding sites are on Necker Island and Nihoa Island, and monk seals have been observed at Gardner Pinnacles and Maro Reef.

There is a small breeding population of monk seals found throughout the main Hawaiian Islands. Births have been documented on most major islands and most commonly on Kauai and Niihau (Gilmartin & Forcada, 2009; National Marine Fisheries Service, 2007b, 2010d). Hawaiian monk seals give birth throughout the year, but peak pupping season is typically spring through summer. Monk seals prefer sandy, protected beaches for pupping (National Marine Fisheries Service, 2024g). Based on one study, on average, 10 to 15 percent of the monk seals migrate among the northwestern Hawaiian Islands and the main Hawaiian Islands (Carretta et al., 2010). Another source suggests that approximately 35 percent of the main Hawaiian Island seals travel between islands throughout the year (Littnan, 2011). Greater than expected movement between sites within the main Hawaiian Islands and the northwestern Hawaiian Islands (Johanos et al., 2014), has allowed for genetic conductivity between Hawaiian monk seal subpopulations (Schultz et al., 2011).

When hauled out, Hawaiian monk seals seem to prefer beaches of sand, coral rubble, and rocky terraces (Baker et al., 2006; Jefferson et al., 2015). Consistent with ten previous detections of monk seals at Kaula Island, in 2012 there were three individual monk seals that were observed hauled out on the rock ledge on the northwest side of the island (Richie et al., 2012). Aerial surveys of Kaula Island from April

2013 through March 2016 continued to document monk seals routinely hauled out on the rocky ledges at the edge of the island, numbering between five and 11 monk seals seen on each of the six surveys (Normandeau Associates & APEM, 2013a, 2013b, 2014, 2015a, 2015b, 2016).

In the Main Hawaiian Islands, monk seals are generally solitary and have no established rookeries unlike pinnipeds in Southern California. Hawaiian monk seals do, however, routinely haul out for molting and pupping in locations including at the Navy's PMRF, Pearl Harbor, and other military lands. When foraging, monk seals spend most of their time in nearshore, shallow marine habitats, but can rapidly cover large areas in search of food and may travel hundreds of miles in a few days (D'Amico, 2013; Littnan, 2011; Stewart et al., 2006; Wilson et al., 2012).

From 1996 to 2002 and in an effort to better understand the range of foraging monk seals, Stewart et al. (2006) used satellite-linked radio transmitters to document the movements of 147 Hawaiian monk seals from all six northwestern Hawaiian Islands breeding colonies. Foraging patterns were complex and varied among colonies by season, age and sex, but in general monk seals were found to forage extensively within the atoll barrier reefs and on the leeward slopes of reefs and islands at all colony sites. They also ranged away from these sites along the Hawaiian Islands submarine ridge to most nearby seamounts and submerged reefs and banks (Stewart et al., 2006).

Between February 2010 and July 2011, 12 data tags on monk seals in the main Hawaiian Islands were successfully deployed, retrieved, and analyzed (D'Amico, 2013; Littnan, 2011; Stewart et al., 2006; Wilson et al., 2012). The average foraging trip was approximately 30 km in distance, almost 19 hours in duration, and most seals remained within the 600 m depth contour. Although most trips were less than 50 km, two seals made at least one long pelagic foraging trip during the deployment period (Littnan, 2011). An adult male tagged on Oahu traveled over 3,000 km on a trip which lasted 36 days and a sub-adult female tagged on Kauai traveled 300 km on a trip that lasted almost 4 days. Approximately 54 percent of the seals made regular trips between two or more of the islands, while the remainder showed fidelity to one island (Littnan, 2011).

Hawaiian monk seals are not present in the California Study Area.

C.6.2.9.3 Population Trends

Population dynamics at the different locations in the northwestern Hawaiian Islands and the main Hawaiian Islands have varied considerably (Antonelis et al., 2006).

The smaller subpopulation in the Main Hawaiian Islands has been increasing in recent years, whereas the larger population in the northwestern Hawaiian Island was thought to have been in a long-term decline (Antonelis et al., 2006; Baker et al., 2016c; Baker et al., 2011; Baker & Johanos, 2004). However, a new approach was developed to estimate the abundance range-wide and for individual island-specific subpopulations (Baker et al., 2016c). The new approach incorporates multiple methods of estimating site-specific abundances (e.g., direct counts, counts corrected for seals at sea, capture-recapture) and combines the results into a model (Harting et al., 2017). The Monte Carlo-style model was employed to overcome inconsistent field survey data, which, due to the difficulty of surveying numerous remote islands simultaneously, are collected years apart and often using differing, non-standardized methods. Based on the most recent counts and modeling results, the range-wide abundance is estimated at 1,437 monk seals (Carretta et al., 2022). Of particular importance is that the model also indicted the monk seal population increased at a rate of 2 percent per year from 2013 to 2019, countering previous trend analyses indicating the population was in decline (Carretta et al., 2022; Robinson et al., 2022).

C.6.2.9.4 Population Threats

Monk seals are susceptible to fishery and marine debris entanglements at rates higher than most other pinnipeds (Carretta et al., 2023b; Henderson, 2001). Records collected in the main Hawaiian Islands show at least 140 seal hooking and entanglement incidents from 1976 to 2014 (National Marine Fisheries Service, 2016p). In 2021, 29 Hawaiian monk seal hookings were observed, with two hookings resulting in serious injury (Carretta et al., 2024; Mercer, 2023). Monk seals have also been observed entangled in nearshore fishing gillnets, which has resulted in injury and mortality.

There have been several hundred documented cases of marine debris entanglement of monk seals, 10 of which have resulted in mortality (Carretta et al., 2023b; Henderson, 2001; Henderson, 2019; Mercer, 2021). In the northwestern Hawaiian Islands, derelict fishing gear has been identified as a top threat to the monk seal (Donohue & Foley, 2007), while in the main Hawaiian Islands, high risks are associated with health hazards from exposure to pollutants and infectious disease agents associated with terrestrial animals. The Hawaiian Monk Seal Research Program reported that in Hawaii since 2001, there have been at least 8 deaths of Hawaiian monk seals attributed to parasitic toxoplasmosis from feral cats in the main Hawaiian Islands (Hawaiian Monk Seal Research Program, 2015; Rogers, 2016). In 2015, the Hawaiian Monk Seal Research Program began a vaccination program to protect Hawaiian monk seals from morbillivirus because of the threat it poses given that monk seals do not otherwise carry antibodies to the virus (National Oceanic and Atmospheric Administration, 2015d).

Natural population threats to Hawaiian monk seals include disease, parasites, reduced prey availability, and predator attacks, particularly by both killer whales and sharks. Shark predation is one of the major sources of mortality for this species especially in the northwestern Hawaiian Islands. Galapagos sharks are a major source of juvenile mortality in the northwestern Hawaiian Islands, with most predation occurring in the French Frigate Shoals (Antonelis et al., 2006; Gilmartin & Forcada, 2009). Another species-specific threat includes aggressive male monk seals that have been documented to injure and sometimes kill females and pups (National Marine Fisheries Service, 2010e, 2010f).

Climate change has increasingly become a threat to marine mammals, including pinnipeds. Since monk seals rely on coastal habitats for survival, monk seals may be affected by future sea level rise and loss of habitat as predicted by global climate models. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.2.10 Guadalupe Fur Seal (A*rctocephalus townsendi*)

C.6.2.10.1 Status and Management

The Guadalupe fur seal is listed as threatened under the ESA and depleted under the MMPA. Critical habitat for the Guadalupe fur seal has not been designated, and the only areas likely to meet the requirements as critical habitat are outside of U.S. jurisdiction (National Oceanic and Atmospheric Administration, 1985). All fur seals alive today are recent descendants from one breeding colony at Isla Guadalupe and Isla San Benito off Mexico and are considered a single stock (Carretta et al., 2017b; Pablo-Rodriguez et al., 2015).

C.6.2.10.2 Habitat and Geographic Range

Guadalupe fur seals are not found in the Hawaii Study Area.

On shore, Guadalupe fur seals are typically found hauled out along coastlines with abundant large rocks, often at the base of large cliffs. They are also known to inhabit caves, which provide protection and cooler temperatures, especially during the warm summer breeding season (Belcher & Lee, 2002). Adult males, juveniles, and nonbreeding females may remain at sea for much of the year (Reeves et al., 1992). Several observations suggest that this species travels alone or in small groups of fewer than five individuals (Belcher & Lee, 2002; Seagars, 1984).

Navy funded tagging studies tracking Guadalupe fur seal movements from Guadalupe Island north along the U.S. West Coast show that non-pups (adults and juveniles of both sexes) occur in highest concentrations in offshore waters near the Patten Escarpment or at approximately the 2,000 m depth contour and in the SOCAL Range Complex and PMSR (Norris, 2019; Norris, 2022; Norris & Elorriaga-Verplancken, 2020). Pups, however, migrate closer to shore than non-pups and are known to migrate farther north into waters off Oregon, Washington, and British Columbia. Based on the tagging results and unpublished data, a "core range" and a broader "geographic range" representing Guadalupe fur seal distribution was defined along the West Coast (Norris, 2022). Detailed information on the distribution of this species in the Study Area is provided in the *U.S. Navy Marine Species Density Database Phase IV for the Hawaii-California Training and Testing Study Area* technical report (U.S. Department of the Navy, 2024).

C.6.2.10.3 Population Trends

Guadalupe fur seals were once plentiful off the coasts of California and Mexico, ranging from the Gulf of the Farallones near San Francisco, to the Revillagigedo Islands, Mexico (Aurioles-Gamboa et al., 1999). However, over-harvesting in the 19th century led them to the brink of extinction. With implementation of protective measures in the 20th century by both the U.S. and Mexico, the population began to slowly recover and expand into its historical range extending from central Mexico to waters off Washington State (Aurioles-Gamboa et al., 2010; D'Agnese et al., 2020; Melin & DeLong, 1999; Norris & Elorriaga-Verplancken, 2020; Stewart, 1981; Stewart et al., 1993b). An unpublished abundance of 43,360 Guadalupe fur seals based on pup counts was provided by Norris (2022) as the mean of two separately derived abundance estimates of 37,940 and 48,780 fur seals. Current and ongoing studies indicate that the population is continuing to increase.

C.6.2.10.4 Population Threats

Most recent data indicates that there was a total of 13 reported serious injuries and/or deaths of Guadalupe fur seals along the U.S. west coast from 2013 to 2017 due to human-related causes (primarily marine debris entanglement)(Carretta et al., 2023b). In 2015 an UME was declared for Guadalupe fur seal. 80 strandings of Guadalupe fur seals, which were approximately eight times higher than the historical average, occurred along the entire coast of California, consisted of mostly weaned pups and juveniles in the one-to-two-year age-range, and included animals in distress but alive as well as dead individuals. Findings from the majority of these stranded Guadalupe fur seals were that they were malnourished and had secondary bacterial and parasitic infections (National Marine Fisheries Service, 2015a). It is likely that a shift in the prey may have resulted in these young animals being unable to obtain adequate food due to anomalously persistent warm ocean conditions (Bond et al., 2015).

Climate change has increasingly become a threat to marine mammals, including baleen whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.2.11 Southern Sea Otter (*Enhydra lutris neris*)

C.6.2.11.1 Status and Management

The southern sea otter is listed as threatened under the ESA and depleted under the MMPA. The southern sea otter is managed by the USFWS, and critical habitat has not been designated for this species. Southern sea otters are not found in the Hawaii Study Area. Between 1987 and 1990, the USFWS managed a program to translocate southern sea otters from the mainland to SNI to ensure the population would survive a catastrophic event, such as an oil spill, off the mainland coast (U.S. Fish and Wildlife Service, 2012a). The program would also help to reestablish a population on an island where they historically occurred and promote the recovery of the species (Bodkin, 2015). A total of 140 sea otters were moved to SNI, but by 1993, fewer than 15 survived. The USFWS declared the translocation program a failure in 2012 and ended the program, but despite that, the population has continued to grow (Yee et al., 2023).

The NDAA for Fiscal Year 2016 included provisions directing the Secretary of the Navy to establish Southern Sea Otter Military Readiness Areas at SNI and SCI. Within these Military Readiness Areas, the 2016 NDAA established that the ESA sections 4 and 9 and MMPA sections 101 and 102 do not apply to the incidental taking of any southern sea otter during military readiness activities. Any sea otter within the Military Readiness Areas is to be treated for the purposes of section 7 of the ESA as a member of a species that is proposed to be listed as endangered or threatened under the ESA.

As an additional component of the management of the species, the 2016 NDAA required that the Navy conducts monitoring and research within the Military Readiness Areas to determine the effects of military readiness activities on the growth or decline of the southern sea otter population and on the nearshore ecosystem. The monitoring and research were designed in consultation with the USFWS, and reports to Congress are required periodically. The first of these reports was completed in 2017 (U.S. Department of the Navy, 2017a; U.S. Department of the Navy et al., 2016). Subsequent follow-on reports have been provided every three years thereafter, and findings from these reports will continue to be reviewed by the Navy and USFWS to ensure the plan continues to adequately monitor interactions between military readiness activities and the sea otter population.

C.6.2.11.1.1 Special Status of San Nicholas Island population

Shortly after their discovery by 18th century European explorers, fur traders hunted sea otters across the north Pacific. By the turn of the 20th century, sea otters were nearly extinct. Southern sea otters are descendants of a small remnant colony that survived along the central California coast. In 1986, the USFWS sought to enhance recovery of the species through translocation of southern sea otters from the central coast of California to SNI. Inconsistencies between the ESA and MMPA led the USFWS to seek congressional authorization for the Southern Sea Otter Translocation Program, and President Reagan signed Public Law 99-625 into law on November 7, 1986. The law authorized the USFWS to establish an experimental population of sea otters and provided provisions for a sea otter management zone and specific exemptions from the ESA for Department of Defense activities. Between 1987 and 1990, the USFWS moved 140 sea otters to SNI. By 1993, fewer than 15 sea otters remained at the island. Many animals swam back to the central coast; some were captured and returned to the central coast after swimming into the designated sea otter management zone, and some died due to being moved (U.S. Fish and Wildlife Service, 2012a). The USFWS eventually declared the translocation program a failure and removed regulations governing the program (77 FR 75266, December 19, 2012).

Since declaring the translocation a failure, the sea otter colony has increased significantly with more than 100 animals routinely observed at SNI. At the start of the translocation program, the USFWS had committed to removing sea otters from SNI if the translocation program was ever declared a failure. They instead determined that it would be in the best interest of southern sea otter recovery to leave them at the island. The USFWS acknowledged that the Navy was given an exemption from ESA for sea otters on SNI but stated in their final rule (77 FR 75266) that there was no exemption from the MMPA for Navy activities. Clarification of Department of Defense ESA and MMPA responsibilities for sea otters in Southern California was provided by Congress in the National Defense Authorization Act for Fiscal Year 2016, which included provisions directing the Secretary of the Navy to establish Southern Sea Otter Military Readiness Areas at SNI and San Clemente Island (Figure 3.7 7). Within these Military Readiness Areas, the 2016 NDAA established that the ESA sections 4 and 9 and MMPA sections 101 and 102 do not apply to the incidental taking of any southern sea otter during Navy testing and training activities. Any sea otter within the Areas is to be treated for the purposes of section 7 of the ESA as a member of a species that is proposed to be listed as endangered or threatened under the ESA.

As an additional component of the management of the species, the 2016 NDAA requires that the Navy conduct monitoring and research within the Southern Sea Otter Military Readiness Areas (Figure C-18) to determine the effects of military readiness activities on the growth or decline of the southern sea otter population and on the nearshore ecosystem. The monitoring and research are designed in consultation with the USFWS, and reports to Congress are required periodically. The first of these reports was completed in 2017 (U.S. Department of the Navy, 2017a; U.S. Department of the Navy et al., 2016). Subsequent follow-on reports are to be provided every three years thereafter, and findings from these reports will continue to be reviewed by the Navy and USFWS to ensure the plan continues to adequately monitor interactions between military readiness activities and the sea otter population.

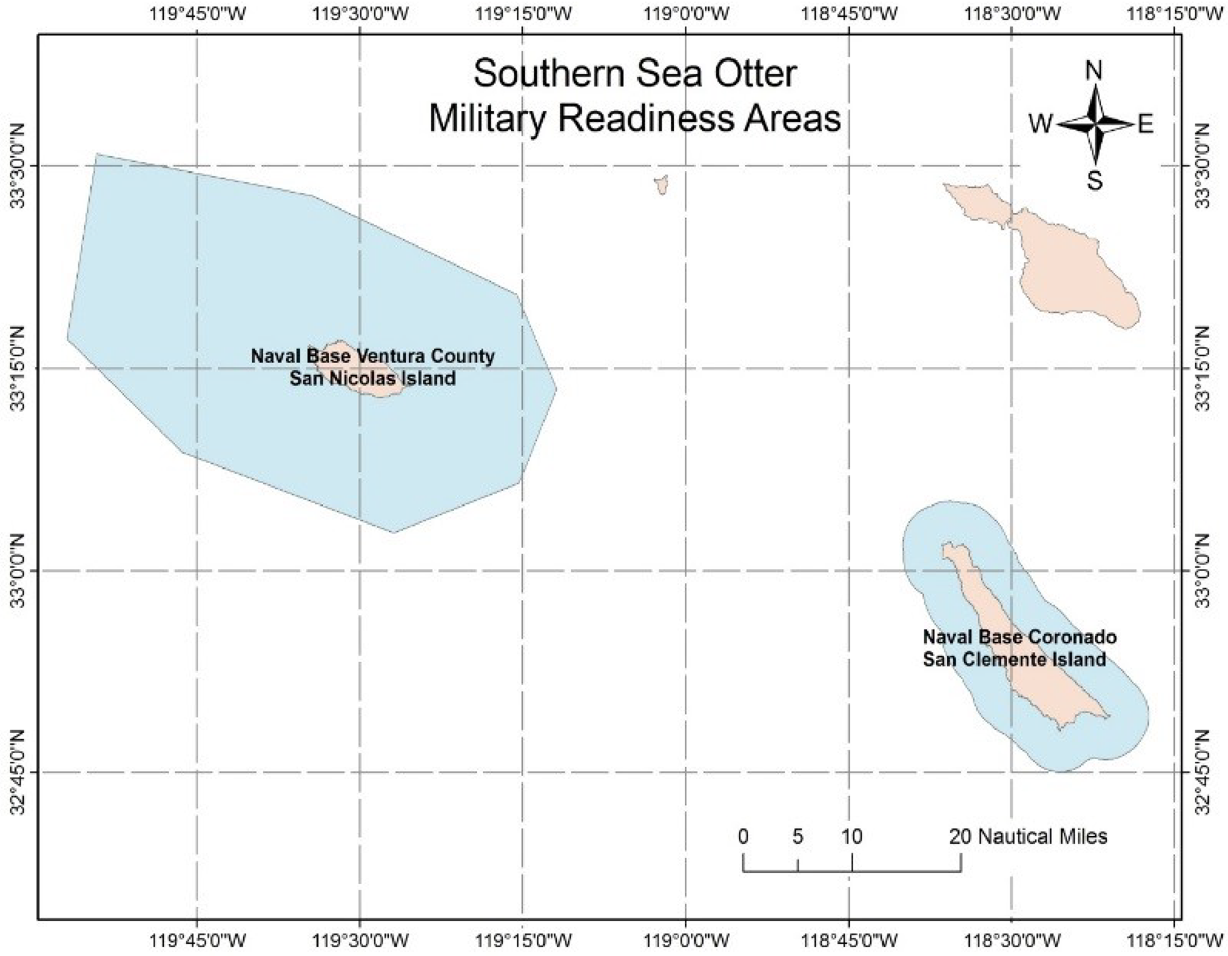


Figure C-18: Southern Sea Otter Military Readiness Areas as Established by the 2016 NDAA

C.6.2.11.2 Habitat and Geographic Range

The distribution of the southern sea otter is currently limited to nearshore waters off the coast of central California, ranging from Pigeon Point in the north to south of Point Conception, and off SNI (Hatfield et al., 2016; Hatfield et al., 2019; Tinker et al., 2017). Sea otter distribution is typically defined by habitat features, frequently water depth and distance from shore, which are linked to the otter's dive depth limitations or preferences when foraging (Bodkin et al., 2004; Thometz et al., 2016; Tinker et al., 2017; Tinker et al., 2021). The primary foraging depth range for the southern sea otter is between 2 and 35 m (Tinker et al., 2006), depths found only closer to shore in the Study Area. Surveys conducted in 2019 along the central California mainland extended from shore to the 60 m depth contour (Hatfield et al., 2019) and surveys off SNI from 2017 through 2020 extended out to the 30 m depth contour (Yee et al., 2020). Kelp canopy has also been shown to be a strong indicator of sea otter occurrence in California waters (Lafferty & Tinker, 2014; Yee et al., 2020).

Sea otters are occasionally be present in deeper waters when moving between areas or in attempts to establish new habitat (Burn & Doroff, 2005). Although uncommon, sea otters have been known to transit between SNI and the mainland coast, likely in search of new habitat (Hatfield, 2005).

The sea otter population off SNI is subject to different habitat conditions and stressors than those inhabiting the central California coastline (Tinker et al., 2007). Navy management and restricted access to the area has had a beneficial effect on the otters. The abundance of sea otter prey at San Nicolas exceeds that at the central California coastline by as much as three orders of magnitude (Tinker et al., 2007). As a result of greater prey availability, sea otters on SNI have double the average food intake rate, spend only half as much time foraging, and have better body conditions than sea otter occurring along the central California coastline (Tinker et al., 2007).

Detailed information on the distribution of this species in the Study Area is provided in the *U.S. Navy Marine Species Density Database Phase IV for the Hawaii-California Training and Testing Study Area* technical report (U.S. Department of the Navy, 2024).

C.6.2.11.3 Population Trends

U.S. Fish and Wildlife Service (2021b) reported an abundance of 99 sea otters in waters around SNI. From 2020 to 2023, Yee et al. (2023) measured a 10 percent annual increase in the population on SNI and estimated an abundance around the island of 146 otters, as of April 2023.

C.6.2.11.4 Population Threats

Sea otters are preyed upon by sharks, particularly white shark, which are likely the greatest threat to population growth off California (Carretta et al., 2022; Tinker et al., 2017; U.S. Fish and Wildlife Service, 2021b). Miller et al. (2020c) found that in Southern sea otters that were subjected to necropsy between 1998 and 2012 (n=560), the most prominent cause of death was infectious disease. The toxoplasmosis parasite (often attributed to feral cat feces in urban area storm run-off) impacts sea otters along the U.S. West Coast (Simeone et al., 2015), and the emergence of a nematode parasite in southern sea otters-associated with hepatitis has also been reported (Miller et al., 2020b). Heart disease associated with the inadvertent consumption domoic acid that accumulates in prey (specifically crabs and clams) has been revealed as a potential threat to population growth, because the risk was shown to be more pronounced in younger adults in their prime compared with older otters. Having a greater effect on adults in their prime reproductive years has long-term consequences for the recovery of the species, particularly with warmer ocean temperatures giving rise to more frequent and longer-lasting harmful algal blooms, which are linked to elevated concentrations of domoic acid in the marine environment (Moriarty et al., 2021; Wells et al., 2015).

C.6.3 Species Not Listed under the Endangered Species Act

C.6.3.1 Bryde's Whale (*Balaenoptera edeni*)

C.6.3.1.1 Status and Management

Bryde's whale is protected under the MMPA and is not listed under the ESA. NMFS recognizes two stocks of Bryde's whales in the U.S. Pacific, the Eastern Tropical Pacific stock (whales found east of 150° W, including the Gulf of California and waters off California) and the Hawaii stock (Carretta et al., 2023b). Bryde's whales in Hawaii or Southern California are considered to belong to their respective separate stocks, with the transition at 150° W longitude as defined by NMFS.

C.6.3.1.2 Habitat and Geographic Range

Bryde's whales occur primarily in offshore oceanic waters of the north Pacific (Barlow, 2006; Bradford et al., 2017), with a consistent lower density region near the Main Hawaiian Islands (Becker et al., 2021). They typically do not move poleward of 40° in either hemisphere and tend to occur primarily in tropical and subtropical zones, where water temperatures are relatively warm (Jefferson et al., 2015). Data suggest that winter and summer grounds partially overlap in the central north Pacific (Murase et al., 2015; Ohizumi, 2002; Ohizumi et al., 2002). Long migrations are not typical of Bryde's whales, although limited shifts in distribution toward and away from the equator, in winter and summer, have been observed (Best, 1996; Cummings, 1985).

A total of 48 Bryde's whale sightings were made during three systematic shipboard surveys of the Hawaiian Islands EEZ in 2002, 2010, and 2017, although the number of sightings varied substantially between years, with a low of 2 sightings in 2017 and a high of 32 sightings in 2010 (Bradford et al., 2021). Based on predictive habitat-based density models derived from these line-transect survey data, there was also great annual variability in the distribution patterns of Bryde's whales within the Hawaiian Islands EEZ (Becker et al., 2021). The substantial annual variability in both sighting numbers and distribution patterns likely reflects a fluctuating distribution of the whales relative to habitat or prey within a broader geographic region of the central North Pacific (Becker et al., 2022b). Based on line-transect survey data collected between 1997 and 2012 in the central North Pacific, relatively high densities of Bryde's whales are predicted within the U.S. EEZ of the Hawaiian Islands during the summer and fall (Forney et al., 2015). Given that the majority of survey data have been collected during summer and fall, less is known about the abundance and distribution of Bryde's whales within waters of the Hawaiian Islands EEZ in winter and spring. A sighting of a single Bryde's whale was made during a systematic survey of waters within the Main Hawaiian Islands in February of 2009 (Oleson et al., 2009), suggesting that this species is present during the winter months. Acoustic monitoring data collected using the Navy's instrumented training range hydrophones off the north coast of Kauai from August through October of 2014 also suggest that this species may be present year-round in Hawaii (Martin et al., 2017).

Bryde's whales were previously only occasionally sighted in the waters off SOCAL (Carretta et al., 2010; Smultea, 2012; Smultea et al., 2011), but sightings and acoustic monitoring indicates that the presence of the species is no longer considered anomalous (Carretta et al., 2017b; Debich et al., 2015b; Kerosky et al., 2012; Smultea et al., 2012; Smultea & Jefferson, 2014; Smultea et al., 2010). During aerial surveys conducted year-round between 2008 and 2013 off the Southern California coast, Bryde's whales were sighted on two occasions (Jefferson et al., 2014). These were the first sightings in this area since 1991 when a Bryde's whale was sighted within 300 NM of the California coast (Barlow, 1995). There was also an off-effort Bryde's whale sighting in waters off California during a 2014 systematic ship survey (Barlow, 2016). The peak in recorded Bryde's whale vocalizations has varied but generally occurs between late July and November in the southern portion of the California Study Area (Debich et al., 2015a; Debich et al., 2015b; Kerosky et al., 2012).

C.6.3.1.3 Population Trends

Little is known of population status and trends for most Bryde's whale populations, and there are no trend data for Bryde's whale abundance in the eastern tropical Pacific (Carretta et al., 2023b). Based on acoustic data there appeared to be in increase in the number of Bryde's whales within the Southern California Bight between 2000 and 2010 (Kerosky et al., 2012); however, during four systematic ship surveys of these waters in 2005, 2008, 2014, and 2018, there was only one verified off-effort Bryde's whale sighting in 2014 (Barlow, 2016).

C.6.3.1.4 Population Threats

Bryde's whales are susceptible to interactions with fisheries, entanglement in fishing gear, and vessel strikes. Off the U.S. West Coast, the California gillnet fishery is the only fishery likely to interact with this species in the region. However, monitoring data from 2001 through 2013 indicate that there have been no entanglements of Bryde's whales during this time period (Carretta et al., 2023b).

In Hawaii, Bryde's whales are most likely threatened by the deep-set longline fishery and the shallow-set longline fishery. The most recent monitoring data from 2017 through 2021 indicate that there were no Bryde's whales observed hooked or entangled in these fisheries (Bradford, 2018a; Bradford, 2018b; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019). Although there have been zero recorded entanglements, there is the potential for injury and mortality of Bryde's whales from fisheries interactions to go unobserved (Carretta et al., 2023b). Additionally, there was one documented vessel strike of an individual belonging to the Eastern Tropical Pacific Stock near Washington in 2010 (Carretta et al., 2023b).

Climate change has increasingly become a threat to marine mammals, including baleen whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.2 Humpback Whale (*Megaptera novaeangliae:* Hawaii Distinct Population Segment)

C.6.3.2.1 Status and Management

NMFS has identified 14 DPSs of humpback whales worldwide, with 4 DPSs occurring in the North Pacific (Carretta et al., 2023b). Humpback whales that occur seasonally in the HCTT Study Area are from three of the four DPSs identified by low-latitude wintering habitats: Hawaii DPS, Mexico DPS, and Central America DPS (Bettridge et al., 2015a; Carretta et al., 2023b; National Marine Fisheries Service, 2016q; Young, 2023). The three previously defined stocks of North Pacific humpback whales did not align with the DPS structure, so NMFS reevaluated the stock structure to incorporate both the locations of foraging and overwintering areas and population demographics. As a result, NMFS defined five stocks in the North Pacific:

- Central America/Southern Mexico-California-Oregon-Washington stock
- Mainland Mexico-California-Oregon-Washington stock
- Mexico-North Pacific stock
- Hawaii stock
- Western North Pacific stock

Humpback whales from the first four stocks listed above occur seasonally in the HCTT Study Area; humpbacks from the Western North Pacific stock do not occur in the Study Area. Humpback whales wintering in Hawaii are identified as the Hawaii DPS and comprise the Hawaii stock. Humpback whales from the Hawaii DPS/stock forage across the North Pacific (Figure C-10). Humpback whales from the Mexico DPS migrate to summer foraging habitat from California northward along the U.S. West Coast, Canada, Alaska, into the Bering Sea, and off the coast of Russia and are divided into the Mexico-North Pacific stock and the Mainland Mexico-California-Oregon-Washington stock. Humpback whales from the Central America DPS forage is waters off California and the Pacific Northwest and make up the Central America/Southern Mexico-California-Oregon-Washington stock (Carretta et al., 2023b). For additional information on the revised stock and DPS structure and population dynamics refer to Martien et al. (2023), Wade (2021), Martien et al. (2021) and Martien et al. (2019).

Humpback whales in the Hawaii DPS are not listed under the ESA, because the population is believed to have fully recovered to its pre-whaling abundance (Barlow et al., 2011; Bettridge et al., 2015a; Muto et al., 2017; National Marine Fisheries Service, 2016j; Wade et al., 2016). Humpback whales from the Mexico DPS are listed as threatened and those from the Central America DPS are listed as endangered under the ESA (National Marine Fisheries Service, 2016j).

C.6.3.2.2 Habitat and Geographic Range

Humpback whales are distributed worldwide in all major oceans and most seas (Bettridge et al., 2015a; National Marine Fisheries Service, 2016j). They typically are found during the summer in high-latitude feeding grounds, including Alaska and British Colombia, and during the winter migrate to breeding areas off Hawaii, Mexico, Central America, and Okinawa where breeding and calving occurs. As a result, humpback migrations are complex and cover great distances (Bettridge et al., 2015a; Calambokidis et al., 2009b; Calambokidis et al., 2008). Whales migrating from Hawaii to summer feeding habitat in the Gulf of Alaska will cover 2,600 NM over several weeks. Mate et al. (1998), used satellite tags to track three migrating whales using independent routes to Alaskan waters, and the fastest averaged 93 NM per day. At that rate, the whale would have reached waters off the coast of Alaska in about 39 days.

Hawaii Study Area

Humpback whales that breed in Hawaii generally migrate to northern British Columbia and southeast Alaska to feed (Bettridge et al., 2015a; Calambokidis et al., 2008). Animals breeding in Hawaii have also been "matched" (i.e., identified as the same individual using photo-identification methods) to humpbacks feeding in the Gulf of Alaska, the Aleutian Islands, and Bering Sea (Calambokidis et al., 2008). In all these feeding areas, humpback whales from Hawaii must cross paths with humpback whales migrating from Mexico and Central America (Figure C-10). In addition, based on the identification of individual whales, there is evidence that some humpback whales (most likely males) move between winter breeding areas in Hawaii and Mexico (Forestall and Urban-Ramirez 2007) and Hawaii and Japan (Salden et al. 1999).

In the Hawaii portion of their range, peak densities are from February through March, although the breeding season typically spans December through April (Baird et al., 2015d; Mobley et al., 1999; Mobley et al., 2001b; Norris et al., 1999). New survey data collected in offshore waters of the Main Hawaiian Islands in 2020 supported the development of the first habitat-based density model for humpback whale for the Hawaiian Islands EEZ (Becker et al., 2022b). This model provided further evidence that peak numbers of humpback whales occur within these waters from approximately 19 February through 22 March. Acoustic recordings near the northwestern Hawaiian Islands indicate that humpback whales were present in that portion of the HCTT Study Area from early December through early June (Lammers et al. 2011). It is not yet known if this represents a previously undocumented breeding stock or if the whales occurring at the northwestern Hawaiian Islands are part of the same population that winters near the Main Hawaiian Islands (Bettridge et al., 2015a). Acoustic recordings over multiple years (including 2016) using the PMRF hydrophones have demonstrated a seasonal presence of humpback whales off Kauai from November to May (Martin et al., 2016; Martin et al., 2017). The majority of humpback whales in Hawaii during the breeding season have been detected within the 200 m isobath (Mobley, 2005; Mobley et al., 2015; Mobley & Pacini, 2013; Mobley et al., 2001b). This presence may include very nearshore and inland water areas (Richie et al., 2016).

From December 2013 to January 2014, a passive acoustic recording device onboard an unmanned glider moving in the deep ocean approximately 100–300 km south of Oahu recorded humpback whale songs during all recording periods (Klinck et al., 2015). While the acoustic data do not provide an indication for how far away the animals are from the recorder, they would have definitely been offshore as opposed to nearshore shallow water areas previously documented as their preferred habitat. Mate et al. (2019), reported on the movements of 24 humpback whales tagged off Maui. While in Hawaiian waters the whales mainly remained in the Maui Nui region and Penguin Banks with a few whales venturing to Middle Bank in the Papahānaumokuākea Marine National Monument. Twelve tags were still transmitting when the whales began their northward migration to foraging habitat, and three of those whales were tracked to waters off British Columbia, Canada. The trajectories of other tracks indicated that several whales were headed towards the Gulf of Alaska and the Aleutian Islands.

There have been six locations identified in the main Hawaiian Islands as a single reproductive area for humpback whales (Baird et al. 2015). The greatest densities of humpback whales (including calves) have been in the four-island region consisting of Maui, Molokai, Kahoolawe, and Lanai, as well as Penguin Bank (Mobley et al., 2001b) and around Kauai (Mobley, 2005). A March 2007 pilot survey across the Northwestern Hawaiian Islands documented the existence of extensive wintering habitat used by humpback whales in the Northwestern Hawaiian Islands (Johnston et al., 2007). Two humpback whale reproductive BIAs (a parent and a child BIA) have been delineated in the main Hawaiian Islands during the overwintering breeding season (Kratofil et al., 2023)(Figure C-19). The BIAs were updated from the original BIAs (Baird et al., 2015d) based on satellite tag data collected from 1995 to 2019. The parent BIA encompasses 23,042 km^2 and the child BIA encompasses 6,679 km^2, including what are likely the most important reproductive areas for humpback whales in the Main Hawaiian Islands (Kratofil et al., 2023). The BIAs are in effect from December through May.

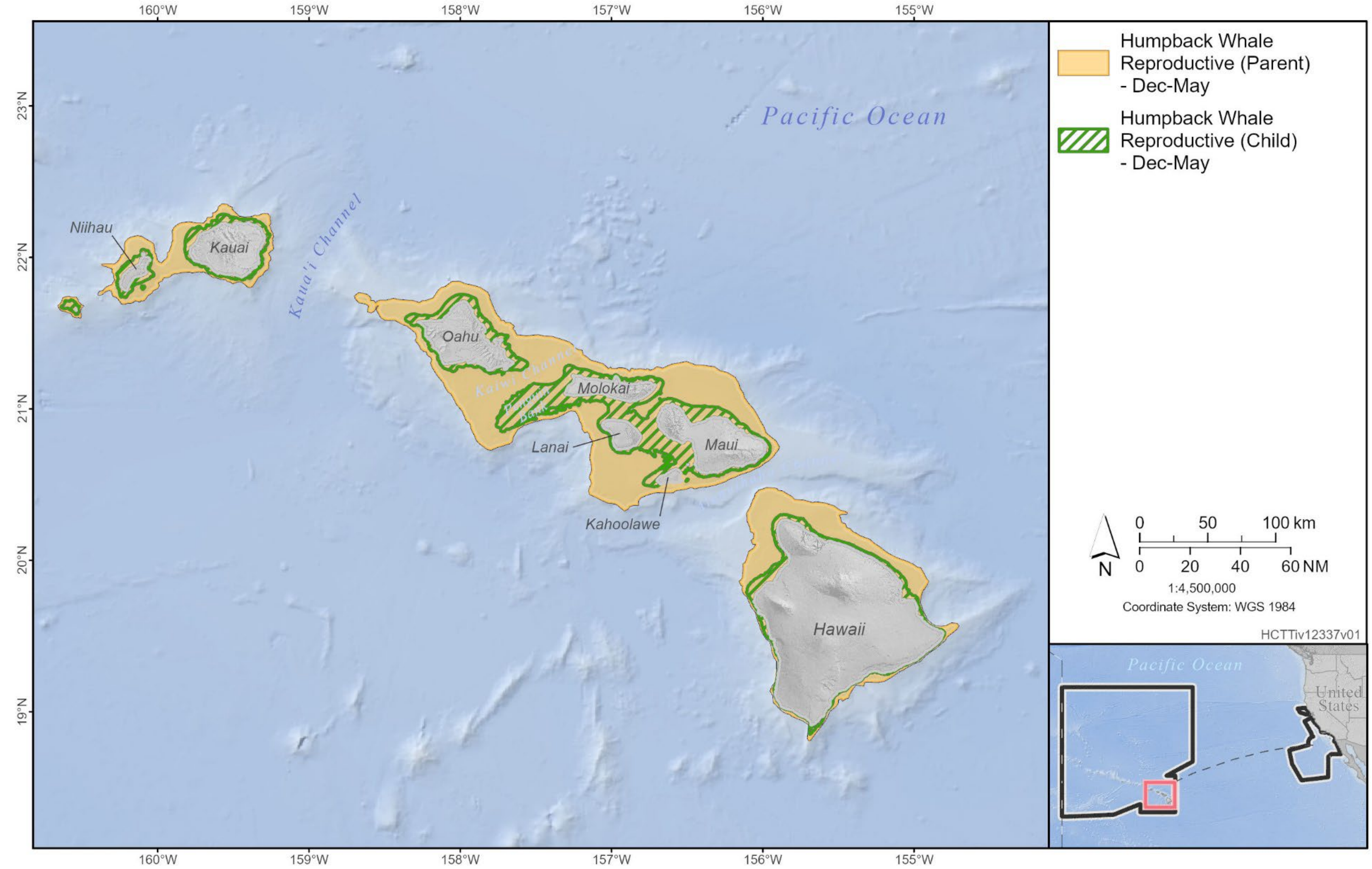


Figure C-19: Humpback Whale Reproductive BIAs in the Hawaii Study Area

Humpback whales migrating from breeding grounds in Hawaii to feeding grounds at higher latitudes may cross eastern portions of the HCTT Study Area Transit Corridor. Whales tagged off Maui, Kauai, and the island of Hawaii in 2018 spent a limited amount of time in the Hawaii Range Complex, with just 15 percent of tagged whales occurring within W-188A and 14 percent in W-188B (Mate et al., 2018). A higher proportion of whales tagged off Kauai entered Navy training areas; however, this may be a function of tagging location bias given Kauai's closer proximity to Navy training areas. No whales tagged off the island of Hawaii entered Navy training areas.

C.6.3.2.3 Population Trends

Until recently, there was evidence that the population of humpback whales in the Hawaiian Islands had been increasing since the early 1980s and had reached numbers greater than some pre-whaling abundance estimates (Barlow et al., 2011; Wade et al., 2016). Various abundance estimates for humpback whales in Hawaii also provided evidence of an increasing trend, with the annual estimated rate of increase varying from approximately 5 percent to 10 percent per year (Young, 2023). More recently however, declines in the numbers of humpback whales in the feeding areas in Alaska have raised some uncertainty regarding the current trend of the Hawaii stock (Young, 2023).

C.6.3.2.4 Population Threats

Humpback whales are susceptible to entanglement in fishing gear and ship strikes. Pot and trap fishery entanglements are the most-common source of injury to humpback whales along the U.S. west coast (Carretta et al., 2022). Between 2015 through 2019, there have been 81 observations of humpback whale interactions with pot and trap fisheries (Carretta et al., 2022). Of these interactions, serious injuries and mortalities were calculated as 51.75 humpback whales during this time period (Carretta et al., 2022). From 2015 to 2019, there was 79 humpback whale interactions with gillnet and unidentified fisheries (Carretta et al., 2022). Gillnet related serious injuries and mortalities totaled to 54.75 for this time period (Carretta et al., 2022).

Available data from NMFS indicate that there have been 14 reports of humpback whales struck by ships from 2016 through 2020, totaling to 2.6 humpback whale serious injuries or mortalities per year (Carretta et al., 2022).

Humpback whales are also potentially affected by underwater noise, jet skis and similar fast waterborne tourist-related traffic, and pollutants (Muto et al., 2017).

C.6.3.3 Common Minke Whale (*Balaenoptera acutorostrata*)

C.6.3.3.1 Status and Management

The common minke whale is protected under the MMPA and is not listed under the ESA. NMFS has designated three stocks of minke whale in the U.S. North Pacific: (1) the Hawaii stock, (2) the California/Oregon/Washington stock, and (3) the Alaska stock (Carretta et al., 2023b). Minke whales in Hawaii or California are considered to belong to their respective separate stocks.

C.6.3.3.2 Habitat and Geographic Range

The common minke whale's range is known to include the open ocean, coastal waters, and extends from subarctic to arctic waters (Jefferson et al., 2015; Kuker et al., 2005). The migration paths of some populations of minke whales include travel between low-latitude breeding grounds in the winter to high-latitude feeding grounds in the summer (Jefferson et al., 2015), although in some areas there appears to be resident populations (Dorsey et al., 1990).

Common minke whales previously were considered a rare species in Hawaiian waters due to limited sightings and detections (Barlow, 2006; Carretta et al., 2017b; Klinck et al., 2015; Lammers et al., 2015). However, during a 2002 survey around the Hawaiian Islands, minke whales were confirmed as the source of the mysterious “boing” sound of the north Pacific Ocean, specifically offshore of Kauai and closer in, near the PMRF, Barking Sands region (Barlow et al., 2004; Rankin & Barlow, 2005). This information subsequently allowed for acoustic detections of minke whales, although they remain rarely observed during visual surveys and its now widely accepted that their cryptic surfacing behavior is the reason for the low sighting rates (Barlow, 2006; Bradford et al., 2013; Bradford et al., 2017; Lammers et al., 2015; Rankin et al., 2007). In the summer, minke whales are likely absent from low-productivity tropical waters (Jefferson et al., 2015; Perrin et al., 2009a), and based on acoustic data, it is likely that in summer they have migrated north out of Hawaiian waters to feed (Martin et al., 2022). During three separate line-transect surveys of the Hawaii EEZ during summer and fall, minke whales were only seen and/or acoustically detected during the fall months (Barlow, 2006; Bradford et al., 2017; Bradford et al., 2021). Research involving passive acoustic detection now suggests minke whales are somewhat common in Hawaii in the winter(Klinck et al., 2015; Rankin & Barlow, 2005; Rankin et al., 2007; U.S. Department of the Navy, 2011). Acoustic recordings over multiple years using the PMRF hydrophones have demonstrated a seasonal presence of minke whales off Kauai from November to May (Martin et al., 2017).

Common minke whales occur year-round off California (Forney & Barlow, 1998b; Forney et al., 1995), mainly in nearshore areas (Barlow & Forney, 2007; Becker et al., 2020; Hamilton et al., 2009; Smultea & Jefferson, 2014), and “resident” minke whales off the U.S. West Coast appear behaviorally distinct from migratory whales that occur further north (Dorsey et al., 1990{Carretta, 2023 #14843)}. During year-round aerial surveys conducted in the SOCAL Range Complex from 2008 through 2013, minke whales were sighted 19 times (Jefferson et al., 2014). Sighting data collected during 9 systematic ship surveys between 1991 and 2018 off the U.S. West Coast provided sufficient sample sizes to develop the first habitat-based density model for minke whale in this study area (Becker et al., 2020). Consistent with previous observations, the model predicted greatest numbers of minke whales in nearshore areas over the continental shelf, with relatively low variability in annual distribution patterns.

C.6.3.3.3 Population Trends

There are no data on population trends for common minke whales in the Hawaiian stock (Carretta et al., 2023b). Based on abundance estimates from design- and model-based estimates derived from line-transect ship survey data collected between 1991 and 2018 off the U.S. West Coast (Barlow, 2016; Becker et al., 2020), there are no apparent trends in the population size of the California/Oregon/Washington stock of common minke whales.

C.6.3.3.4 Population Threats

Common minke whales are susceptible to entanglement in fishing gear and vessel strikes. Off California, minke whales interact with coastal set gillnets and offshore set gillnets. From 2017 through 2021, the estimated bycatch of this species in the California swordfish drift gillnet fishery was 0.02 minke whales annually based on modeling using recorded entanglement interactions from 1990 through 2021 (Carretta et al., 2024; Carretta, 2022a). Additionally, there were two recorded interactions of minke whales with other fisheries off California from the same 5-year period. One of the two interactions was with an unidentified fishery, resulting in serious injury, while the other was with the Dungeness crab pot fishery in San Diego and did not result in serious injury (Carretta et al., 2023a). In Hawaii, common minke whales are threatened by the deep-set longline fishery and the shallow-set longline fishery. The most recent monitoring data from 2014 through 2018 indicate that there have been no observed entanglements or hookings of this species in these fisheries (Bradford, 2018a; Bradford, 2018b; Bradford, 2021; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019). There were no reported vessel strikes of minke whales from 2017 through 2021 off the U.S. West Coast; however, strikes of this species are more likely to go undetected (Carretta et al., 2023a).

Most recent data from 2017 through 2021 indicate that there was one recorded mortality of a minke whale from the California/Oregon/Washington stock from a shooting, likely attributed to fisheries interactions (Carretta et al., 2023a).

Climate change, along with other anthropogenic threats, has increasingly become a threat to marine mammals, including baleen whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.4 Gray Whale (*Eschrichtius robustus*; Eastern North Pacific Stock)

C.6.3.4.1 Status and Management

NMFS recognizes two stocks of gray whales in the North Pacific: the Eastern North Pacific stock and the Western North Pacific stock (Weller et al., 2013). Both stocks could be present in the California portion of the Study Area during their northward and southward migrations (Mate et al., 2015b; Sumich & Show, 2011). The Western subpopulation has previously been known as the Korean-Okhotsk population (Carretta et al., 2023b). This stock is critically endangered, shows no apparent signs of recovery, and should be very rare in the California Study Area given their low abundance.

The Eastern North Pacific stock (also known as the California-Chukchi population) has recovered from whaling exploitation and was removed from listing under the ESA in 1994 (Swartz et al., 2006). An UME for the eastern North Pacific stock declared by NMFS in 2019 was closed in March 2024, but it lasted from December 17, 2018, to – November 9, 2023, and resulted in hundreds of gray whale deaths in the eastern North Pacific leading to a 40 percent decline in the population (National Marine Fisheries Service, 2024d). The two stocks are genetically distinguished but the population may be changing due to evolving migratory patterns of both stocks (Carretta et al., 2023b).

C.6.3.4.2 Habitat and Geographic Range

Gray whales are not expected to occur regularly in the Hawaii Study Area or in the transit corridor. There were two sightings of what appeared to be a juvenile gray whale off the island of Hawaii in February 2022, but this is the first recorded sighting of this species in the central Tropical Pacific and it is considered very unusual (Baird et al., 2022).

The vast majority of gray whales occurring off the West Coast and in the California Study Area are from the larger Eastern North Pacific stock (Carretta et al., 2023b). Whales from the eastern subpopulation are found in the Chukchi, Beaufort, and Bering seas in summer and fall foraging seasons; however, a small number feed along the coast between Kodiak Island, Alaska and northern California during that time (Calambokidis et al., 2017; Gosho et al., 2011). This group of whales was designated as the Pacific Coast Feeding Group by the International Whaling Commission and for purposes of abundance estimation was defined as gray whales feeding between northern California and northern British Columbia, Canada from June through November (Carretta et al., 2023b).

Gray whales are known to make one of the longest annual migrations of any mammal, 15,000–20,000 km roundtrip (Jefferson et al., 2015; Jones & Swartz, 2009). Eastern North Pacific gray whales begin their migration from breeding areas off Mexico and along the coast of North America in late fall through early spring to reach foraging areas by summer (Carretta et al., 2023b; Urbán et al., 2021; Weller et al., 2012a) and would only be present in the California Study Area while during northbound and southbound migrations. A year-long (2013–2014) survey effort in the nearshore waters off San Diego encountered gray whales in January, February, and in the April-June timeframe (Graham & Saunders, 2015). For purposes of this analysis, the Action Proponents assumed that a very small percentage of gray whales migrating off California could be individuals from the endangered Western North Pacific stock and that the majority would be from the unlisted Eastern North Pacific stock.

The timing of the October-July gray whale migrations that pass through the California Study Area can be loosely categorized into three phases (Calambokidis et al., 2015b; Jones & Swartz, 2009; Mate et al., 2013; Mate et al., 2015b; Mate & Urban-Ramirez, 2003; Rugh et al., 2008; Rugh et al., 2005; Urbán et al., 2021). Calambokidis et al. (2015b) noted these migration phases are not distinct; the timing for a phase may vary based on environmental variables, and that a migration phase typically begins with a rapid increase in migrating whales, followed by moderate numbers over a period of weeks, and then slowly tapering off. A southward migration from summer feeding areas includes all age classes as they migrate primarily to the nearshore waters and lagoons of Baja California, Mexico. During this southward migration from October through March, the whales generally are within 10 km of the coast (Calambokidis et al., 2015b) although there are documented exceptions where migrating gray whales have bypassed the coast by crossing sections of the open ocean (Mate et al., 2015b; Mate & Urban-Ramirez, 2003). In the California Study Area, migrating gray whales may deviate farther from the mainland as some are routinely seen near the Channel Islands and to the west of SCI (Sumich & Show, 2011).

(Calambokidis et al., 2015c) delineated four migratory BIAs off the U.S. West Coast for the Eastern North Pacific stock of gray whales. The four areas were defined by season as well as age and sex classes to capture the variation in migratory behavior of the species. Calambokidis et al. (2024) modified the BIA delineations by incorporating new data and historical sightings, focusing on regional differences in migratory behavior, considering that the Phase B northbound migration used by mother-calf pairs should also be treated as a reproductive BIA, and applying the new parent-child hierarchy. Four migratory BIAs were created (Figure C-20):

- West Coast to Gulf of Alaska (parent) June – November
- Southbound (child) November - February
- Northbound Phase A (child) January - May
- Northbound Phase B (child) March - May

The parent migratory BIA was revised from the original southbound BIA defined by Calambokidis et al. (2015c) and extended north to connect with the Gulf of Alaska migratory BIA (Wild et al., 2023). The revised BIA is also referred to as the transboundary migratory BIA. The southbound (child) BIA is for all age and sex classes and extends 10 km from shore off California (and broadens to 15 km off Oregon and 30 km off Washington). The Northbound Phase A (child) BIA, primarily for adults and juveniles, extends 8 km from shore off California, and broadens to 15 km off Oregon and 20 km off Washington. The Northbound Phase B (child) BIA is primarily for mother-calf pairs and extends 5 km from shore north of the Southern California Bight and fall entirely within the Northbound Phase A (child) BIA.

Calambokidis et al. (2024) noted that two satellite-tagged Western North Pacific gray whales were documented using migratory corridors off the U.S. West Coast; however, the data used to delineate the BIAs were almost entirely from gray whales in the Eastern North Pacific stock.

In addition to the migratory BIAs, a reproductive BIA was delineated to coincide with the Northbound Phase B (child) migratory BIA for mother-calf pairs (Figure C-20) and a feeding BIA (Figure C-21), which is located north of the Study Area, off the coast of the Pacific Northwest.

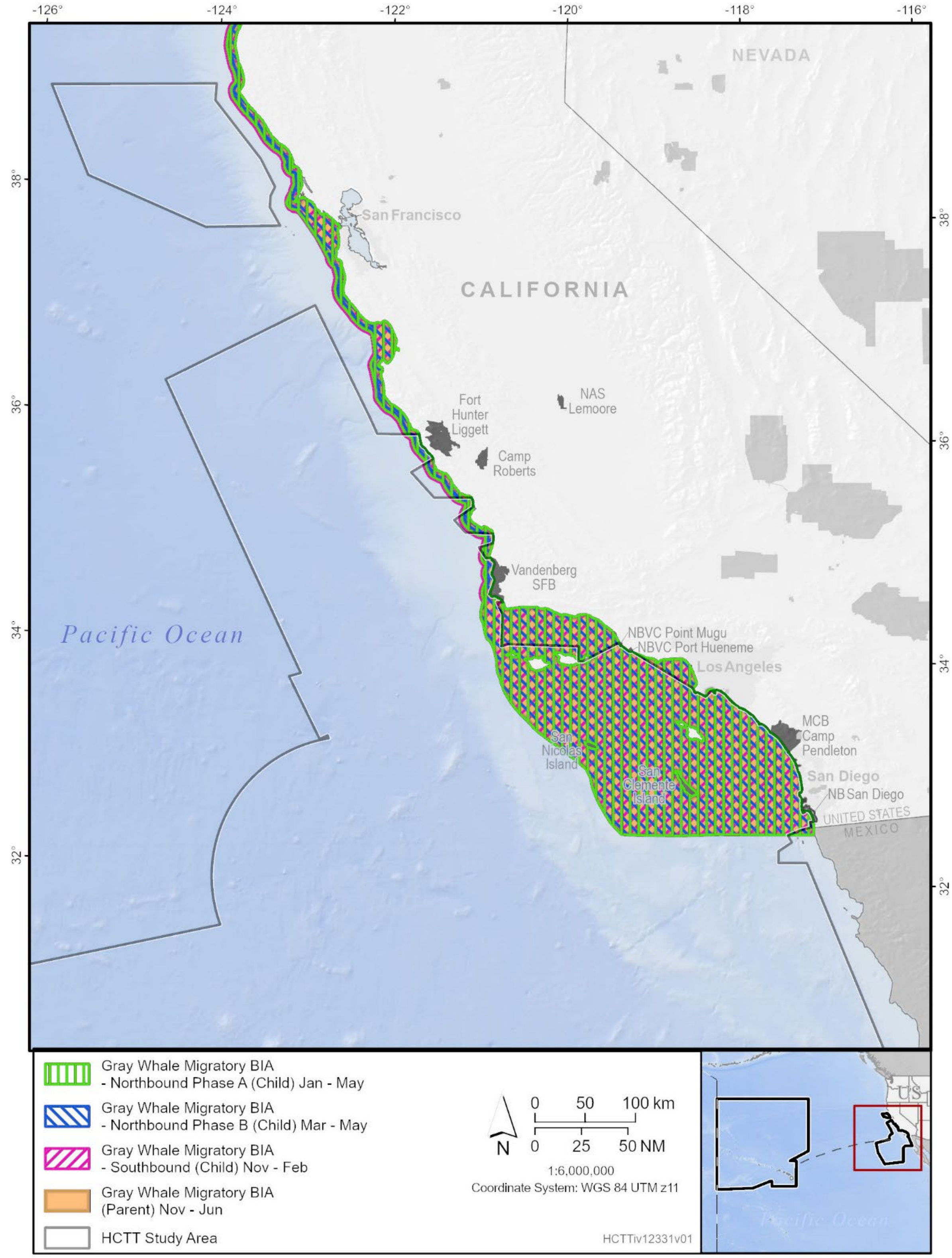


Figure C-20: Gray Whale Migratory BIAs in the California Study Area

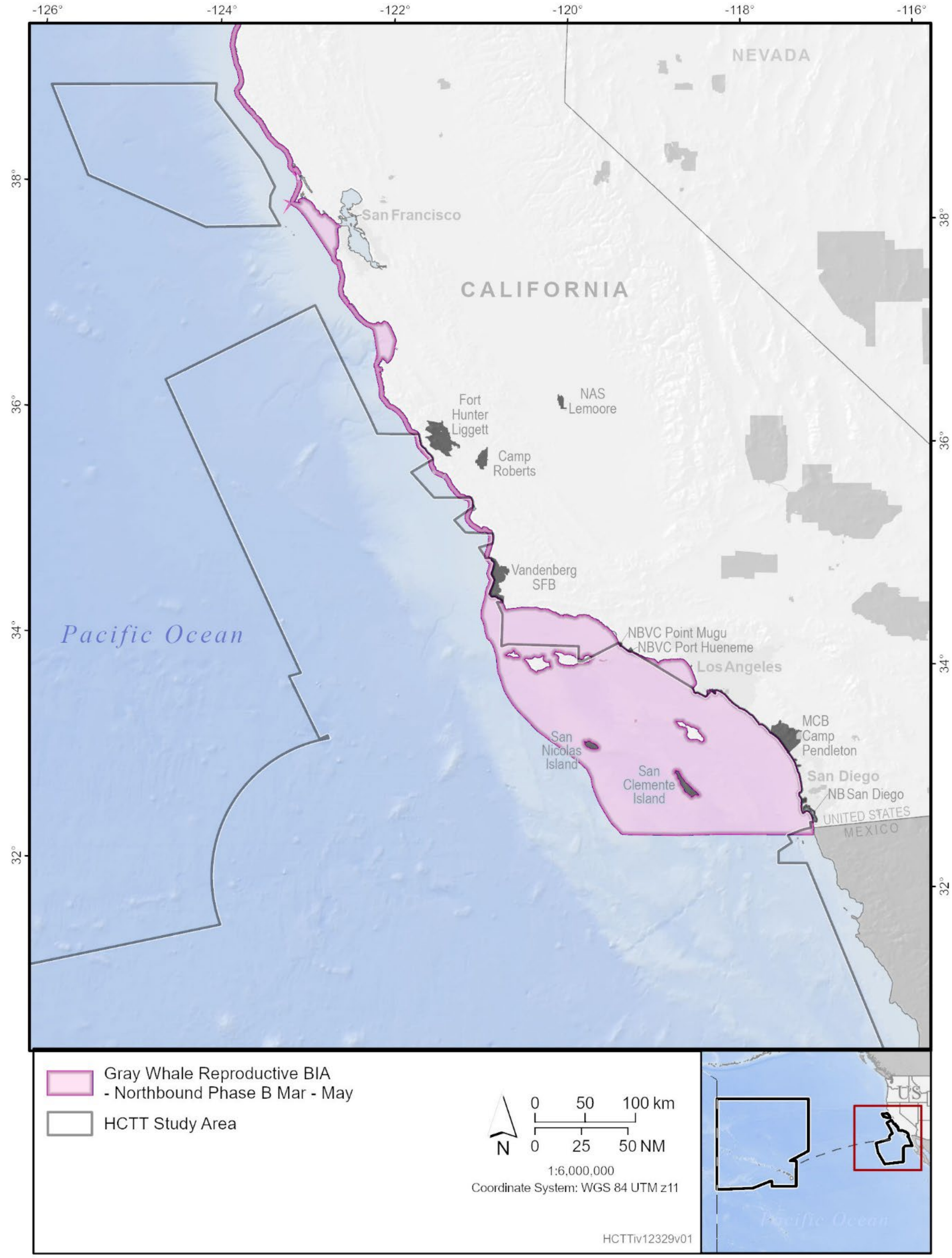


Figure C-21: Gray Whale Reproductive BIA in the California Study Area

Based on the identified migratory seasons, gray whales should occur off the California coast for most of the year with the exception of the July – October timeframe (Calambokidis et al., 2024; Calambokidis et al., 2015b).

C.6.3.4.3 Population Trends

The Eastern North Pacific stock of gray whales is estimated to be 26,960 with 243 whales in the Pacific Coast Feeding Group (Carretta et al., 2023b). Despite the UME from 2019 – 2024, the eastern population is expected to recover based on historical responses to similar events (e.g., 1999 – 2000 event). The Western North Pacific subpopulation of gray whale was once considered extinct but now small numbers are known to exist (Carretta et al., 2017b; Cooke et al., 2015; International Union for Conservation of Nature, 2011; International Whaling Commission, 2014; Mate et al., 2015b; Weller et al., 2013). There are no current population trend data available at this time (Carretta et al., 2017b), however, previous data on population growth indicated a positive growth of roughly 2.5 to 3.2 percent per year (National Marine Fisheries Service, 2014f).

C.6.3.4.4 Population Threats

Gray whales have historically been harvested by subsistence hunters in Alaska and Russia. The International Whaling Commission sets catch limits on the annual subsistence harvest for these areas. For example, the Chukotka indigenous hunters (located on the Chukchi Peninsula) took a total of 127 gray whales in 2013 (Ilyashenko & Zharikov, 2014). In 2010, a gray whale discovered dead onshore in Humboldt, California had two embedded harpoons in its flesh; one of these harpoons had 10 m of rope attached (Carretta et al., 2016a).

Gray whales are also susceptible to entanglement in fishing gear and ship strikes, particularly incidental catches in coastal fisheries (Carretta et al., 2021b). Based on photographic data of western gray whales on their feeding ground off Sakhalin Island (Russia), approximately 19 percent of whales in the sample had detectable anthropogenic scarring resulting from fishing gear entanglement (Bradford et al., 2009).

Natural population threats to gray whales include disease, parasites, reduced prey availability, and predator attacks. The gray whale is preyed on particularly by killer whales. Many individuals exhibit attack scars indicating not all attacks are fatal, however fatalities are known. Killer whale predation of gray whales has been documented in California waters off Monterey (PHYSORG, 2017) as well as the targeting of calves during the spring migration into colder northern waters (Jones & Swartz, 2009).

C.6.3.5 Dwarf Sperm Whale (*Kogia sima*)

There are two species of *Kogia*: the pygmy sperm whale (discussed in Section C.6.3.6) and the dwarf sperm whale, which had previously been considered to be the same species. Dwarf and pygmy sperm whales are difficult to distinguish from one another at sea, and many misidentifications have been made. Sightings of either species are often categorized as the genus *Kogia* (Jefferson et al., 2015).

C.6.3.5.1 Status and Management

The dwarf sperm whale is protected under the MMPA and is not listed under the ESA. Dwarf sperm whales within the Pacific U.S. EEZ are divided into two separate stocks: (1) the Hawaiian stock and (2) the California, Oregon, and Washington stock (Carretta et al., 2023b).

C.6.3.5.2 Habitat and Geographic Range

Dwarf sperm whales tend to occur largely in offshore waters (Jefferson et al., 2015), although they may be relatively coastal in some areas with deep waters nearshore (MacLeod et al., 2004). Although the dwarf sperm whale appears to prefer more tropical waters than the pygmy sperm whale, the exact habitat preferences of the species are not well understood.

There were a total of six pygmy sperm whale sightings during systematic ship surveys within the Hawaiian Islands EEZ in 2002, 2010, and 2017, and none of the sightings were in waters within 140 km of the Main Hawaiian Islands (Bradford et al., 2021). During small boat surveys between 2002 and 2012 in the main Hawaiian Islands, this species was the fifth most frequently encountered species of odontocete in waters shallower than 1,000 m with a strong peak in the sighting rate where depths are between 500 and 1,000 m (Baird et al., 2013b; Oleson et al., 2013). Dwarf sperm whales have been seen near Niihau, Kauai, Oahu, Lanai, and Hawaii. Photo-identification of individuals off Hawaii Island since 2003 has provided evidence of long-term site fidelity, with a third of identified individuals being seen in more than one year, and therefore suggesting the existence of an island-resident population (Baird et al., 2015c; Oleson et al., 2013).

Along the U.S. Pacific coast, no reported sightings of this species have been confirmed as dwarf sperm whales and it is likely that most *Kogia* species off California are pygmy sperm whale (*Kogia breviceps*) (Carretta et al., 2023b). There were no *Kogia* detected during 15 aerial surveys conducted in the SOCAL Range Complex from 2008 through 2012 (Smultea et al., 2014). This may be somewhat due to their pelagic distribution, cryptic behavior (i.e., “hidden” because they are not very active at the surface and do not have a conspicuous blow), and physical similarity to the pygmy sperm whale (Jefferson et al., 2008; McAlpine, 2009). However, the presence of dwarf sperm whales off the coast of California has been demonstrated by at least five dwarf sperm whale strandings in California between 1967 and 2000 (Carretta et al., 2010).

BIAs were redefined for a year-round Small and Resident Population area for dwarf sperm whales off the west coast of the Island of Hawaii by (Kratofil et al., 2023), and incorporated additional sighting data not available when the original BIA was defined (Baird et al., 2015c). The parent BIA is 1,341 km^2 in size and encompasses all sighting locations in waters less than 2,000 m (Figure C-22). The child BIA represents an area of intensified use relative to the entire range of this island-associated population and encompasses 457 km^2.

C.6.3.5.3 Population Trends

In the Hawaiian Islands, there are no data available for assessing a population trend (Carretta et al., 2023b).

There is no information available to estimate the population size of dwarf sperm whales off the U.S. west coast. There are no known sighting records of this species despite many vessel surveys along the west coast, and sightings of unidentified *Kogia* species are likely to be pygmy sperm whales (Carretta et al., 2023b). Due to the lack of data, there is no way of estimating an abundance trend.

C.6.3.5.4 Population Threats

Dwarf sperm whales are threatened by the deep-set longline fishery and the shallow-set longline fishery in Hawaii. The most recent monitoring data from 2014 through 2018 indicate that there has been one probable entanglement of this species in the deep-set longline fishery and none observed in the shallow-set longline fishery (Bradford, 2018a; Bradford, 2018b; Bradford, 2021; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019). Off the U.S. West Coast, dwarf sperms whales would be most likely to interact with the California swordfish gillnet fishery; however, there are no observed entanglements of this species from 1990 through 2014 (Carretta et al., 2019a; Carretta, 2023).

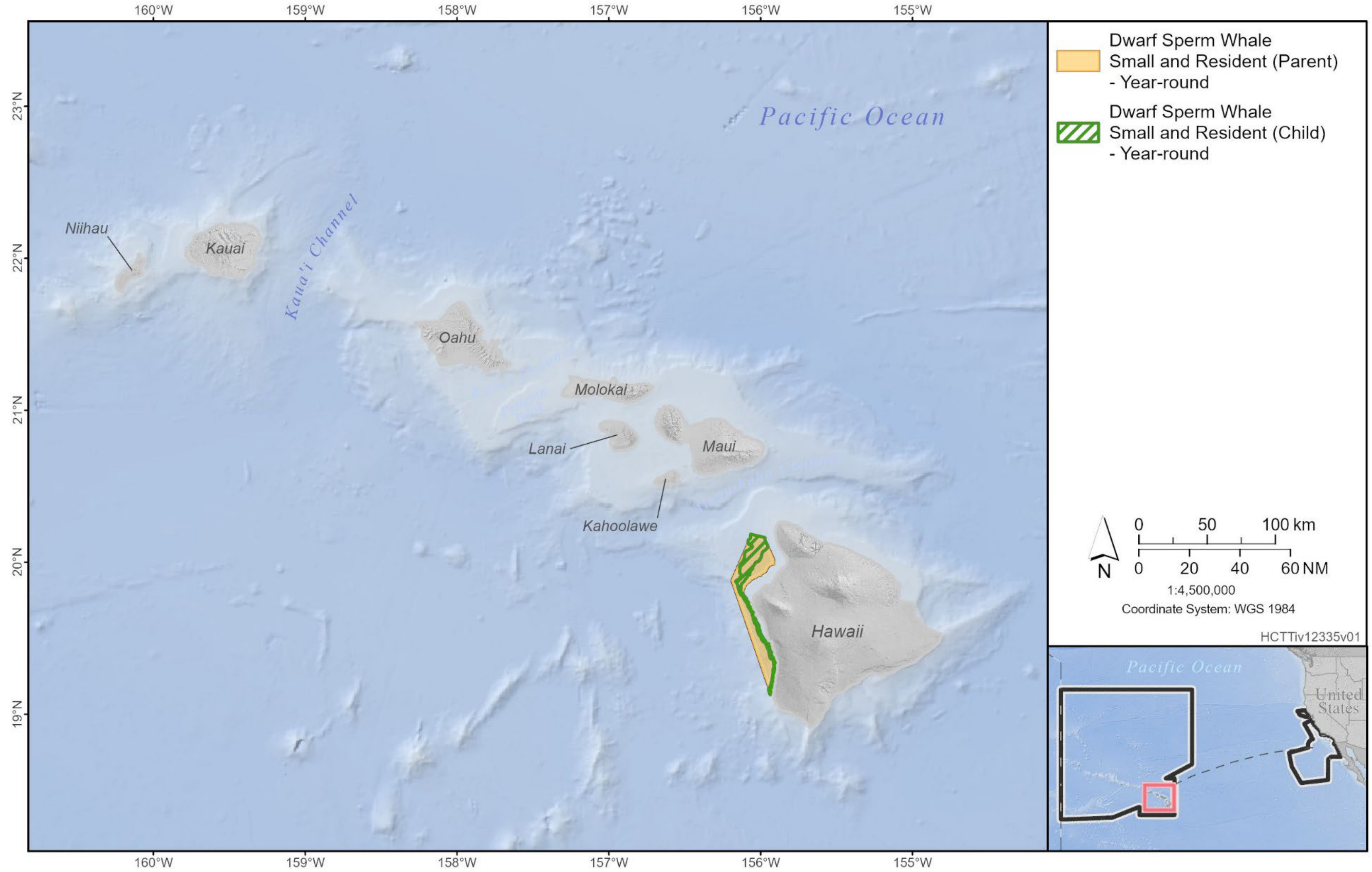


Figure C-22: Dwarf Sperm Whale Small and Resident BIAs in the Hawaii Study Area

C.6.3.6 Pygmy Sperm Whale (*Kogia breviceps*)

There are two species of *Kogia*: the pygmy sperm whale (*Kogia breviceps*) and the dwarf sperm whale (*Kogia sima*; discussed in Section C.6.3.5). Dwarf and pygmy sperm whales are difficult to detect and distinguish from one another at sea, and many misidentifications have been made. Sightings of either species are often categorized as the genus *Kogia* (Jefferson et al., 2015).

C.6.3.6.1 Status and Management

The pygmy sperm whale is protected under the MMPA but is not listed under the ESA. Pygmy sperm whales are divided into two discrete stocks: (1) the Hawaiian stock and (2) the California, Oregon, and Washington stock (Carretta et al., 2023b).

C.6.3.6.2 Habitat and Geographic Range

Pygmy sperm whales occur in tropical to temperate waters of all oceans, generally in deep oceanic waters beyond the continental shelf (Jefferson et al., 2015). The pygmy sperm whale frequents more temperate habitats than the dwarf sperm whale, which is more of a tropical species. Movement patterns for the pygmy sperm whale are poorly understood, and available data are insufficient to identify any seasonal patterns in the distribution of pygmy sperm whales in the HCTT Study Area.

Sightings of pygmy sperm whales are rarely reported in Hawaii (Baird et al., 2013c; Oleson et al., 2013). There were a total of five pygmy sperm whale sightings during systematic ship surveys within the Hawaiian Islands EEZ in 2002, 2010, and 2017, and none of the sightings were in waters of the Main Hawaiian Islands (Bradford et al., 2021). During small boat surveys between 2000 and 2012 in the Main Hawaiian Islands, this species was observed, but less commonly than the dwarf sperm whale (Baird, 2005; Baird et al., 2003b; Baird et al., 2013c; Barlow et al., 2004; Oleson et al., 2013). Pygmy sperm whales are one of the more commonly stranded species in the Hawaiian Islands, and this frequency of strandings indicates that the species is likely more common than sightings suggest (Maldini et al., 2005).

The majority of field sightings of *Kogia* off the U.S. west coast are likely to have been pygmy sperm whales, but the limited number of confirmed sightings cannot be used to produce a reliable population estimate (Carretta et al., 2023b). *Kogia* species are thus treated as a genus in this region by scientists who have published species density estimates for this study area. Several studies have suggested that this species generally occurs beyond the continental shelf edge (Bloodworth & Odell, 2008; MacLeod et al., 2004) and all confirmed pygmy sperm whale sightings off the U.S. west coast have been well offshore (Barlow, 2016; Hamilton et al., 2009). For California, a total of six pygmy sperm whale sightings have been made in offshore waters along the U.S. west coast during systematic surveys conducted between 1991 and 2014 (Barlow, 2016; Hamilton et al., 2009). There were no *Kogia* detected during 15 aerial surveys conducted in the SOCAL Range Complex from 2008 through 2012 (Smultea & Jefferson, 2014).

C.6.3.6.3 Population Trends

There are no data available for an analysis of population trends for pygmy sperm whales in the Pacific (Carretta et al., 2023b).

C.6.3.6.4 Population Threats

Pygmy sperm whales are susceptible to interactions with fisheries and entanglement in fishing gear. Off California, pygmy sperm whales have been observed entangled in coastal set gillnets and offshore set gillnets in waters. The California swordfish gillnet fishery is the fishery most likely to interact with this species in the region; however, monitoring data from indicate that there have been no entanglements of pygmy sperm whales in the fishery in the most recent monitoring period (Carretta et al., 2017a; Carretta et al., 2023b). Additionally, there is one recorded pygmy sperm whale death in 2002 due to a shooting that was likely from an interaction with a net fishery (Carretta et al., 2023b).

Off Hawaii, pygmy sperm whales are threatened by the deep-set longline fishery and the shallow-set longline fishery. The most recent monitoring data from 2014 through 2018 indicate that there has been one observed entanglement of this species in the deep-set longline fishery, which resulted in serious injury (Bradford, 2018a; Bradford, 2018b; Bradford, 2021; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019).

Anthropogenic noise such as sonar may result in unknown levels of injury or mortality to pygmy sperm whales and other species (Carretta et al., 2023b). In 1988, a mass stranding including 2 pygmy sperm whales occurred near the Canary Islands (Carretta, 2023; D'Amico, 2009). In 2005, there was an UME off the U.S. East Coast that resulted in a mass stranding of 2 dwarf sperm whales, 33 short-finned pilot whales, and a minke whale (Hohn et al., 2006). These strandings may be associated with military sonar use that occurred in theses area during the time of the strandings; however, a definitive association has not been confirmed (Hohn et al., 2006).

Climate change has increasingly become a threat to marine mammals. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.7 Killer Whale (*Orcinus orca;* Hawaii, Eastern North Pacific Offshore stocks, and *Orcinus orca rectipinnus*; Eastern North Pacific/West Coast Transient stocks)

Different geographic forms of killer whale are distinguished by distinct social and foraging behaviors and other ecological traits. In the North Pacific, these recognizable geographic forms are variously known as ‘‘residents,’’ ‘‘transients,” and “offshore” ecotypes (Hoelzel et al., 2007; Morin et al., 2024). Recently, new analyses suggest that species status may be warranted for two of the different eastern North Pacific ecotypes, the Bigg’s or transient killer whales and the resident killer whales, including the Eastern North Pacific Southern Resident stock (Morin et al., 2024). However, the proposal before the Society for Marine Mammalogy Committee on Taxonomy failed to receive the 2/3 majority needed to pass, so killer whales will remain as a single species, but Bigg’s and resident killer whales will now be recognized as separate subspecies. *Orcinus orca rectipinnus* and *Orcinus orca ater*, respectively. In the HCTT Study Area, both the transient and offshore ecotypes are known to occur, and Southern Resident killer whales may seasonally occur along the northern portion of the California Study Area north of Monterey Bay (Hanson et al., 2018);National Marine Fisheries Service, 2021 #15025;Millman, 2019 #3115}.

C.6.3.7.1 Status and Management

Eight killer whale stocks are recognized within the North Pacific U.S. EEZ, with only the Hawaiian stock occurring in Hawaii and three stocks occurring in the California Study Area consisting of the West Coast Transient stock, the Eastern North Pacific Offshore stock, and the Eastern North Pacific Southern Resident stock (Carretta et al., 2023b). Three separate pods comprise the Southern Resident stock, identified as the J, K, and L pods (Ford et al., 2000). Killer whales are protected under the MMPA and the Southern Resident killer whale stock or DPS is listed as endangered under the ESA. The other two stocks are not listed under the ESA.

C.6.3.7.2 Habitat and Geographic Range

Killer whales are found in all marine habitats from the coastal zone (including most bays and inshore channels) to deep oceanic basins and from equatorial regions to the polar pack ice zones of both hemispheres. Although killer whales are also found in tropical waters and the open ocean, they are most numerous in coastal waters and at higher latitudes (Dahlheim & Heyning, 1999). Forney and Wade (2006) found that killer whale densities increased by 1–2 orders of magnitude from the tropics to the poles.

Although killer whales apparently prefer cooler waters, they have been observed in Hawaiian waters (Baird et al., 2013c; Barlow, 2006; Mobley et al., 2001a; Shallenberger, 1981). There are also documented strandings for this species from the Hawaiian Islands (Maldini et al., 2005). Sightings are extremely infrequent in Hawaiian waters, and typically occur during winter, suggesting those sighted in Hawaii are seasonal migrants to Hawaii (Baird et al., 2003a; Baird et al., 2013c; Mobley et al., 2001a). During three separate systematic ship surveys of the Hawaiian EEZ in summer and fall, there were two killer whale sightings in 2002 a single sighting in 2010, and a single sighting in 2017, with the average group size ranging from 4.7 to 7.4 animals (Barlow, 2006; Bradford et al., 2017; Bradford et al., 2021). Baird (2006) documented 21 killer whale sightings within the Hawaiian EEZ, primarily around the main Hawaiian Islands during relatively nearshore small boat surveys. In the period from 2000 to 2012, there were two sightings with each pod consisting of four killer whales (Baird et al., 2013b). A single adult female was also sighted off Kauai in July 2011 (Cascadia Research Collective, 2012). A pod of killer whales was observed off the southwest coast of the island of Hawaii in May 2013 (Pacific Fishery Management Council, 2014).

All three ecotypes of killer whale are known to occur along the west coast of North America, from the entire Alaskan coast, in British Columbia and Washington inland waterways, and along the outer coasts of Washington, Oregon, and California, but the endangered resident ecotype’s range typically does not extend south of Monterey California (Calambokidis & Barlow, 2004; Carretta et al., 2017b; Dahlheim et al., 2008; Ford & Ellis, 1999; Forney et al., 1995; Hanson et al., 2018). In the southern portion of the California Study Area, only the transient and offshore ecotypes may be present (Carretta et al., 2023b). During seven systematic ship surveys of waters off the U.S. west coast between 1991 and 2014, there were 37 killer whale sightings, only five of which were off southern California (Barlow, 2016). An additional 10 killer whale sightings were made on a more recent survey in 2018, and except for a single sighting off Point Conception (34.4°N), all the sightings were made well north of the HCTT Study Area (Henry et al., 2020). Based on two sightings from 15 aerial surveys conducted in the SOCAL Range Complex from 2008 through 2012, killer whales were ranked 12th in occurrence as compared to other cetaceans (Jefferson et al., 2014; Smultea et al., 2014).

C.6.3.7.3 Population Trends

No data are available on current population trends for the Hawaiian stock of killer whales (Carretta et al., 2023b). The Eastern North Pacific stock is considered to be stable based on an estimated 98 percent survival rate and positive annual recruitment (Ford et al., 2014).

The population of West Coast Transient stock of killer whales grew rapidly from the mid-1970s to the mid-1990s coinciding with an increase in harbor seal abundance, their primary prey. Since 1990s, growth slowed, but continued to increase. However, no reliable estimate of population trend is available for the stock (Young, 2023).

C.6.3.7.4 Population Threats

The Hawaii stock of killer whales is susceptible to interactions with the deep-set longline fishery and the shallow-set longline fishery. However, the most recent monitoring data from 2014 through 2018 indicate that there have been no observed entanglements or hookings of killer whales in these fisheries (Bradford, 2018a; Bradford, 2018b; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019). Additionally, there have been no documented anthropogenic-caused injuries or mortalities of the Eastern North Pacific offshore stock of killer whale; however, this stock, like other killer whale stocks, is likely susceptible to interactions with fisheries and vessel strikes (Carretta et al., 2023b).

Exposure to contaminants may also harm killer whales. The presence of high levels of persistent organic pollutants, such as PCB, DDT, and flame-retardants, has been documented in southern resident killer whales (Krahn et al., 2007). Although the consequences of these pollutants on the fitness of individual killer whales and the population itself remain unknown, in other species these pollutants have been reported to suppress immune responses (Wright et al., 2007), impair reproduction, and exacerbate the energetic consequences of physiological stress responses when they interact with other compounds in an animal’s tissues (Wright et al., 2007). Because of their long-life span, position at the top of the food chain, and their blubber stores, killer whales would be capable of accumulating high concentrations of contaminants.

Climate change has increasingly become a threat to marine mammals, including killer whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.8 False Killer Whale (*Pseudorca crassidens;* the Hawaii pelagic stock and the Northwestern Hawaiian Islands stock)

NMFS currently recognizes three stocks of false killer whale in Hawaiian waters: the Hawaii pelagic stock, the Northwestern Hawaiian Islands stock, and the Main Hawaiian Islands insular stock (Bradford et al., 2015; Carretta et al., 2015; Carretta et al., 2023b; Forney et al., 2010; National Oceanic and Atmospheric Administration, 2012; Oleson et al., 2010a). All stocks of false killer whale are protected under the MMPA. The Hawaii Pelagic stock and the Northwestern Hawaiian Islands stock of false killer whales are not listed as threatened or endangered under the ESA. The Main Hawaiian Islands insular stock is listed as endangered under the ESA as a DPS (National Oceanic and Atmospheric Administration, 2012).

The species is not typically expected to be present in the California Study Area, and false killer whales are not included by NMFS as a managed species in California waters (Carretta et al., 2017b; Carretta et al., 2023b). Strandings and sightings of false killer whales have been recorded in Southern California and north, but these have generally been considered extralimital. During the unusually warm oceanographic conditions in 2014, whale watching boats photographed false killer whales in Southern California waters, but there were none sighted during the SWFSC systematic survey that year (Barlow, 2016). False killer whales do occur in waters off the Baja California Peninsula, Mexico within the HCTT Study Area (Hamilton et al., 2009).

C.6.3.8.1 Habitat and Geographic Range

This species is known to occur in deep oceanic waters off Hawaii, and elsewhere in the Pacific. False killer whales have been detected in acoustic surveys and are commonly observed in the eastern tropical Pacific, including waters off the Baja California Peninsula, Mexico within the Study Area (Carretta et al., 2015; Carretta et al., 2023b; Miyashita et al., 1996; Oswald et al., 2003; Wade & Gerrodette, 1993; Wang et al., 2001). False killer whale are also regularly found within Hawaiian waters and have been reported in groups of up to 100 over a wide range of depths and distance from shore (Baird et al., 2003b; Baird et al., 2013a; Bradford et al., 2018; Bradford et al., 2014; Bradford et al., 2015; Oleson et al., 2013; Shallenberger, 1981).

The ranges and stock boundary descriptions for false killer whales in the Hawaiian Islands are complex and overlapping. For example, all three stocks are known to overlap in the vicinity of Kauai and Niihau, which is where the Navy's underwater instrumented range has been in use since the 1980s. All significant information regarding the range of the three stocks was presented in Bradford et al. (2015), and later updated for the pelagic stock (Bradford et al., 2020). A summary of the data used to delineate the stock boundaries, and the research supporting those data are provided in the Final 2022 Pacific Stock Assessment Report (Carretta et al., 2023b) that is synthesized in the next few paragraphs for the stocks in the Hawaiian Islands.

The Main Hawaiian Islands insular stock is considered resident to the main Hawaiian Islands consisting of Kauai, Oahu, Molokai, Lanai, Kahoolawe, Maui, and Hawaii, although they have been satellite tracked as far as 115 km from the main Hawaiian Islands (Bradford et al., 2020; Bradford et al., 2012; Bradford et al., 2015; Carretta et al., 2015; Forney et al., 2010; National Oceanic and Atmospheric Administration, 2012; Oleson et al., 2010a). The Main Hawaiian Islands insular stock boundary is a 72 km radius extending around the main Hawaiian Islands, with the offshore extent of the radii connected on the leeward sides of Hawaii Island and Niihau to encompass the offshore movements of Main Hawaiian Islands insular stock animals within that region.

False killer whales in the Northwestern Hawaiian Islands stock have been seen as far as 93 km from the Northwestern Hawaiian Islands and near shore around Kauai and Oahu (Baird et al., 2012; Bradford et al., 2015). The Northwestern Hawaiian Islands stock boundary is defined by a 93 km radius around Kauai, Niihau, and the Northwestern Hawaiian Islands, with the boundary around the Northwestern Hawaiian Islands expanded latitudinally at the eastern end to encompass animal movements observed outside the 93 km radius.

Given new telemetry data that indicated that pelagic stock animals occurred within 5.6 km of the main Hawaiian Islands and throughout the Northwestern Hawaiian Islands, the previous inner pelagic stock boundary at 11 km from shore around each of the main Hawaiian Islands was removed (Bradford et al., 2020). The pelagic stock now has no inner or outer boundary within the Hawaiian Islands EEZ. There is now an overlap zone between the entirety of the Main Hawaiian Islands insular stock area and the pelagic stock area. There is also now an overlap zone between the entirety of the Northwestern Hawaiian Islands stock area and the pelagic stock area. All three stock boundaries overlap out to the Main Hawaiian Islands insular stock boundary between Kauai and Niihau and the Northwestern Hawaiian Islands stock boundary between Kauai and Oahu (Carretta et al., 2023b).

As noted previously, false killer whales are not usually expected to be present in the California Study Area. Older records document only a handful of sightings from areas such as Monterey Bay, Santa Catalina, and the Channel Islands (Baird, 2009a; Jefferson et al., 2008; Miller & Scheffer, 1986). False killer whales were not detected during the 15 aerial surveys conducted in the SOCAL Range Complex from 2008 through 2012 (Smultea & Jefferson, 2014). A nearshore marine mammal survey off San Diego Bay in March 2014 detected a false killer whale pod that was assumed to be the same pod that had been seen 6 days before off Dana Point (Graham & Saunders, 2015). Two years later in April–March

2016, a whale watch vessel out of Dana Point again sighted a pod of false killer whales in the same area (Ritchie, 2016). This species normally prefers warmer tropical waters found outside of southern California and the presence of this species to the north of its usual habitat was likely due to the warmer than normal water temperatures associated with a known El Niño event. Such documented strandings and sightings of false killer whales in California waters have thus been considered extralimital. False killer whales do occur in warmer waters off the Baja California Peninsula, Mexico within the HCTT Study Area (Hamilton et al., 2009).

C.6.3.8.2 Population Trends

No data are available for the derivation of population trends for either the Hawaii Pelagic stock or the Northwestern Hawaiian Islands stock of false killer whales in Hawaii (Carretta et al., 2023b).

C.6.3.8.3 Population Threats

Because false killer whales feed on large prey at the top of the food chain (e.g., squid, tunas) they may be impacted by competition with fisheries (Cascadia Research Collective, 2010). In Hawaiian waters, false killer whales are particularly susceptible to fishery interactions and entanglements (Baird et al., 2015f; Bradford & Forney, 2016). Most recent data from 2017 through 2021 indicates that there were no entanglements or hooks of false killer whales in the shallow-set longline fishery; however, there were 54 reported entanglements or hooks of this species in the deep-set longline fishery during the same time period (Carretta et al., 2023b). Of the deep-set longline fishery interactions, 18 were encountered in the outside of the Hawaii EEZ, where 14 were considered seriously injured and 2 were found dead. Of the remaining 36 encounters in the Hawaii EEZ, 23 whales were considered seriously injured and 5 were found dead (Bradford, 2018a; Bradford, 2018b; Bradford, 2021; Bradford et al., 2017; Bradford et al., 2021; Carretta et al., 2023b; McCracken, 2019).

Like many marine mammals, false killer whales also accumulate high levels of toxins in their blubber over the course of their long lives, but the consequence of that bioaccumulation remains unknown.

Climate change has increasingly become a threat to marine mammals. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.9 Pygmy Killer Whale (*Feresa attenuata*)

C.6.3.9.1 Status and Management

The pygmy killer whale is protected under the MMPA and is not listed under the ESA. For the MMPA stock assessment reports, there is a single Pacific management stock including animals found within the U.S. EEZ of the Hawaiian Islands and the adjacent high seas (Carretta et al., 2023b).

C.6.3.9.2 Habitat and Geographic Range

The pygmy killer whale is generally an open ocean deepwater species (Davis et al., 2000; McSweeney et al., 2009; Oleson et al., 2013; Würsig et al., 2000). Movement patterns for this species are poorly understood. During a NMFS 2014 systematic ship survey off the U.S. west coast, when there were unusually warm water conditions, a group of 27 pygmy killer whales was sighted in offshore waters of southern California (Barlow, 2016). Given that there is a remote likelihood for this species to occur regularly off the U.S. west coast, the 2022 Pacific Stock Assessment report does not include pygmy killer whales as a managed stock in California waters (Carretta et al., 2023b).

This species' range in the open ocean generally extends to the southern regions of the North Pacific Gyre and the southern portions of the North Pacific Transition Zone. Many sightings have occurred from cetacean surveys of the eastern tropical Pacific (Au & Perryman, 1985; Barlow & Gisiner, 2006; Wade & Gerrodette, 1993). This species is also known to be present in the western Pacific (Wang & Yang, 2006). Its range is generally considered to be south of 40° N and continuous across the Pacific (Donahue & Perryman, 2008; Jefferson et al., 2008). There was a total of 11 sightings of pygmy killer whales during three systematic ship surveys of the Hawaiian Islands EEZ in 2002 (3 sightings), 2010 (5 sightings), and 2017 (3 sightings), with average group size ranging from 14.6 to 25.7 animals (Bradford et al., 2021) .

A year-round Small and Resident Population area has been identified for pygmy killer whales off the Island of Hawaii (Baird et al., 2015a). The delineated area extends along the coast of Hawaii Island from northwest of Kawaihae to South Point and along the southeast coast of the island, as determined by locations from two satellite-tagged individuals, photo-identification data, extensive vessel-based survey data, and expert judgment (Baird et al., 2015a). Two year-round, non-hierarchical Small and Resident Population BIAs have been delineated in the main Hawaiian Islands for pygmy killer whales (Kratofil et al., 2023) (Figure C-23). The BIAs were updated from the original BIAs (Baird et al., 2015d) based on additional analyses. One BIA encompasses 7,416 km^2 of waters surrounding Oahu and Maui Nui, and the second BIA encompasses 5,201 km^2 around the Island of Hawaii (Kratofil et al., 2023).

C.6.3.9.3 Population Trends

Abundance estimates for pygmy killer whales have broad and overlapping confidence intervals, thus precluding a robust assessment of population trends (Carretta et al., 2023b).

C.6.3.9.4 Population Threats

Pygmy killer whales are susceptible to interactions with fisheries, particularly the deep-set longline fishery and the shallow-set longline fishery. However, the most recent monitoring data from 2014 through 2018 indicate that there have been no observed entanglements or hookings of pygmy killer whales in these fisheries (Bradford, 2018a; Bradford, 2018b; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019). Off the U.S. West Coast, the California swordfish drift gillnet fishery is the fishery most likely to interact with this species in the region. However, monitoring data from 1999 through 2014 indicate that there have been no entanglements of this species in the fishery during this time period (Bradford & Forney, 2017).

Anthropogenic noise such as sonar may result in unknown levels of injury or mortality to pygmy killer whales (Carretta et al., 2023b; Wang & Yang, 2006). In 2005, two pygmy killer whale strandings occurred near Taiwan, which could possibly be associated with the use of sonar during military training exercises that were occurring in the area (Wang & Yang, 2006).

Climate change has increasingly become a threat to marine mammals. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

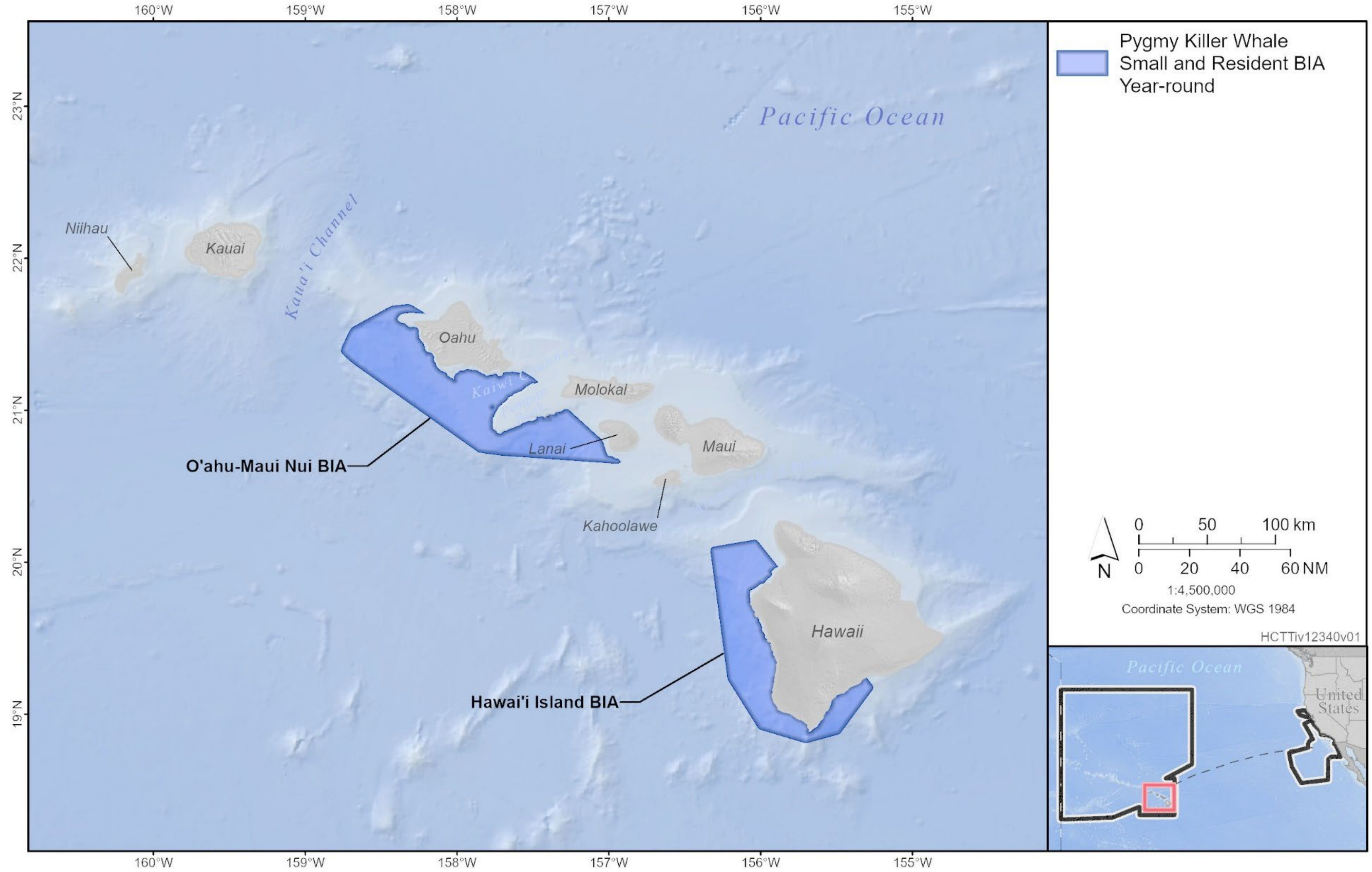


Figure C-23: Pygmy Killer Whale Small and Resident BIA in the Hawaii Study Area

C.6.3.10 Short-finned Pilot Whale (*Globicephala macrorhynchus*)

C.6.3.10.1 Status and Management

Short-finned pilot whales are protected under the MMPA and are not listed under the ESA. For MMPA stock assessment reports, short-finned pilot whales within the Pacific U.S. EEZ are divided into two discrete stocks: (1) the Hawaiian stock; and (2) the California, Oregon and Washington stock (Carretta et al., 2023b).

C.6.3.10.2 Habitat and Geographic Range

The short-finned pilot whale is widely distributed throughout most tropical and warm temperate waters of the world and occurs in waters over the continental shelf break, in slope waters, and in areas of high topographic relief (Baird et al., 2013b; Olson, 2009). While pilot whales are typically distributed along the continental shelf break, movements over the continental shelf are commonly observed in the northeastern U.S. (Payne & Heinemann, 1993) and close to shore at oceanic islands like Hawaii, where the shelf is narrow and deeper waters are found nearby (Baird, 2013; Gannier, 2000; Mignucci-Giannoni, 1998). Short-finned pilot whales are not considered a migratory species, although seasonal shifts in abundance have been noted in some portions of the species' range. A number of studies in different regions suggest that the distribution and seasonal inshore/offshore movements of pilot whales coincide closely with the abundance of squid, their preferred prey (Bernard & Reilly, 1999; Hui, 1985; Payne & Heinemann, 1993).

Short-finned pilot whales in the Hawaiian Islands were the most commonly encountered species of odontocete during near-shore surveys in depths over 2,000 m and were one of the most common species encountered during the NMFS 2002 (25 sightings), 2010 (36 sightings), and 2017 (35 sightings) systematic ship surveys of the Hawaiian EEZ (Baird et al., 2013b; Barlow, 2006; Bradford et al., 2013; Bradford et al., 2021; Oleson et al., 2013). Small boat surveys from 2003 through 2007 photo-identified 250 individuals seen in more than one year, suggesting site fidelity (Abecassis et al., 2015; Mahaffy et al., 2015; Oleson et al., 2013). Habitat-based models developed from systematic ship survey data collected in the central North Pacific show some of the highest short-finned pilot whale densities around the Hawaiian Islands (Becker et al., 2012b; Forney et al., 2015). Sighting data from systematic ship surveys conducted within waters of the Hawaiian Islands EEZ from 2000 to 2020 supported the development of an updated habitat-based density model for short-finned pilot whale and confirmed the strong island association indicated from the previous models (Becker et al., 2022a).

A year-round Small and Resident Population parent BIA and three child BIAs have been delineated for short-finned pilot whales in waters of the Main Hawaiian Islands (Kratofil et al., 2023) (Figure C-24). The BIAs were updated from the original BIA (Baird et al., 2015d) based on additional analyses. The parent BIA encompasses 58,999 km^2 of waters surrounding all of the Main Hawaiian Islands. The child BIAs encompass three communities representing core habitat in the Main Hawaiian Islands: a western community (4,040 km^2), a central community (2,427 km^2), and an eastern community (2,658 km^2) that encompass waters mainly on the leeward sides of Kauai, Oahu. Lanai, and the Island of Hawaii (Kratofil et al., 2023) (Figure C-24).

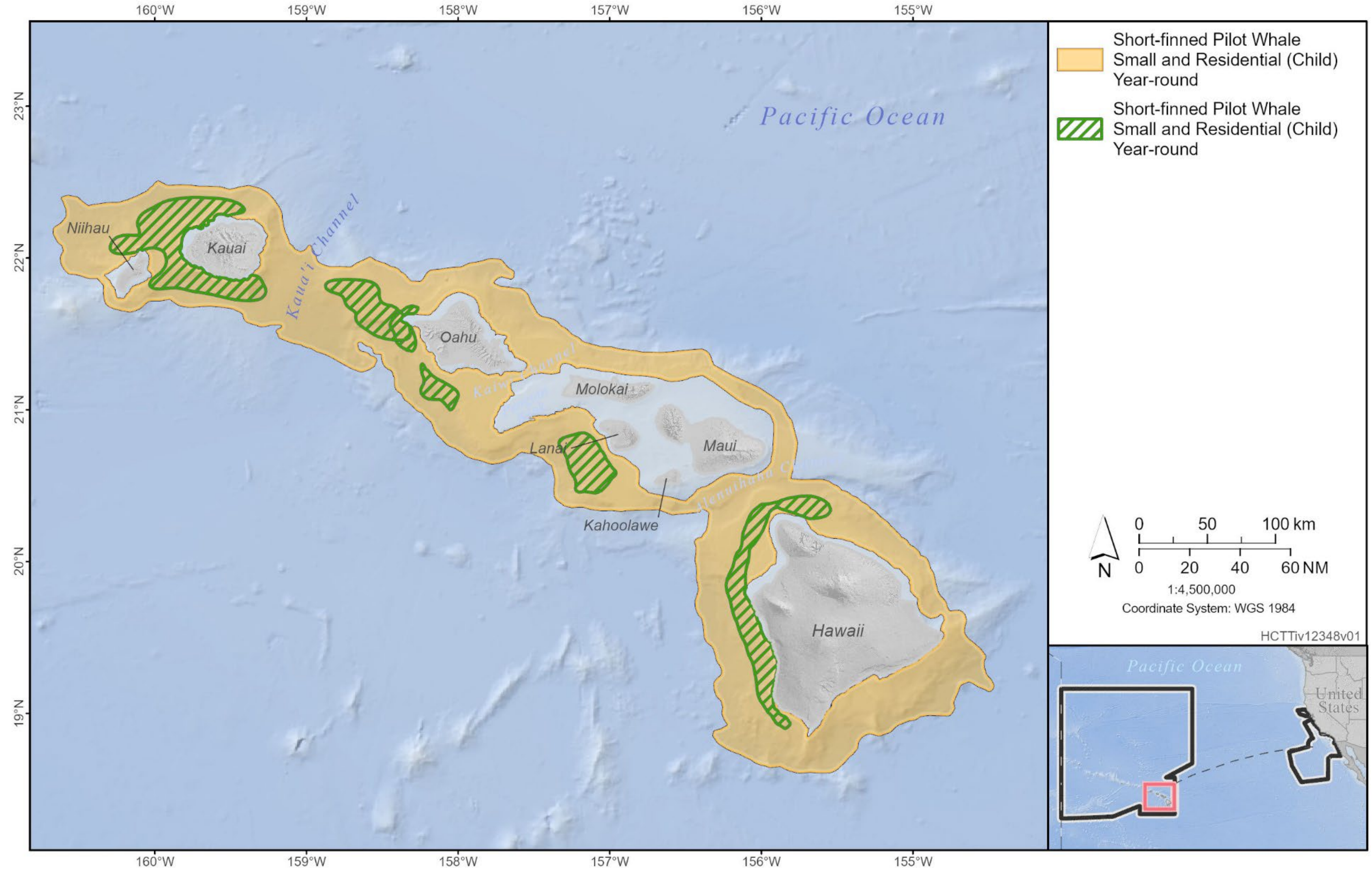


Figure C-24: Short-Finned Pilot Whale Small and Resident BIAs in the Hawaii Study Area

Short-finned pilot whale distribution off Southern California changed dramatically after El Niño in 1982–1983, when squid did not spawn as usual in the area, and pilot whales virtually disappeared from the area for 9 years (Shane, 1995). There have been nine short-finned pilot whale sightings during seven systematic ship surveys conducted by NMFS off California, Oregon, and Washington between 1991 and 2014, with three of these off southern California (Barlow, 2016; Barlow & Forney, 2007). During a NMFS 2018 survey of waters off the U.S. west coast and the Baja California Peninsula, Mexico, there was one short-finned pilot whale sighting at about 30°N (Henry et al., 2020). There were two additional short-finned pilot whale sightings during 16 ship surveys conducted in the Southern California Bight between 2004 and 2008 (Douglas et al., 2014b). Short-finned pilot whales were not sighted during 18 aerial surveys conducted in the Southern California Bight between 2008 and 2013 (Jefferson et al., 2014). A group of approximately 50 individuals was encountered off San Diego in May 2015 and included an individual photo identified previously off Ensenada, Mexico (Kendall-Bar et al., 2016).

C.6.3.10.3 Population Trends

For Hawaiian waters, the available data preclude an assessment of population trends for short-finned pilot whales (Carretta et al., 2023b).

Pilot whales appeared to have returned to California waters as evidenced by an increase in sighting records, as well as incidental fishery bycatches (Barlow, 2016; Barlow & Forney, 2007; Douglas et al., 2014a). Because these changes likely reflect a change in distribution based on a changing environment rather than a change in the population, there can be no assessment of the current population trend for short-finned pilot whales in California (Carretta et al., 2023b).

C.6.3.10.4 Population Threats

Short-finned pilot whales are susceptible to entanglement in fishing gear and marine debris. The most recent monitoring data from 2010 through 2014 indicate that the estimate of mortality and serious injury for this species in the California drift gillnet fishery is 6 individuals, or 1.2 whales annually off the U.S. West Coast. Short-finned pilot whales were also historically observed entangled in the Southern California squid purse seine fishery; however, interactions are now less common due to the decline of the presence of this species in the region.

In Hawaii, short-finned pilot whales are threatened by the deep-set longline fishery and the shallow-set longline fishery. The most recent monitoring data from 2017 through 2021 indicate that there have been no observed entanglements or hookings of this species in the shallow-set longline fishery; however, there was one recorded entanglement of a short-finned pilot whale in the deep-set longline fishery, which resulted in serious injury (Bradford, 2018a; Bradford, 2018b; Bradford, 2021; Bradford et al., 2017; Bradford et al., 2021; Carretta et al., 2023b; McCracken, 2019). Additionally, several stranded short-finned pilot whales have been found off Hawaii with large amounts of marine debris from fisheries, such as fishing nets and fishing lines in their stomachs (Carretta et al., 2023b).

Anthropogenic noise such as sonar may result in unknown levels of injury or mortality to short-finned pilot whales (Carretta et al., 2023b). In 2005, there was an UME off the U.S. East Coast that resulted in a stranding of 33 short-finned pilot whales and a minke whale (Hohn et al., 2006). These strandings may be associated with military sonar use that occurred in theses area during the time of the strandings; however, a definitive association has not been confirmed (Hohn et al., 2006).

Climate change has increasingly become a threat to marine mammals, including toothed whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.11 Melon-headed Whale (*Peponocephala electra*)

C.6.3.11.1 Status and Management

The melon-headed whale is protected under the MMPA and is not listed under the ESA. For the MMPA stock assessment reports, there are two Pacific management stocks within the Hawaiian Islands EEZ based on photo-identification, social network analysis, movement data, and genetics (Oleson et al., 2013). These stock are (1) the Kohala Resident stock, which includes melon-headed whales off the Kohala and west coast of Hawaii Island in waters less than 2,500 m deep; and (2) the Hawaiian Islands stock, which includes melon-headed whales inhabiting waters throughout the U.S. EEZ of the Hawaiian Islands (Aschettino et al., 2012; Baird et al., 2015d; Carretta et al., 2017b; Carretta et al., 2023b; Oleson et al., 2013).

C.6.3.11.2 Habitat and Geographic Range

Melon-headed whales are found worldwide in tropical and subtropical waters but movement patterns for this species are poorly understood. They have occasionally been reported at higher latitudes, but these movements are considered to be beyond their normal range, because the records indicate these movements occurred during incursions of warm water currents (Jefferson et al., 2015; Perryman, 2008; Perryman et al., 1994). In the north Pacific, occurrence of this species is well known in deep waters off many areas, including the Hawaii Study Area (Aschettino et al., 2012; Au & Perryman, 1985; Ferguson, 2005; Perrin, 1976; Wang et al., 2001).

The melon-headed whale is regularly found within Hawaiian waters (Baird et al., 2010; Baird et al., 2015e; Baird et al., 2003a; Baird et al., 2003b; Mobley et al., 2000; Shallenberger, 1981). Large groups are seen regularly, especially off the Waianae coast of Oahu, the north Kohala coast of Hawaii, and the leeward coast of Lanai (Baird, 2006; Oleson et al., 2013; Shallenberger, 1981). There was a total of nine sightings of melon-headed whales during three systematic ship surveys of the Hawaiian Islands EEZ in 2002 (one sighting), 2010 (one sighting), and 2017 (seven sightings) (Bradford et al., 2021). The single sightings in 2002 and 2010 included groups of 89 (Baird, 2006) and 153 melon-headed whales (Bradford et al., 2013), respectively, and the mean group size in 2017 was 187.9 animals (Bradford et al., 2021).

Brownell et al. (2009) found that melon-headed whales near oceanic islands rest near shore during the day, and feed in deeper waters at night. Melon-headed whales are known to enter shallow water areas on occasion although these are generally characterized as animals being “out of habitat” and/or “mass strandings”; a few hundred did so at Hanalei Bay, Kauai and Sasanhaya Bay, Rota (Mariana Islands) on July 4, 2004 (Jefferson et al., 2006), and similar numbers did so in the Philippines entering Manila Bay in Feb 2009 and the bay at Odiongan, Romblon in March of 2009 (Aragones et al., 2010). In surveys around the main Hawaiian Islands, melon-headed whales showed no clear pattern in depth use (Baird et al., 2013b).

A year-round, non-hierarchical Small and Resident Population BIA has been delineated for melon-headed whales off the Island of Hawaii (Kratofil et al., 2023) (Figure C-25). The BIA was updated from the original BIA (Baird et al., 2015d) based on additional analyses and encompasses 3,816 km^2 off the northwest coast of the Island of Hawaii (Kratofil et al., 2023).

During ship-based bird surveys in the eastern tropical Pacific, melon-headed whales were observed from the U.S.-Mexico border south to Peru, typically associated with pelagic sea birds while foraging (Pitman & Ballance, 1992). The species is not expected to be present in the California Study Area.

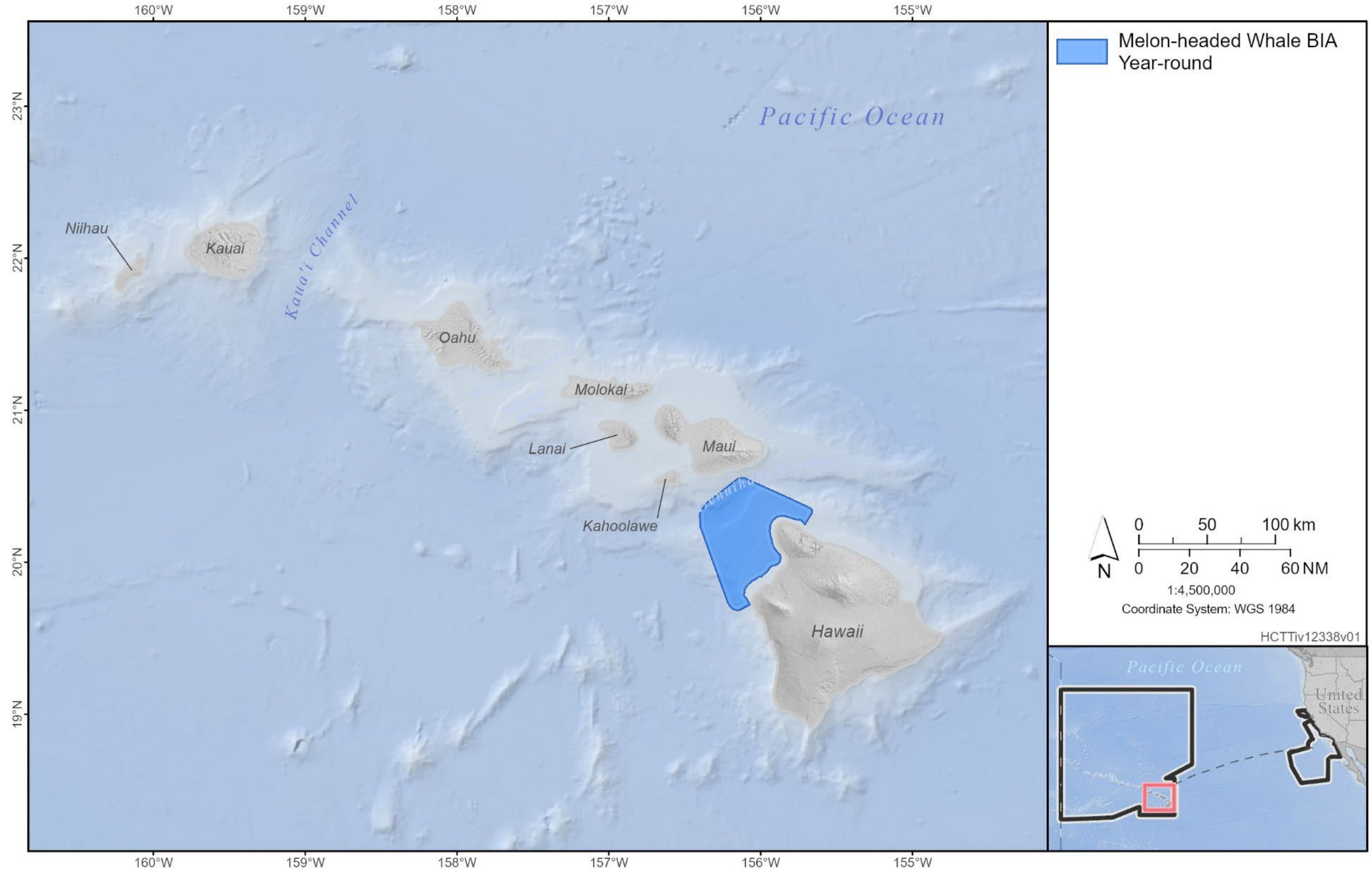


Figure C-25: Melon-Headed Whale Small and Resident BIA in the Hawaii Study Area

C.6.3.11.3 Population Trends

Population data are currently insufficient to identify trends for the Kohala resident stock of melon-headed whales (Carretta et al., 2023b). Current abundance estimates for the Hawaiian Islands stock of melon-headed whales have broad and overlapping confidence intervals, and thus preclude a robust trend analysis. In addition, given the typically large group sizes observed in the field, the addition of a single sighting in a given year can have a substantial effect on the resulting abundance estimate, making it even more challenging to conduct a robust trend assessment (Carretta et al., 2023b) .

C.6.3.11.4 Population Threats

In Hawaii, melon-headed whales are threatened by the deep-set longline fishery and the shallow-set longline fishery. The most recent monitoring data from 2014 through 2018 indicate that there have been no observed entanglements or hookings of this species in these fisheries (Bradford, 2018a; Bradford, 2018b; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019). Additionally, there are no reported interactions of melon-headed whales and nearshore fisheries in Hawaii; however, nearshore fisheries are not monitored for protected species bycatch.

Anthropogenic noise such as sonar may result in unknown levels of injury or mortality to melon headed whales (Carretta et al., 2023b; Celi et al., 2013; Hohn et al., 2006). In 2004, a mass stranding of 150–200 melon-headed whales occurred in Kauai during the same timeframe as a multi-national military exercise using sonar (Hohn et al., 2006). The use of sonar in this instance is considered to be a plausible cause of this stranding event.

Climate change has increasingly become a threat to marine mammals. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.12 Long-beaked Common Dolphin (*Delphinus delphis bairdii*)

C.6.3.12.1 Status and Management

The Society for Marine Mammalogy's Committee on Taxonomy and NMFS currently recognize all common dolphins as a single species, *D. delphis*. Long-and short-beaked common dolphins are still recognized as separate subspecies, *D. delphis bairdii* and *D. delphis delphis*, respectively. In the future it is possible that they will again be recognized as separate species, but additional taxonomic analyses are required.

This species is protected under the MMPA and is not listed under the ESA. For the MMPA stock assessment reports, there is a single Pacific management stock for those animals found within the U.S. EEZ off the U.S. west coast, which is called the California stock (Carretta et al., 2023b).

C.6.3.12.2 Habitat and Geographic Range

Long-beaked common dolphins are not present in the Hawaii Study Area.

The long-beaked common dolphin appears to be restricted to waters relatively close to shore (Jefferson & Van Waerebeek, 2002; Perrin, 2008a), apparently preferring shallower and warmer water than the short-beaked common dolphin (Becker et al., 2016; Becker et al., 2022a; Gerrodette & Eguchi, 2011; Perrin, 2008a). Off California and Baja California, Mexico, long-beaked common dolphins are commonly found within 50 NM of the coast (Becker et al., 2022a; Carretta et al., 2011; Gerrodette & Eguchi, 2011). This species is found off Southern California year-round, but it may be more abundant there during the warm-water months (May to October) (Barlow & Forney, 2007; Bearzi, 2005b; Douglas et al., 2014b; Henderson et al., 2014; Heyning & Perrin, 1994). Stranding data, sighting records, and habitat-based density models suggest that this species' abundance fluctuates seasonally and from year to year off California (Becker et al., 2020; Carretta et al., 2011; Douglas et al., 2014b; Henderson et al., 2014). Southern California waters represent the northern limit to this species' range and the seasonal and inter-annual changes in abundance off California are assumed to reflect the shifts in the movements of animals between U.S. and Mexican waters (Becker et al., 2022a).

C.6.3.12.3 Population Trends

There appears to be an increasing trend in the abundance of long-beaked common dolphin in California waters over the last 30 years based on both design- and model-based analyses, but a robust trend analysis has not yet been conducted (Barlow, 2016; Becker et al., 2020; Carretta et al., 2017b; Carretta et al., 2023b).

C.6.3.12.4 Population Threats

Long-beaked common dolphins are susceptible to entanglement in fishing gear. Most recent monitoring data from 2015 through 2019 indicate that the estimate of human-caused mortality or serious injury of this species in the California drift gillnet fishery is 1.7 individuals annually (Carretta, 2021). Additionally, stranding data along the U.S. West Coast during 2015 through 2019 yields a minimum estimate of 84 fishery-related mortalities of long-beaked common dolphins (Carretta, 2023). Mortalities from hook and line fisheries during the same time period are estimated as 3.2 individuals annually (Carretta, 2023).

Climate change has increasingly become a threat to marine mammals, including dolphins. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021). There have been reports dating to 2007 of long-beaked common dolphin mortalities due to domoic acid toxicity, which is likely related to ocean acidification and increased CO_2 levels.

C.6.3.13 Short-beaked Common Dolphin (*Delphinus delphis delphis*)

C.6.3.13.1 Status and Management

The Society for Marine Mammalogy's Committee on Taxonomy and NMFS currently recognize all common dolphins as a single species, *D. delphis*. Long-and short-beaked common dolphins are still recognized as separate subspecies, *D. delphis bairdii* and *D. delphis delphis*, respectively. In the future it is possible that they will again be recognized as separate species, but additional taxonomic analyses are required.

This species is protected under the MMPA and is not listed under the ESA. For the MMPA stock assessment reports, there is a single Pacific management stock for those animals found within the U.S. EEZ off the U.S. west coast, which is called the California, Oregon, and Washington stock (Carretta et al., 2023b).

C.6.3.13.2 Habitat and Geographic Range

Short-beaked common dolphins are not present in the Hawaii Study Area.

Historically along the U.S. west coast, short-beaked common dolphins were sighted primarily south of Point Conception (Dohl et al., 1983), but now they are commonly encountered as far north as 42°N (Hamilton et al., 2009), and occasionally as far north as 48°N (Forney, 2007). Seasonal distribution shifts are pronounced, with a significant southerly shift south of Point Arguello in the winter (Becker et al., 2014; Campbell et al., 2014; Forney & Barlow, 1998b; Henderson et al., 2014). Short-beaked common dolphins are a warm temperate to tropical species, and based on habitat models developed using line-transect survey data collected off the U.S. west coast from 1991 to 2018, densities are greatest when waters are warmest (Barlow et al., 2009b; Becker et al., 2010; Becker et al., 2016; Becker et al., 2014; Becker et al., 2018; Forney & Barlow, 1998b; Forney et al., 2012). The distribution of short-beaked common dolphins off the U.S. west coast exhibits substantial seasonal and annual variability due to changes in oceanographic conditions, often resulting in shifts both north-south and inshore-offshore (Barlow et al., 2009b; Becker et al., 2020; Becker et al., 2016; Becker et al., 2014; Becker et al., 2018; Becker et al., 2017; Forney & Barlow, 1998b; Forney et al., 2012; Henderson et al., 2014). Short-beaked common dolphin abundance off California has increased dramatically since the late 1970s, along with a smaller decrease in abundance in the eastern tropical Pacific, suggesting a large-scale northward shift in the distribution of this species in the eastern North Pacific (Carretta et al., 2017b; Forney & Barlow, 1998b; Forney et al., 1995). The trend for an increase in the population off California appears to be continuing given current data from the most recent 2014 and 2018 NMFS surveys (Barlow, 2016; Becker et al., 2020).

Short-beaked common dolphins are found in the California Study Area throughout the year, distributed between the coast and at least 345 mi. from shore (Barlow, 2016; Barlow & Forney, 2007; Forney & Barlow, 1998b). Based on multiple line-transect studies conducted by NMFS, the short-beaked common dolphin is the most abundant cetacean species with a widespread distribution off southern California (Barlow, 2016; Barlow & Forney, 2007; Becker et al., 2020; Becker et al., 2017; Campbell et al., 2014; Carretta et al., 2011; Douglas et al., 2014b; Forney et al., 1995). From 2004 to 2008 during ship surveys conducted quarterly off southern California, short-beaked common dolphins were encountered year-round, with highest encounters during the summer (Douglas et al., 2014b). From 2008 to 2013 during 18 aerial surveys conducted in the Southern California Bight, short-beaked common dolphins were the most-frequently observed species (Jefferson et al., 2014).

C.6.3.13.3 Population Trends

Based on an analysis of sighting data collected during quarterly surveys off southern California from 2004 to 2013, short-beaked common dolphins showed annual variations in density, but there was no significant trend evident during the period of this study (Campbell et al., 2014). However, Barlow (2016) noted a nearly monotonic increase in the abundance of short-beaked common dolphins from 1991 to 2014 off the U.S. west coast. Predictions from habitat-based density models indicate a similar increase in short-beaked dolphin abundance off the U.S. west coast from 1996 to 2018, with the most current (2018) abundance estimate of 1,056,308 dolphins (CV = 0.207) (Barlow, 2016; Becker et al., 2020; Carretta et al., 2023b). However, the increase in short-beaked dolphin abundance could be due to a northward movement of animals from waters off Mexico and a future trend analysis is required to better assess potential population trends (Barlow, 2016; Becker et al., 2022a).

C.6.3.13.4 Population Threats

Short-beaked common dolphins are susceptible to entanglement in fishing gear. Most recent monitoring data from 2015 through 2019 indicate that the estimate of human-caused mortality or serious injury of this species in the California drift gillnet fishery is an average of 26.7 individuals annually (Carretta, 2021). Additionally, the estimated mean annual bycatch of short-beaked common dolphins in the set gillnet fisheries was 16 individuals, or 3.8 individuals annually, from 2015 through 2019 (Carretta, 2023).

Similar to long-beaked common dolphins, they are also susceptible to mortalities or injuries from hook and line fisheries (Carretta, 2023).

Climate change has increasingly become a threat to marine mammals, including dolphins. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.14 Common Bottlenose Dolphin (*Tursiops truncatus*)

C.6.3.14.1 Status and Management

The common bottlenose dolphin is protected under the MMPA and is not listed under the ESA. For the MMPA stock assessment reports, bottlenose dolphins within the Pacific U.S. EEZ are divided into seven stocks: (1) Kauai and Niihau, (2) Oahu, (3) Maui Nui, (4) Hawaii Island, (5) the Hawaii Pelagic stock, (6) California Coastal stock, and (7) the California, Oregon and Washington Offshore stock (Carretta et al., 2024).

C.6.3.14.2 Habitat and Geographic Range

Common bottlenose dolphins typically are found in coastal and continental shelf waters of tropical and temperate regions of the world (Jefferson et al., 2008; Wells et al., 2009). Common bottlenose dolphins occur throughout the Hawaiian Islands, and they are typically observed throughout the main islands and from the Island of Hawaii to Kure Atoll (Baird et al., 2013c; Shallenberger, 1981). In the Hawaiian Islands, this species is found in both shallow coastal waters and deep offshore waters (Baird et al., 2003b; Barlow et al., 2008; Bradford et al., 2013; Mobley et al., 2000). The offshore variety is typically larger than the inshore. Photo-identification and genetics indicate the presence of island associated populations of bottlenose dolphins in the Hawaiian Islands (Martien et al., 2012). During three systematic surveys of the Hawaiian Islands EEZ in 2002, 2010, and 2017, there were a total of 38 sightings of bottlenose dolphins, of which 27 groups were identified as members of the Hawaii pelagic stock and the rest identified as members of one of the four island-associated stocks (Bradford et al., 2021). Habitat-based models developed from systematic ship survey data collected in the central North Pacific show some of the highest common bottlenose dolphin densities around the Hawaiian Islands (Becker et al., 2012b; Forney et al., 2015). More recently, habitat-based density models were developed using systematic survey data collected within waters of the Hawaiian Islands EEZ from 2000 to 2020 using only those common bottlenose dolphin sightings identified as members of the Hawaii pelagic stock (Becker et al., 2022a). Model predictions showed a strong island association for the pelagic stock, with highest densities occurring near all the islands within the EEZ.

Five year-round Small and Resident Population BIAs (a parent BIA, three child BIAs, and one non-hierarchical BIA) have been delineated in the main Hawaiian Islands for the populations of common bottlenose dolphins (Kratofil et al., 2023) (Figure C-26). The BIAs were updated from the original BIAs (Baird et al., 2015d) based on additional analyses. The parent BIA encompasses 36,634 km^2 of waters surrounding Niihau to the west and extending east to surround the island of Maui. The child BIAs encompass 2,772 km^2 around Kauai/Niihau, 8,487 km^2 around Oahu, and 10,622 km^2 around Maui Nui. In addition, a year-round non-hierarchical Small and Resident Population BIA was delineated and encompasses 8,299 km^2 around the Island of Hawaii (Kratofil et al., 2023) (Figure C-26).

Common bottlenose dolphins are known to occur year-round in both coastal and offshore waters of Monterey Bay, Santa Monica Bay, San Diego Bay, and SCI, California (Bearzi, 2005a, 2005b; Bearzi et al., 2009b; Carretta et al., 2000; Henkel & Harvey, 2008). In the California Study Area, they are routinely encountered in San Diego Bay in transit to the waters off Coronado where they feed (Graham & Saunders, 2015).

During surveys off California, offshore common bottlenose dolphins were generally found at distances greater than 1.9 mi. from the coast and throughout the waters of Southern California (Barlow, 2016; Barlow & Forney, 2007; Bearzi et al., 2009b; Hamilton et al., 2009). Sighting records off California and Baja California suggest a continuous distribution of offshore common bottlenose dolphins in these regions (Mangels & Gerrodette, 1994). Based on habitat models derived from line-transect survey data collected between 1991 and 2009 off the U.S. west coast, offshore common bottlenose dolphins exhibit a disjunctive longitudinal distribution, suggesting that there may be two separate populations in this area, although additional genetic data are required for confirmation (Becker et al., 2016). This pattern was also apparent in the density predictions from more recent models that included additional survey data collected in 2014 and 2018 (Becker et al., 2020).

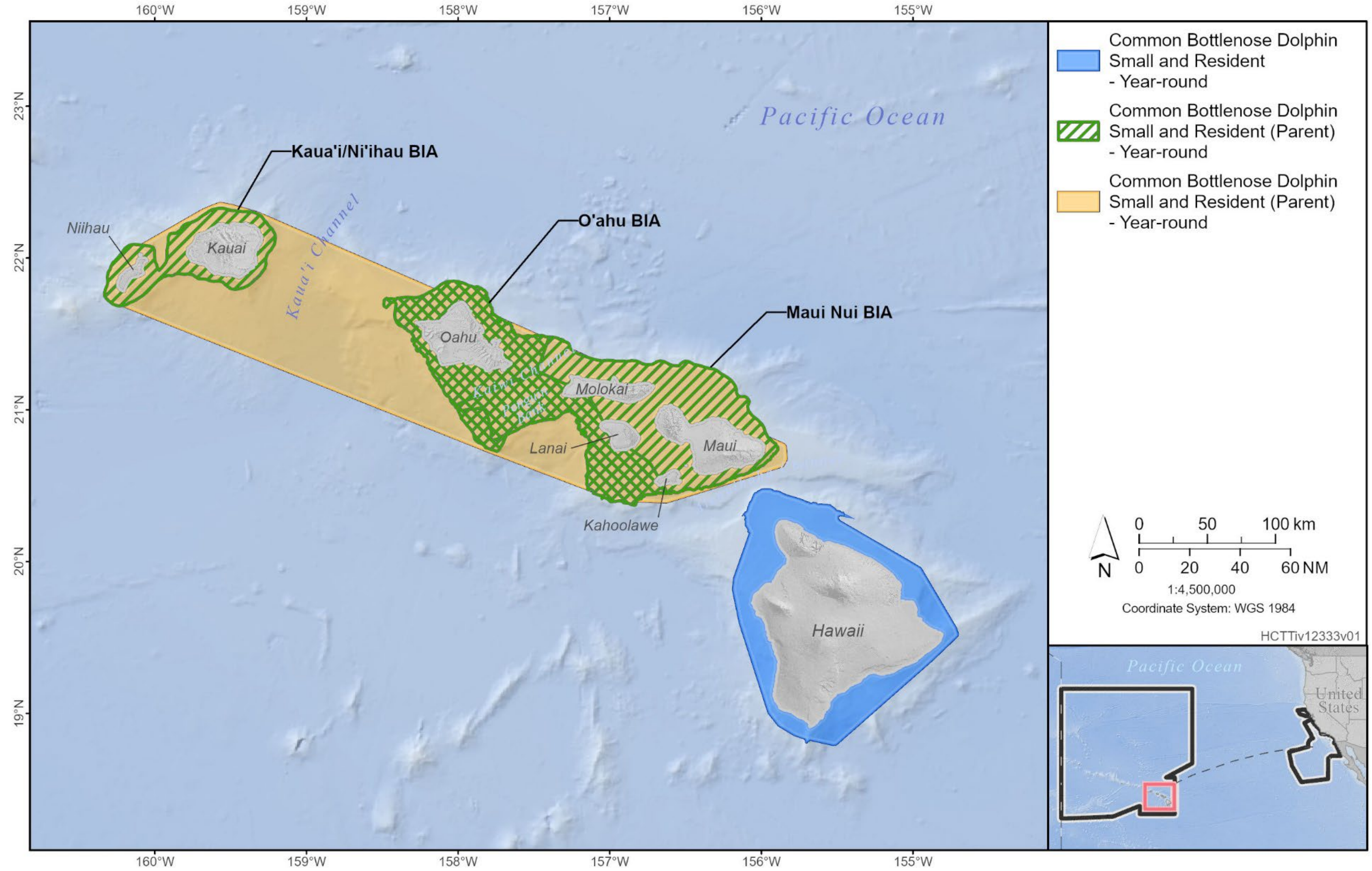


Figure C-26: Common Bottlenose Dolphin Small and Resident BIAs in the Hawaii Study Area

California coastal bottlenose dolphins are found within about 0.6 NM of the shore, generally from Point Conception to as far south as San Quintin, Mexico (Carretta et al., 1998; Defran & Weller, 1999; Hwang et al., 2014). Coastal common bottlenose dolphins also have been consistently sighted off central California and as far north as San Francisco since the 1982–83 El Niño, when they apparently traveled further north tracking prey due to the northern extent of warmer waters and continued using those more northern waters after that El Niño had ended (Hwang et al., 2014). Off Southern California, animals are found within 500 m of the shoreline 99 percent of the time and within 250 m of the shoreline 90 percent of the time (Hanson & Defran, 1993; Hwang et al., 2014). The dolphins in the nearshore waters of San Diego, California differ somewhat from other coastal populations of this species in distribution, site fidelity, and school size (Bearzi, 2005a, 2005b; Carretta et al., 2017b; Defran et al., 2015; Defran & Weller, 1999). Photo identification analyses suggest that there may be two separate stocks of coastal bottlenose dolphins that exhibit limited integration, a California Coastal stock and a Northern Baja California stock (Defran et al. 2015), but this is not yet reflected in the Pacific Stock Assessment Report (Carretta et al., 2023b). The results from relatively contemporaneous surveys at Ensenada, San Diego, Santa Monica Bay, and Santa Barbara between 1996 and 2001 provided samples of the speed and distances individual coastal bottlenose dolphins routinely traveled (Hwang et al., 2014). The minimum travel speed observed was 53 km per day and the maximum was 95 km per day; and the total distances traveled between points was between 104 km and 965 km (Hwang et al., 2014).

C.6.3.14.3 Population Trends

For the Hawaii Pelagic stock, the available abundance estimates have broad and overlapping confidence intervals, thus precluding an assessment of population trends (Carretta et al., 2017b; Carretta et al., 2023b). For the four island-associated insular stocks (Kauai and Niihau, Oahu, Maui Nui, and Hawaii Island), there is currently insufficient information to assess population trends for these stocks (Carretta et al., 2023b).

For the California Study Area, the California Coastal stock population size has remained stable from 1987 to 2005, but more recent abundance estimates suggest the population may be increasing (Carretta et al., 2017b; Carretta et al., 2023b; Dudzik et al., 2006; Weller et al., 2016). For the California, Oregon and Washington Offshore stock, an apparent trend is not evident from either design- or model-based abundance estimates, but a robust trend analysis has not been conducted for the population (Barlow, 2016; Becker et al., 2020; Carretta et al., 2017b; Carretta et al., 2023b).

C.6.3.14.4 Population Threats

Common bottlenose dolphins are susceptible to interactions with fisheries and entanglements in fishing gear. Off the U.S. West Coast, bottlenose dolphins have been observed entangled in both coastal and offshore fisheries. The minimal annual takes for this species across commercial fisheries are estimated as 1.6 dolphins per year (Carretta, 2023).

In Hawaii, common bottlenose dolphins are threatened by the deep-set longline fishery and the shallow-set longline fishery. The most recent monitoring data from 2017 through 2021 indicate that there have 1 observed entanglement or hooking of this species in the shallow-set longline fishery and 6 taken in the deep-set longline fishery (Bradford, 2018a; Bradford, 2018b; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019). During the same five-year period, there was observed shooting of a bottlenose dolphin, likely attributed to fisheries interactions as well. There are no estimates of mortality or serious injury of this species in Hawaii nearshore gillnet or hook and line fisheries because nearshore fisheries are not monitored for protected species bycatch (Bradford & Lyman, 2018; Carretta, 2023). However, in areas where gillnet fishing is allowed, fishermen are often required to frequently check their nets for bycatch (Carretta et al., 2024). Although these fisheries are not monitored, there have been observations of bottlenose dolphins with serious injuries from hooks likely from nearshore fisheries (Carretta et al., 2024).

Furthermore, this species is known to steal bait and catches off commercial fishing lines in Hawaii, which may cause serious injury. Photo observations of bottle nose dolphins in the Maui Nui region revealed that approximately 27 percent of well-marked, adult individuals had scarring likely attributed to hookings and interactions with fishing gear (Carretta et al., 2024; Machernis et al., 2021).

Climate change has increasingly become a threat to marine mammals, including dolphins. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.15 Pantropical Spotted Dolphin (*Stenella attenuata*)

C.6.3.15.1 Status and Management

The species is protected under the MMPA and is not listed under the ESA. For the MMPA stock assessment reports, the species has been divided into four stocks based on genetics and the frequency of sightings in pelagic waters around Hawaii (Courbis et al., 2014; Oleson et al., 2013). The four management stocks within the Hawaiian Islands EEZ are (1) the Oahu stock, which includes spotted dolphins within 20 km of Oahu; (2) the Maui Nui stock, which includes spotted dolphins within 20 km of the island group formed by Maui, Molokai, Lanai, and Kahoolawe and their adjacent waters; (3) the Hawaii Island stock, which includes spotted dolphins found within 65 km from Hawaii Island; and (4) the Hawaii Pelagic stock, which includes spotted dolphins inhabiting the waters throughout the Hawaiian Islands EEZ, outside of the insular stock areas (Carretta et al., 2024).

C.6.3.15.2 Habitat and Geographic Range

The pantropical spotted dolphin is distributed in offshore tropical and subtropical waters of the Pacific, Atlantic, and Indian Oceans between about 40° N and 40° S (Baldwin et al., 1999; Perrin, 2008b). The species is much more abundant in the lower latitudes of its range. It is found mostly in deeper offshore waters but does approach the coast in some areas (Jefferson et al., 2008; Perrin, 2001).

Based on sightings during small boat surveys from 2000 to 2012 in the main Hawaiian Islands, pantropical spotted dolphins were the most abundant species of cetacean, although they were frequently observed leaping out of the water which likely increased their detectability (Baird et al., 2013c). This species was also one of the most abundant based on analyses of line-transect data collected in the Hawaiian EEZ in 2002, 2010, and 2017, with a total of 39 sightings during the three surveys (Barlow, 2006; Bradford et al., 2013; Bradford et al., 2021). Known habitat preferences and sighting data indicate the primary occurrence for the pantropical spotted dolphin in Hawaiian waters is shallow coastal waters to depths of 5,000 m, although the peak sighting rates occur in depths from 1,500 to 3,500 m (Baird et al., 2013d; Bradford et al., 2013; Oleson et al., 2013). Habitat-based models developed from systematic ship survey data collected in the central North Pacific show relatively high pantropical spotted dolphin densities around the Hawaiian Islands, particularly around the Main Hawaiian Islands (Becker et al., 2012a; Forney et al., 2015). More recently, sighting data from systematic ship surveys conducted in waters of the Hawaiian Islands EEZ from 2000 to 2020 allowed for the development of separate habitat models for the pelagic and combined insular stocks of pantropical spotted dolphins (Becker et al., 2022a). Consistent with past observations (Baird et al., 2013d; Bradford et al., 2013;

Oleson et al., 2013), the model for the combined insular stocks showed peak abundance in depths from 1,500 to 3,500 m. The habitat model for the pelagic stock predicted low to mid-range density estimates for offshore waters of the Hawaiian Islands EEZ, with highest densities near all the islands, but particularly around the Main Hawaiian Islands (Becker et al., 2022a).

A year-round Small and Resident Population parent BIA and three child BIAs have been delineated in the main Hawaiian Islands for the populations of Pantropical spotted dolphins (Kratofil et al., 2023) (Figure C-27). The BIAs were updated from the original BIAs (Baird et al., 2015d) based on additional analyses. The parent BIA encompasses 57,711 km^2 of waters surrounding Oahu, Maui Nui, and the Island of Hawaii. The child BIAs encompass 12,952 km^2 around Oahu, 6,743 km^2 around Maui Nui, and 10,768 km^2 around the Island of Hawaii (Kratofil et al., 2023).

Pantropical spotted dolphins are not present in the California Study Area, but they do occur in waters off the Baja California Peninsula, Mexico (Hamilton et al., 2009). Dolphins occurring in Mexican waters are considered part of an Eastern Tropical Pacific population that occurs in high numbers in tropical oceanic waters north of the equator and near the coast off Central America (Forney et al., 2012; Wade & Gerrodette, 1993).

C.6.3.15.3 Population Trends

No data are available on current population trends for any of the insular stocks of pantropical spotted dolphins in Hawaii (Carretta et al., 2023b). Abundance estimates for the pelagic stock have broad and overlapping confidence intervals, thus precluding a robust assessment of population trends.

C.6.3.15.4 Population Threats

Common bottlenose dolphins are susceptible to interactions with fisheries and entanglements in fishing gear. Off the U.S. West Coast, bottlenose dolphins have been observed entangled in both coastal and offshore fisheries. The minimal annual takes for this species across commercial fisheries are estimated as 1.6 dolphins per year (Carretta, 2023).

In Hawaii, common bottlenose dolphins are threatened by the deep-set longline fishery and the shallow-set longline fishery. The most recent monitoring data from 2017 through 2021 indicate that there have 1 observed entanglement or hooking of this species in the shallow-set longline fishery and 6 taken in the deep-set longline fishery (Bradford, 2018a; Bradford, 2018b; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019). During the same five-year period, there was observed shooting of a bottlenose dolphin, likely attributed to fisheries interactions as well. There are no estimates of mortality or serious injury of this species in Hawaii nearshore gillnet or hook and line fisheries because nearshore fisheries are not monitored for protected species bycatch (Bradford & Lyman, 2018; Carretta, 2023). However, in areas where gillnet fishing is allowed, fishermen are often required to frequently check their nets for bycatch (Carretta et al., 2024). Although these fisheries are not monitored, there have been observations of bottlenose dolphins with serious injuries from hooks likely from nearshore fisheries (Carretta et al., 2024).

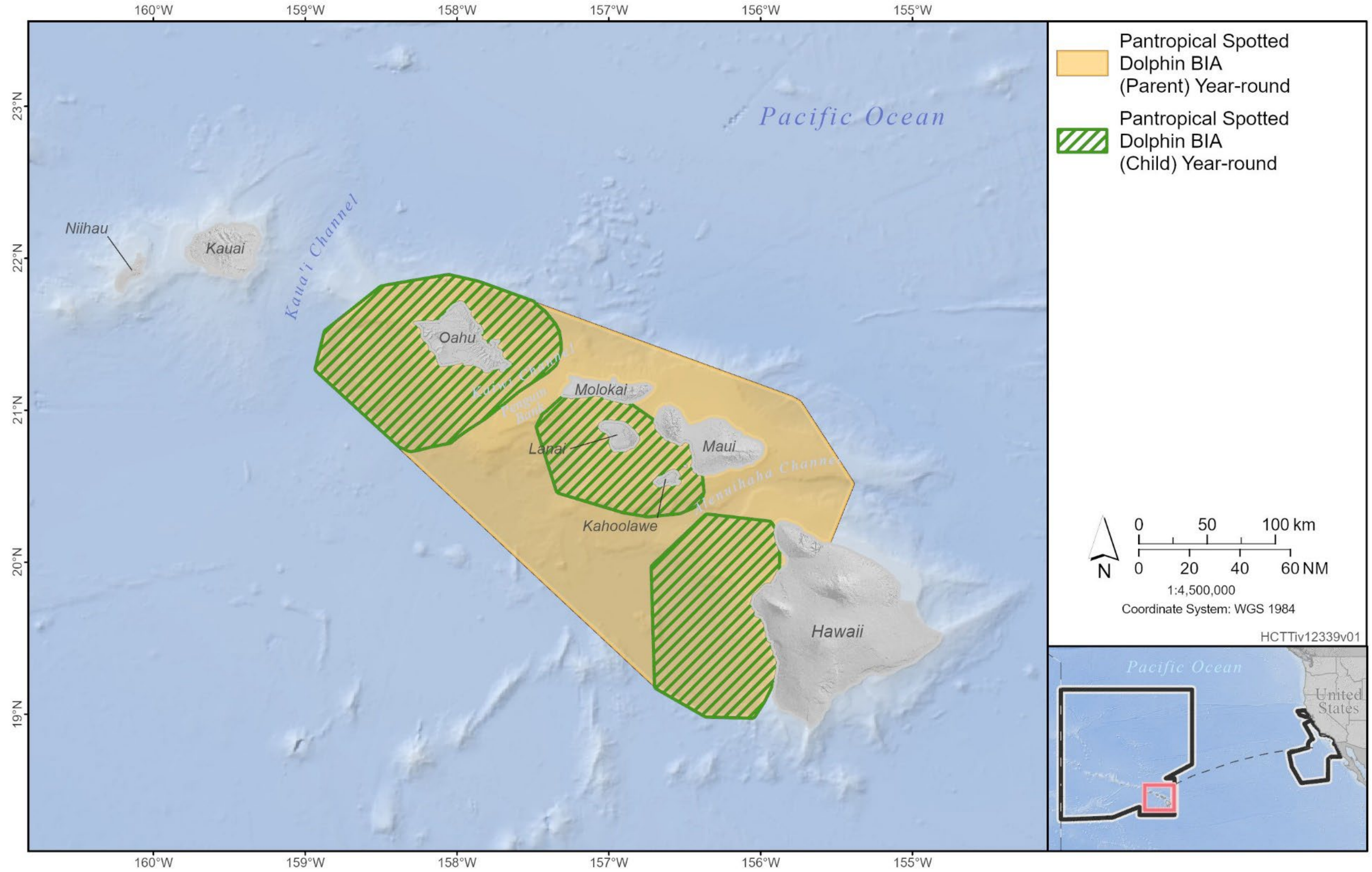


Figure C-27: Pantropical Spotted Dolphin Small and Resident BIAs in the Hawaii Study Area

Furthermore, this species is known to steal bait and catches off commercial fishing lines in Hawaii, which may cause serious injury. Photo observations of bottle nose dolphins in the Maui Nui region revealed that approximately 27 percent of well-marked, adult individuals had scarring likely attributed to hookings and interactions with fishing gear (Carretta et al., 2024; Machernis et al., 2021).

Climate change has increasingly become a threat to marine mammals, including dolphins. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.16 Striped Dolphin (Stenella coeruleoalba)

C.6.3.16.1 Status and Management

This species is protected under the MMPA and is not listed under the ESA. In the eastern north Pacific, NMFS identifies two striped dolphin management stocks within the U.S. EEZ: the Hawaiian stock and the California, Oregon, and Washington stock (Carretta et al., 2023b).

C.6.3.16.2 Habitat and Geographic Range

Although primarily a warm-water species, the range of the striped dolphin extends higher into temperate regions than those of any other species in the genus *Stenella (Jefferson et al., 2015)*. Striped dolphins are generally restricted to oceanic regions and are seen close to shore only where deep water approaches the coast. In some areas (e.g., the eastern tropical Pacific), they are mostly associated with convergence zones and regions of upwelling (Au & Perryman, 1985; Reilly, 1990). The northern limits are the Sea of Japan, Hokkaido, Washington State, and along roughly 40° N across the western and central Pacific (Reeves et al., 2002). In the eastern tropical Pacific, striped dolphins inhabit areas with large seasonal changes in surface temperature and thermocline depth, as well as seasonal upwelling (Au & Perryman, 1985; Reilly, 1990). In some areas, this species appears to avoid waters with sea temperatures less than 68°F (20°C) (Van Waerebeek et al., 1998).

The striped dolphin regularly occurs around the Hawaiian Islands. During three systematic ship surveys of waters within the Hawaiian Islands EEZ in summer/fall of 2002, 2010, and 2017, there was a total of 45 sightings of striped dolphin, with the yearly mean group sizes ranging from 36.3 to 54.9 animals (Bradford et al., 2021). Resulting density estimates from these surveys suggest that they are one of the most abundant species in the Hawaiian EEZ. Based on sighting records, this species occurs primarily seaward of the 1,000 m depth contour. Striped dolphins are occasionally sighted closer to shore in Hawaii, so an area of secondary occurrence is expected from a depth range of 100 to 1,000 m. Occurrence patterns are assumed to be the same throughout the year (Mobley et al., 2000). Habitat-based density models developed from systematic ship survey data collected in the central North Pacific show more uniform striped dolphin densities throughout the Hawaiian EEZ, consistent with this species' known occurrence in deep waters (Becker et al., 2012b; Forney et al., 2015). More recently, sighting data from systematic ship surveys conducted within waters of the Hawaiian Islands EEZ from 2000 to 2020 supported the development of an updated habitat-based density model for striped dolphin and confirmed this species apparent preference for deep waters (Becker et al., 2022a).

Based on sighting records, striped dolphins appear to have a continuous distribution in offshore waters from California to Mexico (Mangels & Gerrodette, 1994). The striped dolphin also occurs far offshore, in waters affected by the warm Davidson Current as it flows northward (Archer, 2009; Jefferson et al., 2008). During ship surveys conducted off the U.S. west coast in the summer and fall from 1991 to 2018, striped dolphins were sighted primarily from 100 to 300 NM offshore of the California coast (Barlow, 2016; Barlow & Forney, 2007; Becker et al., 2020). Striped dolphin encounters increase in deep, relatively warmer waters off the U.S. west coast (Becker et al., 2020; Becker et al., 2012a; Becker et al., 2016; Henderson et al., 2014), and their abudance generally decreases north of about 42°N (Barlow et al., 2009b; Becker et al., 2012a; Becker et al., 2016; Forney et al., 2012). There were only three striped dolphin encounters during 16 ship surveys off southern California from 2004 to 2008 (Douglas et al., 2014b) and they were not detected during 15 aerial surveys conducted in the Southern California Range Complex from 2008 through 2012 (Smultea & Jefferson, 2014).

Striped dolphins are a warm temperate to tropical species, and based on design- and model-based abundance estimates derived from line-transect survey data collected off the U.S. west coast from 1991 to 2018, densities are greatest when waters are warmest (Barlow, 2016; Becker et al., 2020; Becker et al., 2018). The distribution of striped dolphins off the U.S. west coast exhibits substantial annual variability due to changes in oceanographic conditions, often resulting in distribution shifts in and out of U.S. waters (Barlow, 2016; Barlow & Forney, 2007; Becker et al., 2022a). During the anomalously warm water year in 2014, striped dolphin abundance off the U.S. west coast increased dramatically, with striped dolphins seen as far north as 44° N (Barlow, 2016; Becker et al., 2020; Becker et al., 2018).

C.6.3.16.3 Population Trends

For the Hawaiian stock of striped dolphin, current abundance data preclude the assessment of population trends (Carretta et al., 2023b).

For the California, Oregon, and Washington stock of striped dolphins, because there is high annual variability in their distribution and abundance, no long-term trends have been identified (Carretta et al., 2023b).

C.6.3.16.4 Population Threats

Striped dolphins are susceptible to interactions with fisheries and entanglement in fishing gear. Recent monitoring data from 2015 through 2019 indicate that there were no recorded human-caused mortality or serious injury of this species in the California drift gillnet fishery (Carretta et al., 2023a). However, when factoring in a co-efficient of variation for non-detected strandings, the stranding data along the U.S. West Coast during 2015 through 2019 yields a minimum estimate of 20 fishery-related mortalities of striped dolphins (Carretta, 2021; Carretta, 2023; Carretta et al., 2016b).

In Hawaii, striped dolphins are threatened by the deep-set longline fishery and the shallow-set longline fishery. The most recent monitoring data from 2017 through 2021 indicate that there was one entanglement of this species in the shallow-set longline fishery, and none in the deep-set longline fishery (Bradford, 2018a; Bradford, 2018b; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019). Additionally, there are no estimates of mortality or serious injury of this species in Hawaii nearshore gillnet or hook and line fisheries; however, nearshore fisheries are not monitored for protected species bycatch and there have been anecdotal observations of this species interacting with these fisheries (Carretta, 2023).

Climate change has increasingly become a threat to marine mammals, including dolphins. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.17 Spinner Dolphin (*Stenella longirostris*)

Four well differentiated geographical forms of spinner dolphins have been described as separate subspecies but only *Stenella longirostris* (Gray's spinner dolphin) is present in the HCTT Study Area.

C.6.3.17.1 Status and Management

The spinner dolphin is protected under the MMPA and the species is not listed under the ESA. The eastern spinner dolphin (*Stenella longirostris orientalis*) is listed as depleted under the MMPA. Hawaiian spinner dolphins (considered a form of Gray's or pantropical spinner dolphin, *Stenella longirostris longirostris*) are considered as separate stocks from those involved in the tuna purse-seine fishery in the eastern tropical Pacific (Dizon et al., 1994). Under the MMPA, there are six stocks found within the U.S. EEZ of the Hawaiian Islands: (1) Hawaii Island, (2) Oahu/4-Islands, (3) Kauai/Niihau, (4) Pearl & Hermes Reef, (5) Kure/Midway, and (6) Hawaii Pelagic, including animals found both within the Hawaiian Islands EEZ (outside of island-associated boundaries) and in adjacent international waters (Carretta et al., 2023b).

C.6.3.17.2 Habitat and Geographic Range

Spinner dolphins occur in both oceanic and coastal environments and seasonal movement patterns for this species have not been documented. Spinner dolphins are pantropical, ranging through oceanic tropical and subtropical zones in both hemispheres (Jefferson et al., 2015). Based on an analysis of individual spinner dolphin movements in Hawaii, no spinner dolphins from the island associated stocks have been found farther than 10 NM from shore and few individuals move long distances (from one main Hawaiian Island to another) (Hill et al., 2011). Open ocean populations, such as the Hawaii Pelagic stock or those animals in the eastern tropical Pacific, often are found in waters with a shallow thermocline (rapid temperature difference with depth) (Au & Perryman, 1985; Perrin, 2008c; Reilly, 1990). The thermocline concentrates open sea organisms in and above it, which spinner dolphins feed on. In the eastern tropical Pacific, spinner dolphins are associated with tropical surface waters typified by extensive stable thermocline ridging and relatively little annual variation in surface temperature (Au & Perryman, 1985; Perrin, 2008c).

In the Hawaiian Islands, spinner dolphins occur along the leeward coasts of all the major islands and around several of the atolls northwest of the main Hawaiian Islands. Spinner dolphins occur year-round throughout the Hawaiian Islands, with primary occurrence from the shore to 4,000 m depth. This considers nearshore resting habitat and offshore feeding areas. Spinner dolphins are expected to occur in shallow water resting areas (about 50 m deep or less) throughout the middle of the day, moving into deep waters offshore during the night to feed (Heenehan et al., 2016; Heenehan et al., 2017; Norris & Dohl, 1980). Some of these resting areas are in proximity to bathymetric features that result in localized concentration of spinner dolphin prey. For example, there is an escarpment off Hawaii Island's Keahole Point that produces a locally enriched area that spinner dolphins exploit during nightly foraging trips from the nearby Makeko Bay (Heenehan et al., 2017; Norris & Dohl, 1980). Primary resting areas are along the west side of Hawaii, including Makako Bay, Honokohau Bay, Kailua Bay, Kealakekua Bay, Honaunau Bay, and Kauhako Bay, and off Kahena on the southeast side of the island (Heenehan et al., 2016; Heenehan et al., 2017; Norris & Dohl, 1980; Ostman-Lind et al., 2004; Tyne et al., 2017; Tyne et al., 2015). Along the Waianae coast of Oahu, Hawaii, spinner dolphins rest along Makua Beach, Kahe Point, and Pokai Bay during the day (Lammers, 2004). Kilauea Bay on Kauai is also a popular resting areas for Hawaiian spinner dolphins (U.S. Department of the Navy, 2006b). Monitoring for the Rim of the Pacific Exercise in 2006 resulted in daily sightings of spinner dolphins within the offshore area of

Kekaha Beach, Kauai, near the PMRF (U.S. Department of the Navy, 2006b). Spinner dolphins have been observed during Navy monitoring surveys at Kaula Island in 2000, 2003, and 2009–2011 (Richie et al., 2012). Although sightings have been recorded around the mouth of Pearl Harbor, Hawaii, spinner dolphin occurrence is rare there (Lammers, 2004; Richie et al., 2016). Occurrence patterns are assumed to be the same throughout the year.

During three systematic ship surveys of waters within the Hawaiian Islands EEZ in summer/fall of 2002, 2010, and 2017, there was a total of 15 sightings of spinner dolphin, the majority from the pelagic stock (Bradford et al., 2021). Habitat-based models developed from systematic ship survey data collected in the central North Pacific show the strong island association of spinner dolphins (Becker et al., 2012b; Forney et al., 2015), consistent with previously documented distribution patterns (Barlow, 2006).

Five year-round, non-hierarchical Small and Resident Population BIAs have been delineated for spinner dolphins in Hawaiian waters (Kratofil et al., 2023) (Figure C-28). The BIAs were based on the current insular stock boundaries and include Kuaihelani/Holaniku (4,841 km^2), Manawai (2,094 km^2), Kauai/Niihau (7,233 km^2), Oahu/Maui Nui (14,651 km^2), and the Island of Hawaii (9,477 km^2).

Spinner dolphins are not present in the California Study Area.

C.6.3.17.3 Population Trends

For spinner dolphins in Hawaii, differences in survey methodologies or insufficient data have precluded an assessment of any population trend for any of the six identified stocks (Carretta et al., 2023b).

C.6.3.17.4 Population Threats

Spinner dolphins are susceptible to interactions with fisheries and entanglement in fishing gear and marine debris. In Hawaii, the most recent data from 2012 through 2016 indicate that there were 7 recorded spinner dolphins entangled in marine debris or hooked by fishing gear (Bradford, 2018b; Bradford & Lyman, 2015a; Carretta, 2023). Throughout the years, there have been several recorded observations of this species seriously injured by entanglements in fishing equipment.

Spinner dolphins are also threatened by the deep-set longline fishery and the shallow-set longline fishery; however, the monitoring data from 2012 through 2016 indicate that there have been no observed entanglements or hookings of this species in these fisheries (Bradford, 2018b; Bradford & Forney, 2017; Carretta, 2023). There are no reports of interactions of spinner dolphins in nearshore gillnet or hook and line fisheries in Hawaii; however, nearshore fisheries are not monitored for protected species bycatch (Carretta, 2023).

Climate change has increasingly become a threat to marine mammals, including dolphins. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

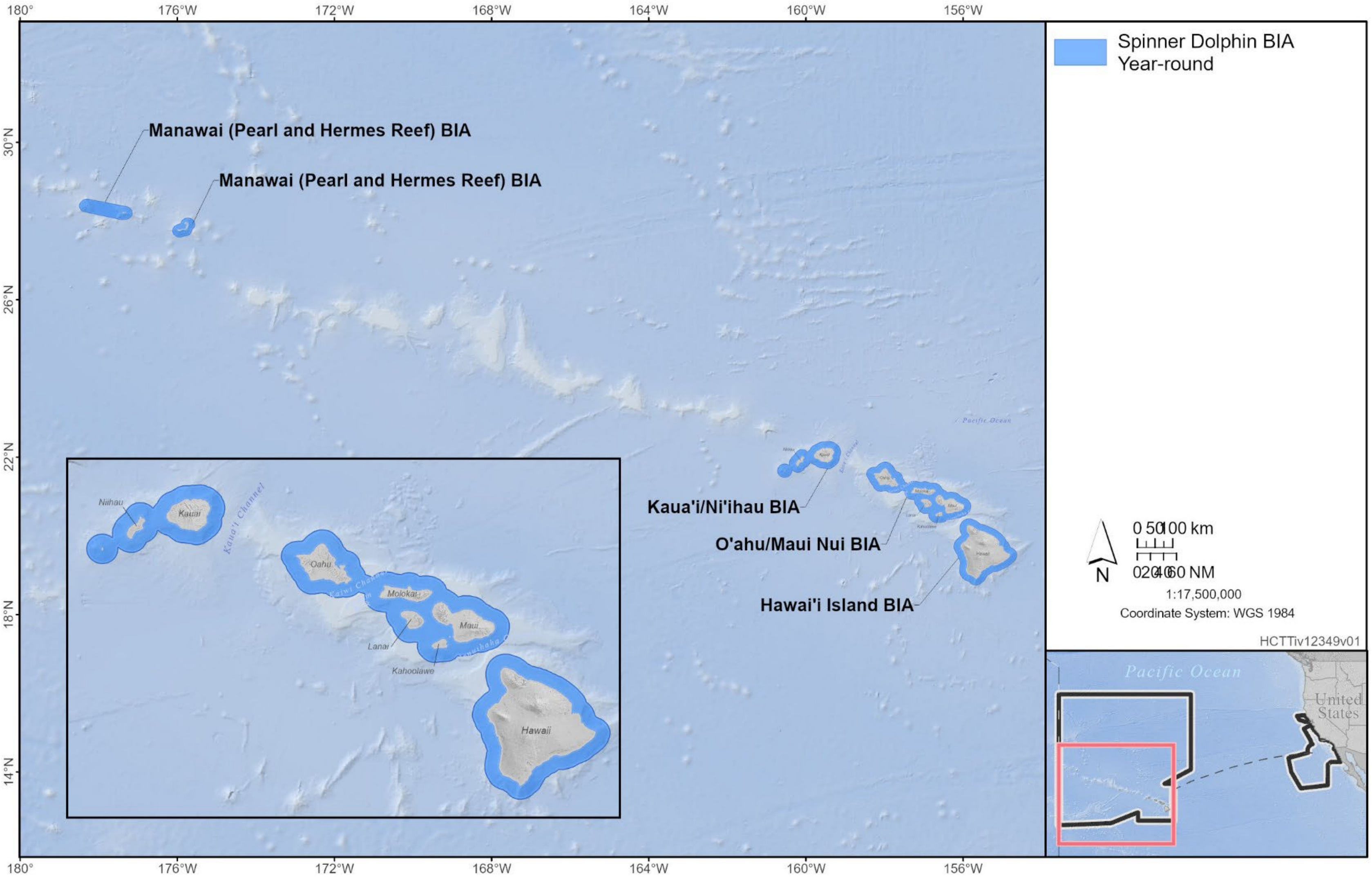


Figure C-28: Spinner Dolphin Small and Resident BIA in the Hawaii Study Area

C.6.3.18 Rough-toothed Dolphin (*Steno bredanensis*)

C.6.3.18.1 Status and Management

This species is protected under the MMPA and is not listed under the ESA. Rough-toothed dolphins are among the most widely distributed species of tropical dolphins, but little information is available regarding population status (Jefferson et al., 2015). There is a single Pacific management stock for rough-toothed dolphins found within the U.S. EEZ of the Hawaiian Islands, but there is no recognized stock of rough-toothed dolphins for the U.S. west coast (Carretta et al., 2023b).

C.6.3.18.2 Habitat and Geographic Range

Rough-toothed dolphins are well known in deep ocean waters off the Hawaiian Islands but are also seen relatively frequently during nearshore surveys (Baird et al., 2015f; Baird et al., 2008; Barlow et al., 2008; Bradford et al., 2013; Carretta et al., 2015; Pitman & Stinchcomb, 2002; Shallenberger, 1981; Webster et al., 2015). During three systematic ship surveys of waters within the Hawaiian Islands EEZ in summer/fall of 2002, 2010, and 2017, there was a total of 67 sightings of rough-toothed dolphin, with yearly mean group size estimates ranging from 15.7 to 25.3 animals (Bradford et al., 2021). Based on density estimates derived from these survey data, rough-toothed dolphin was one of the most abundant species present in the study area in each of the three years. Habitat-based models developed from systematic ship survey data collected in the central North Pacific show the strong island association of rough-toothed dolphins (Becker et al., 2012b; Forney et al., 2015). Sighting data from systematic ship surveys conducted within waters of the Hawaiian Islands EEZ from 2000 to 2020 supported the development of an updated habitat-based density model for rough-toothed dolphin and confirmed the strong island association indicated from the previous models (Becker et al., 2022a). Over a 10-day near-shore survey effort off Kauai in 2014, rough-toothed dolphins were encountered on two occasions and 7 of the 8 individuals photo-identified had been observed in previous years (Baird et al., 2015e). Data from 14 satellite tags deployed off Kauai between 2011–2015 on rough-toothed dolphins indicated a large portion of the core area for those animals overlaps the PMRF range and the channel between Kauai and Niihau (Baird et al., 2015e). The data presented by Baird et al. (2015e) and Webster et al. (2015) are indicative of residency on or near the PMRF range by some of those animals (see also (Baird et al., 2008).

A year-round Small and Resident Population parent BIA and child BIA have been delineated for waters off Kauai, Niihau, and Oahu for rough-toothed dolphins (Kratofil et al., 2023) (Figure C-29). A BIA was not identified for this population in the original BIA effort because there were insufficient data available at that time (Baird et al., 2015d). The parent BIA encompasses 25,083 km^2 of waters extending from the west coast of Oahu to the northwest and surrounding both Kauai and Niihau. The child BIA encompass 1,098 km^2 off the west coast of Kauai to capture the core range for this population (Kratofil et al., 2023). In addition, a year-round, non-hierarchical BIA was delineated for rough-toothed dolphins associated with Maui Nui and the Island of Hawaii. This BIA encompasses 15,112 km^2 of waters from the west coast of the Island of Hawaii, extending north to encompass waters off Maui Nui (Kratofil et al., 2023) (Figure C-28).

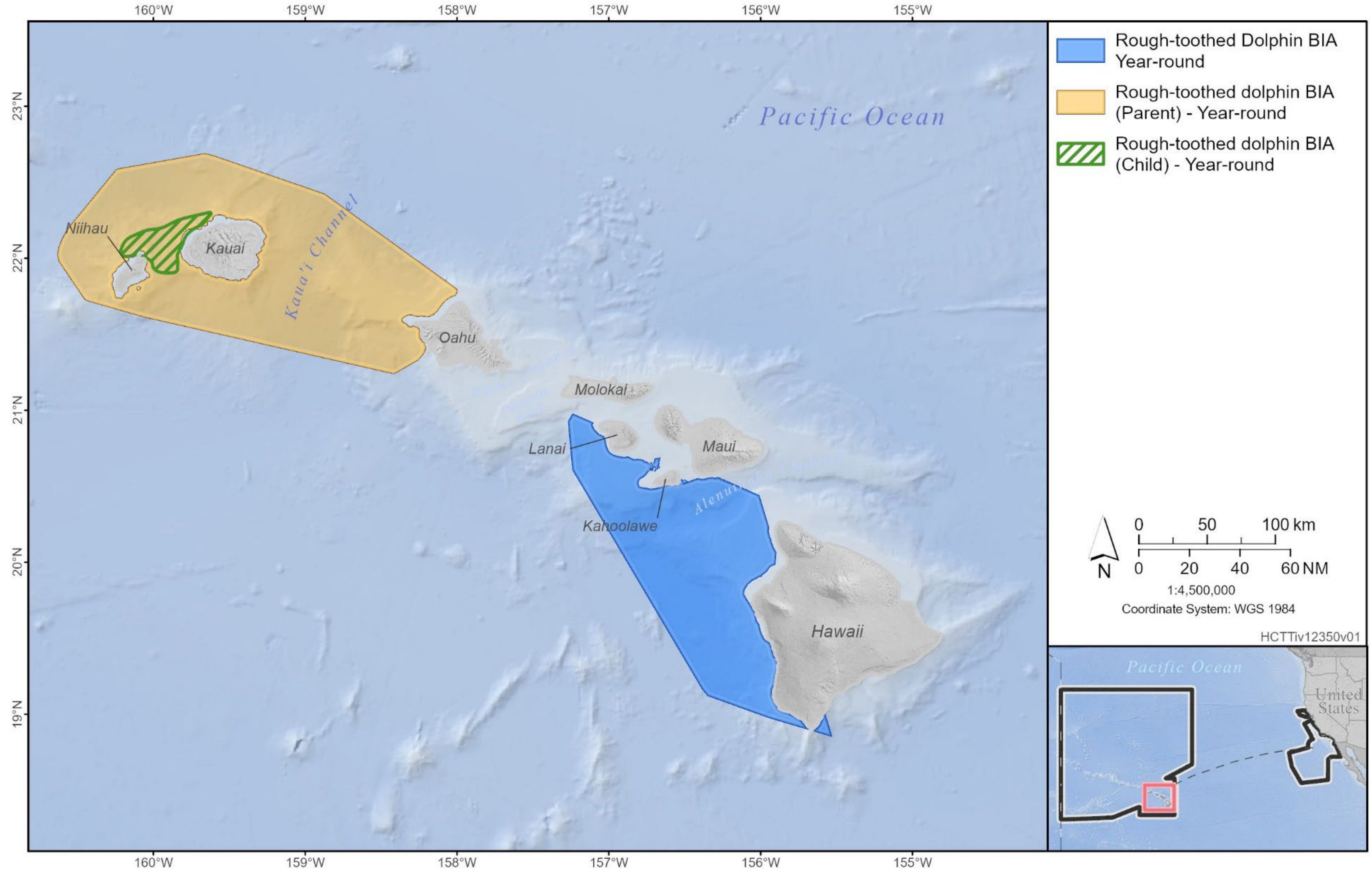


Figure C-29: Rough-Toothed Dolphin Small and Resident BIAs in the Hawaii Study Area

The northernmost range of the rough-toothed dolphin includes the southern portion of the California coast (Jefferson et al., 2015) but this is a tropical to subtropical species and there is no recognized stock for the U.S west coast (Carretta et al., 2023b). During systematic ship surveys off the U.S. west coast and the Baja California Peninsula from 1986 to 2005, there were no documented sightings of rough-toothed dolphins north of 25° N (Hamilton et al., 2009). Three strandings were documented for this species in central and Southern California between 1977 and 2002 with pneumonia identified as a cause of death (Zagzebski et al. 2006). This species has not been observed during seven systematic ship surveys from 1991 to 2014 off the U.S. west coast (Barlow, 2016). During 16 quarterly ship surveys off southern California from 2004 to 2008, there was one encounter with a group of 9 rough-toothed dolphins, which was considered an extralimital occurrence (Douglas et al., 2014b).

C.6.3.18.3 Population Trends

Available abundance estimates for the Hawaii stock of rough-toothed dolphins have broad and overlapping confidence intervals, thus precluding a robust assessment of population trends (Carretta et al., 2023b).

C.6.3.18.4 Population Threats

Rough-toothed dolphins are susceptible to interactions with fisheries and entanglement in fishing gear. In Hawaii, rough toothed dolphins are threatened by the deep-set longline fishery and the shallow-set longline fishery. The most recent monitoring data from 2017 through 2021 indicate that there were 8 observed hooking or entanglement of this species in the deep-set longline fishery, and none in the shallow-set fishery (Carretta et al., 2024; Carretta et al., 2023a). Of the 8 recorded observations occurring both within and out of the Hawaii EEZ, 3 rough-toothed dolphins were determined to be seriously injured and 3 interactions resulted in death.

There are no estimates of mortality or serious injury of this species in Hawaii nearshore gillnet or hook and line fisheries because nearshore fisheries are not monitored for protected species bycatch (Bradford & Lyman, 2018; Carretta, 2023). However, there were 52 photographs taken of rough-toothed dolphins that capture evidence of injuries likely resulting from hook and line fisheries in Hawaii (Carretta, 2023; Welch, 2017). There have also been rough-toothed dolphins observed in nearshore areas entangled in unidentified fishing gear.

Climate change has increasingly become a threat to marine mammals, including dolphins. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.19 Pacific White-sided Dolphin (*Lagenorhynchus obliquidens*)

C.6.3.19.1 Status and Management

This species is not listed under the ESA but is protected under the MMPA. Although there is evidence that two forms of Pacific white-sided dolphin occur off the U.S. west coast (a northern and southern stock), due to the difficulty of distinguishing the two stocks in the field, and given an area of apparent overlap off Southern California (Lux et al., 1997), NMFS currently recognizes a single stock, the California, Oregon, and Washington stock for the U.S. west coast (Carretta et al., 2023b).

C.6.3.19.2 Habitat and Geographic Range

Pacific white-sided dolphins are not present in the Hawaii Study Area.

Pacific white-sided dolphins are found in cold temperate waters across the northern rim of the Pacific Ocean as far north as the southern Bering Sea and as far south as the Gulf of California off Mexico (Ferguson, 2005; Jefferson et al., 2015; Leatherwood et al., 1984; Reeves et al., 2002). This species is also known to inhabit inshore regions of southeast Alaska, British Columbia, and Washington, and occurs seasonally off Southern California (Brownell et al., 1999; Forney & Barlow, 1998b). Sighting records and captures in open sea driftnets indicate that this species also occurs in oceanic waters well beyond the shelf and slope (Ferrero & Walker, 1996; Leatherwood et al., 1984).

Off California, Forney and Barlow (1998b) found significant north/south shifts in the seasonal distribution of Pacific white-sided dolphin, with the animals moving north into Oregon and Washington waters during the summer, and showing increased abundance in the Southern California Bight in the winter. There were a total of 121 sightings of Pacific white-sided dolphins during seven systematic ship surveys off the U.S. west coast in summer and fall between 1991 and 2014, with observed group sizes highly variable among the years (Barlow, 2016). During the unusually warm water conditions present in 2014, there were few sightings of Pacific white-sided dolphins off central and southern California (Barlow, 2016). Based on habitat models developed with systematic survey data collected during summer and fall from 1991 to 2018, the abundance of Pacific white-sided dolphin increased in shelf and slope waters and in relatively cooler waters in the study area (Becker et al., 2020). These patterns are consistent with previous habitat modeling efforts using a subset of the same data (Barlow et al., 2009b; Becker et al., 2010; Becker et al., 2012b; Becker et al., 2014; Forney et al., 2012). Based on ship survey data collected quarterly from 2004 to 2013, Pacific white-sided dolphins occurred year-round off southern California, but the majority of the sightings were in winter and spring when their distribution was more widespread (Campbell et al., 2014).

C.6.3.19.3 Population Trends

Multiple analyses of sightings and stranding data have indicated a significant decline in abundance over time from the Southern California Bight to the Gulf of California in Mexico (Barlow, 2016; Campbell et al., 2015; Salvadeo et al., 2010; Smultea & Jefferson, 2014). However, abundance estimates derived from both design- and model-based analyses for waters off the U.S. west coast show considerable seasonal and yearly variability (Barlow, 1995, 2016; Becker et al., 2020; Becker et al., 2014; Becker et al., 2018; Forney & Barlow, 1998b; Forney et al., 1995), making it difficult to support a robust trend analysis (Carretta et al., 2023b).

C.6.3.19.4 Population Threats

Pacific white-sided dolphins are threatened by interactions with fisheries, entanglement in fishing gear, and scientific research. The most recent monitoring data from 2015 through 2019 indicate that the estimate of mortality or serious injury of this species in the California drift gillnet fishery is 4.0 individuals annually (Carretta, 2023, 2022b). Additionally, injuries and mortalities of this species have resulted scientific research trails off the U.S. West Coast, specifically for sardines and rockfish. Research from 2015 through 2019 resulted in 14 mortalities and 1 serious injury of this species (Carretta, 2021; Carretta et al., 2023b).

Climate change has increasingly become a threat to marine mammals, including dolphins. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.20 Northern Right Whale Dolphin (*Lissodelphis borealis*)

C.6.3.20.1 Status and Management

This species it is not listed under the ESA but is protected by the MMPA. The management stock in U.S. waters consists of a single California, Oregon, and Washington stock (Carretta et al., 2023b).

C.6.3.20.2 Habitat and Geographic Range

The northern right whale dolphin occurs in cool-temperate to subarctic waters of the North Pacific Ocean, from the west coast of North America to Japan and Russia. This oceanic species is distributed from approximately 30°N to 50°N, 145°W to 118°E and generally not as far north as the Bering Sea (Jefferson et al., 2015). Occasional movements south of 30°N are associated with unusually cold water temperatures (Jefferson & Lynn, 1994). This species tends to occur along the outer continental shelf and slope, normally in waters colder than 68°F (20°C) (Jefferson & Lynn, 1994). Northern right whale dolphins generally move nearshore only in areas where the continental shelf is narrow or where productivity on the shelf is especially high (Smith et al., 1986).

Northern right whale dolphins are not present in the Hawaii Study Area.

Off California, the northern right whale dolphin is known to occur year-round, but abundance and distribution vary seasonally (Becker et al., 2014; Dohl et al., 1983; Douglas et al., 2014b; Forney & Barlow, 1998b). Northern right whale dolphins are primarily found off California during the colder water months, with distribution shifting northward into Oregon and Washington as water temperatures increase during late spring and summer (Barlow, 1995; Forney & Barlow, 1998b; Forney et al., 1995; Henderson et al., 2014). In the cool water period, the peak abundance of northern right whale dolphins in the California Study Area corresponds closely with the peak abundance of squid (Forney & Barlow, 1998b; Jefferson & Lynn, 1994). Northern right whale dolphins were sighted year-round during 16 ship surveys conducted from 2004 to 2008 off southern California, but the majority of the sightings were in winter and spring (Douglas et al., 2014b). There were a total of 92 sightings of northern right whale dolphin during seven systematic ship surveys off the U.S. west coast in summer and fall between 1991 and 2014, but the majority of these sightings were north of Point Conception (34.4°N), and there were no sightings off California during the unusually warm water conditions present in 2014 (Barlow, 2016)

As noted above, in the warm water periods, the northern right whale dolphin is not as abundant in California waters due to shifting distributions north into Oregon and Washington (Barlow, 1995; Becker et al., 2018; Forney & Barlow, 1998b; Forney et al., 1995). Based on habitat models developed with line-transect survey data collected off the U.S. west coast during summer and fall from 1991 to 2009, Becker et al. (2016) found that encounters of northern right whale dolphin increased in shelf and slope waters, and encounters decreased substantially in waters warmer than approximately 64°F (18°C). Recent habitat models that included additional survey effort collected in 2014 and 2018 confirmed that in the summer and fall, northern right whale dolphins were generally found in the coolest waters off the U.S. west coast (Becker et al., 2020). Northern right whale dolphins also tend to occur further offshore of California during the summer months (Douglas et al., 2014b; Forney & Barlow, 1998b).

C.6.3.20.3 Population Trends

Abundance estimates derived from both design- and model-based analyses show considerable seasonal and yearly variability (Barlow, 1995, 2016; Becker et al., 2020; Becker et al., 2014; Becker et al., 2018; Forney & Barlow, 1998b; Forney et al., 1995), making it difficult to assess trends in abundance (Carretta et al., 2023b). Examination of sighting and stranding data from the 1950s through 2012 suggest that the relative occurrence of northern right whale dolphin in the Southern California Bight has not changed over that period (Smultea & Jefferson, 2014).

C.6.3.20.4 Population Threats

Northern right whale dolphins are susceptible to interactions with fisheries and entanglement in fishing gear. The most recent monitoring data from 2015 through 2019 indicate that the estimate of mortality or serious injury of this species in the California drift gillnet fishery is 0.53 individuals annually (Carretta et al., 2021a).

Climate change has increasingly become a threat to marine mammals, including dolphins. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.21 Fraser's Dolphin (*Lagenodelphis hosei*)

Since its discovery in 1956, Fraser's dolphin was known only from skeletal specimens until it was once again identified in the early 1970s (Perrin et al., 1973). Although still one of the least-known species of cetaceans, Fraser's dolphin has become much better described as a species in recent years.

C.6.3.21.1 Status and Management

Fraser's dolphin is protected under the MMPA and is not listed under the ESA. For the MMPA stock assessment reports, there is a single Pacific management stock including only animals found within the U.S. EEZ of the Hawaiian Islands (Carretta et al., 2023b).

C.6.3.21.2 Habitat and Geographic Range

In the offshore eastern tropical Pacific, this species is distributed mainly in upwelling modified waters (Aguayo & Sanchez, 1987; Au & Perryman, 1985; Ferguson, 2005; Miyazaki & Wada, 1978; Reilly, 1990).

Fraser's dolphins have been documented within Hawaiian waters with the first published sightings occurring during a 2002 cetacean survey (Barlow, 2006). Fraser's dolphin vocalizations have also been documented in the Hawaiian Islands (Barlow et al., 2008; Barlow et al., 2004). During three systematic surveys of the Hawaiian Islands EEZ in the summer/fall of 2002, 2010, and 2017, there were a total of nine Fraser's dolphin sightings (Bradford et al., 2021). Based on the 2010 survey, Fraser's dolphin was one of the most abundant species within the Hawaiian Islands EEZ, having a notably large mean group size of 283 animals in the four pods observed (Bradford et al., 2013; Bradford et al., 2021). In small boat surveys nearshore around the Hawaiian Islands, Fraser's dolphins have only been seen twice in 10 years (both times off the Kona Coast of Hawaii Island) (Baird et al., 2013b). It is not known whether Fraser's dolphins found in Hawaiian waters are part of the same population that occurs in the eastern tropical Pacific (Carretta et al., 2023b).

Fraser's dolphins are not present in the California Study Area.

C.6.3.21.3 Population Trends

The available abundance estimates for Fraser's dolphins have broad and overlapping confidence intervals, thus precluding a robust assessment of population trends (Carretta et al., 2017b; Carretta et al., 2023b).

C.6.3.21.4 Population Threats

Fraser's dolphins are susceptible to interactions with fisheries and entanglement in fishing gear. In Hawaii, Fraser's dolphins are threatened by the deep-set longline fishery and the shallow-set longline fishery. However, the most recent monitoring data from 2014 through 2018 indicate that there have been no observed entanglements or hookings of this species in these fisheries (Bradford, 2018a; Bradford, 2018b; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019).

Climate change has increasingly become a threat to marine mammals, including dolphins. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.22 Risso's Dolphin (*Grampus griseus*)

C.6.3.22.1 Status and Management

Risso's dolphin is protected under the MMPA and is not listed under the ESA. For the MMPA stock assessment reports, Risso's dolphins within the Pacific U.S. EEZ are divided into two separate stocks: the Hawaiian stock in Hawaiian waters and the California, Oregon and Washington stock that occurs in the California Study Area (Carretta et al., 2023b).

C.6.3.22.2 Habitat and Geographic Range

In the Pacific, Risso's dolphins are found in the waters around the Hawaiian Islands (Bradford et al., 2017) and off the U.S. west coast (Barlow, 2016). Studies have documented that Risso's dolphins are found along the continental slope, over the outer continental shelf (Baumgartner, 1997; Canadas et al., 2002; Cetacean and Turtle Assessment Program, 1982; Davis et al., 1998b; Green et al., 1992; Kruse et al., 1999; Mignucci-Giannoni, 1998), and over submarine canyons (Mussi et al., 2004).

(Bradford et al., 2021) In December–January 2014 using a passive acoustic recording device onboard an unmanned glider south of Oahu, Risso's dolphins were acoustically detected throughout the entire survey except for the southernmost part between Bishop Seamount and McCall Seamount (Klinck et al., 2015). In addition, Risso's dolphins were sighted eight times during Navy monitoring activities within the Hawaii Range Complex between 2005 and 2012 (HDR, 2012). During three systematic ship surveys of waters within the Hawaiian Islands EEZ in summer/fall of 2002, 2010, and 2017, there was a total of 28 sightings of Risso's dolphin, with yearly mean group size estimates ranging from 15.0 to 26.6 animals (Bradford et al., 2021). Most of these sightings were in deep waters. Sighting data from these surveys, as well as additional systematic ship surveys conducted in Hawaiian waters from 2000 to 2020 allowed for the development of a habitat-based density model for Risso's dolphin (Becker et al., 2022b). Model predictions showed highest densities offshore of the islands in approximately 2,500 m to 4,500 m depth, and mid-range densities further offshore.

Risso's dolphin exhibits an apparent seasonal shift in distribution off the U.S. west coast, with movements from California waters north into Oregon and Washington waters in summer (Carretta et al., 2000; Forney & Barlow, 1998b; Green et al., 1992; Soldevilla et al., 2008). During ship surveys conducted quarterly off Southern California from 2004 to 2008, Risso's dolphins were encountered year-round, with highest encounters during the cold-water months (Douglas et al., 2014b), consistent with previously observed seasonal shifts in distribution (Carretta et al., 2000; Forney & Barlow, 1998b; Henderson et al., 2014; Soldevilla, 2008). Off California, they are commonly seen over the slope and in offshore waters (Barlow & Forney, 2007; Forney et al., 1995; Jefferson et al., 2008). This species is frequently observed in the waters surrounding SCI, California (Carretta et al., 2000). Habitat models derived from line-transect survey data collected between 1991 and 2009 off the U.S. west coast show that Risso's dolphins exhibit a disjunctive longitudinal distribution, suggesting that there may be two separate populations in this area, although additional genetic data are required for confirmation (Becker et al., 2016). Habitat models built using additional survey data collected in 2014 and 2018 confirmed this distribution pattern, showing that Risso's dolphin are generally concentrated either along the continental shelf (mainly south of 38° N) or in deep offshore waters, with a distinct longitudinal absence between these two areas (Becker et al., 2020).

C.6.3.22.3 Population Trends

In Hawaii, current abundance data do not support a robust trend analysis for this population (Carretta et al., 2023b).

Based on density estimates derived from aerial survey data collected from 2008 to 2013, the abundance of Risso's dolphin in Southern California waters appears to have increased (Jefferson et al., 2014). Examination of sighting and stranding data from the 1950s through 2012 also indicated an increase in the relative occurrence of this species in the Southern California Bight over this time period (Smultea & Jefferson, 2014). For Risso's dolphins in California, Oregon, and Washington waters, differences in estimated abundance between survey years is most likely due to the inter-annual variability in species distribution rather than a true abundance trend (Barlow, 1995, 2016; Becker et al., 2020; Becker et al., 2014; Becker et al., 2018; Forney & Barlow, 1998b; Forney et al., 1995).

C.6.3.22.4 Population Threats

Risso's dolphins are susceptible to interactions with fisheries and entanglements in fishing gear. Most recent monitoring data for this species from 2010 through 2014 indicate that the estimate of human-caused mortality or serious injury to Risso's dolphins in the California drift gillnet fishery is an average of 1.3 individuals annually (Carretta et al., 2017a; Carretta et al., 2023b). Additionally, stranding data along the U.S. West Coast during 2010 through 2014 yields a minimum estimate of 12 fishery-related mortalities of Risso's dolphins (Carretta et al., 2017a). There are also historical records of this species entangled and seriously injured or killed in ground fisheries and in the Southern California squid purse seine fishery.

In Hawaii, Risso's dolphins are threatened by the deep-set longline fishery and the shallow-set longline fishery. The most recent monitoring data from 2017 through 2021 indicate that there were 4 mortalities or serious injuries of this species in the shallow-set longline fishery, and 4 in the deep-set longline fishery (Bradford, 2018a; Bradford, 2018b; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019). Additionally, there are no estimates of mortality or serious injury of this species in Hawaii nearshore gillnet or hook and line fisheries; however, nearshore fisheries are not monitored for protected species bycatch (Carretta et al., 2023b).

Climate change has increasingly become a threat to marine mammals, including dolphins. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.23 Dall's Porpoise (*Phocoenoides dalli*)

C.6.3.23.1 Status and Management

This species is protected under the MMPA and is not listed under the ESA. Dall's porpoise is managed by NMFS in U.S. Pacific waters as two stocks: (1) a California, Oregon, and Washington stock and (2) an Alaskan stock (Carretta et al., 2023b).

C.6.3.23.2 Habitat and Geographic Range

Dall's porpoise is one of the most common odontocete species in north Pacific waters (Calambokidis & Barlow, 2004; Ferrero & Walker, 1999; Houck & Jefferson, 1999; Jefferson, 1991; Jefferson et al., 2008; Williams & Thomas, 2007; Zagzebski et al., 2006). Dall's porpoise is found from northern Baja California, Mexico, north to the northern Bering Sea and south to southern Japan (Jefferson et al., 1993). However, the species is only common between 32° N and 62° N in the eastern North Pacific (Houck & Jefferson, 1999; Morejohn, 1979). It is typically found in waters at temperatures less than 63° F (17° C) with depths of more than 180 m (Houck & Jefferson, 1999; Reeves et al., 2002).

Dall's porpoises are not present in the Hawaii Study Area.

Dall's porpoise distribution off the U.S. west coast is highly variable between years, most likely due to changes in oceanographic conditions (Barlow et al., 2009b; Becker et al., 2010; Becker et al., 2012b; Becker et al., 2018; Forney & Barlow, 1998b; Forney et al., 2012). North-south movements in California, Oregon, and Washington have been observed, with Dall's porpoise shifting their distribution southward during cooler-water periods on both interannual and seasonal time scales (Becker et al., 2014; Becker et al., 2018; Boyd et al., 2017; Forney & Barlow, 1998b). Based on habitat models developed using 1991–2018 survey data collected in waters off the U.S. west coast during summer and fall, Dall's porpoise density increased in shelf and slope waters , and decreased substantially in waters warmer than approximately 63°F (17°C) (Becker et al., 2020). These patterns are consistent with previous habitat modeling efforts using a subset of the same data (Barlow et al., 2009b; Becker et al., 2010; Becker et al., 2016; Becker et al., 2012b; Becker et al., 2014; Forney et al., 2012; Henderson et al., 2014).

During ship surveys conducted quarterly off southern California from 2004 to 2008, Dall's porpoise was encountered year-round, with highest encounters during the cold-water months (Douglas et al., 2014b). There were only five Dall's porpoise sightings during 18 aerial surveys conducted year-round in the SOCAL Range Complex from 2008 to 2013 (Jefferson et al., 2014).

C.6.3.23.3 Population Trends

Because there is high annual variability in the distribution and abundance of Dall's porpoise off the U.S. west coast, no long-term trends have been identified (Carretta et al., 2023b). Examination of sighting and stranding data from the 1950s through 2012 suggest that the relative occurrence of this species in the Southern California Bight has not changed substantially over this time period (Smultea & Jefferson, 2014).

C.6.3.23.4 Population Threats

Dall's porpoises are susceptible to interactions with fisheries and entanglement in fishing gear. The most recent monitoring data from 2015 through 2019 indicate that the estimate of mortality or serious injury of this species in the California drift gillnet fishery is 0.46 individuals annually (Carretta, 2021). Mortalities of Dall's porpoises have historically been recorded from ground fisheries off the U.S. West

Coast; however, data from 2012 through 2016 indicate there were no mortalities of this species in ground fisheries in the region during that time period (Carretta et al., 2023b; Jannot et al., 2018).

Climate change has increasingly become a threat to marine mammals, including porpoises. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.24 Harbor Porpoise (*Phocoena phocoena*)

C.6.3.24.1 Status and Management

Harbor porpoise is protected under the MMPA and is not listed under the ESA. This species is managed by NMFS in Pacific waters as six separate stocks including: 1) the Morro Bay stock, 2) the Monterey Bay stock, 3) the San Francisco-Russian River stock, 4) the Northern California/Southern Oregon stock, 5) the Northern Oregon/Washington coast stock, and 6) the Inland Washington stock (Carretta et al., 2023b). The southern range limit for this species is considered Point Conception (SAR). Based on published range boundaries (Carretta et al., 2023b), the stocks expected to occur within the California portion of the HCTT Study Area include the Northern California/Southern Oregon stock, the San Francisco-Russian River stock, the Monterey Bay stock, and the Morro Bay stock.

C.6.3.24.2 Habitat and Geographic Range

Harbor porpoises are not present in the Hawaii Study Area.

Harbor porpoise is a cool water species that occurs in cool temperate to subpolar regions of the Northern Hemisphere (Jefferson et al., 2016). In the eastern North Pacific harbor porpoise are found from Alaska south to Point Conception, California, generally in nearshore coastal and inland waters within a mile or two of shore and in waters <200 m deep (Barlow, 1988). Although harbor porpoises exhibit a continuous distribution along the U.S. west coast, significant genetic differences have been identified from multiple regions of California, Oregon, and Washington (Chivers et al., 2007). Genetic differences and density discontinuities identified from aerial surveys led to the identification of the separate harbor porpoise stocks (Carretta et al., 2023b).

Calambokidis et al. (2024) defined two non-hierarchical small and resident BIAs for the Monterey Bay and the Morro Bay stocks of harbor porpoise off California (Figure C-30). The Morro Bay BIA is 4,255 km^2 in size and the Monterey Bay BIA is 3,455 km^2 in size; both encompass waters from land to the 200 m isobath within the defined ranges for the respective stocks, and are identical in size to the original BIAs defined in 2015 (Calambokidis et al., 2015c).

C.6.3.24.3 Population Trends

Stratified distance sampling analysis within a Bayesian hierarchical model were recently used to examine trends in harbor porpoise abundance off the California coast (Forney et al., 2020). For the Morro Bay stock, analysis of data collected between 1986 and 2012 indicated that a marked increase in population occurred after 1991, when gillnet bycatch was largely eliminated within this stock's range.

For the Monterey Bay stock, analysis of data collected between 1986 and 2013 showed that during this period the size of this population increased from approximately 1,500 animals to more than 3,500 animals (Forney et al., 2020). Most of the increase occurred after gillnet fisheries were eliminated within this stock's range.

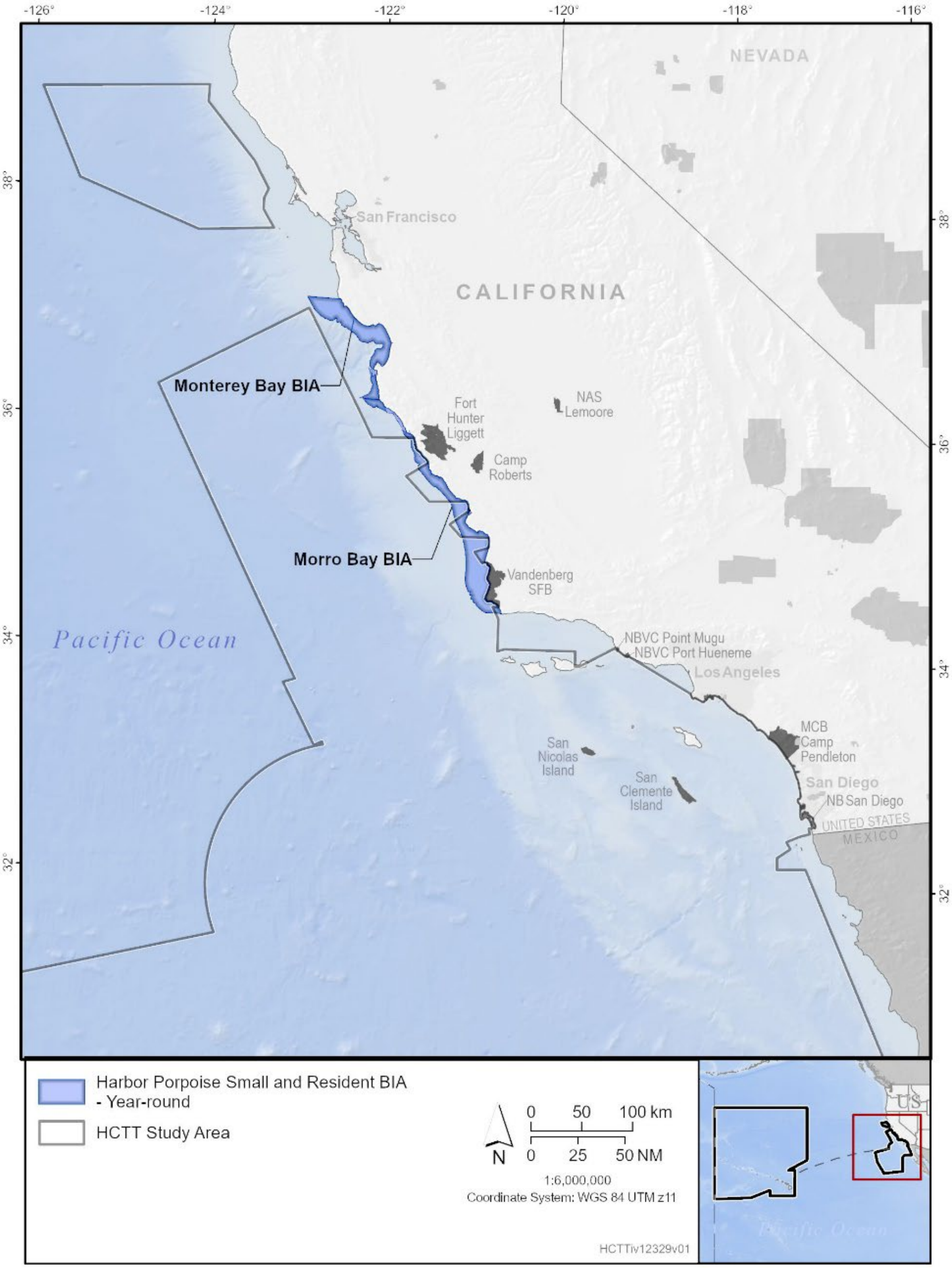


Figure C-30: Harbor Porpoise Small and Resident BIAs in the California Study Area

For the San Francisco-Russian River stock, analysis of data collected between 1986 and 2017 showed fluctuating population numbers over this period. Following the elimination of gillnets from the range of this stock in 1987, the population increased to a peak of approximately 13,500 animals in 2005 (Forney et al., 2020). Subsequently the size of the population appeared to decrease and level out to approximately 7,000 to 8,000 animals from 2010 to 2017. The apparent decrease after 2005 could be due to a shift in the distribution of animals into San Francisco Bay or could be a result of the large uncertainty in the 2002 to 2007 abundance estimates (Forney et al., 2020).

For the Northern California/Southern Oregon stock, analysis of data collected between 1989 and 2016 within the California portion of this stock's range shows a stable population over this time period, although there is high uncertainty in the abundance estimates (Forney et al., 2020). Since this analysis did not include the southern Oregon portion of this stock's range, a population trend analysis is not available for this stock (Carretta et al., 2023b).

C.6.3.24.4 Population Threats

Harbor porpoises are susceptible to interactions with fisheries and entanglement in fishing gear. Historically, white sea bass and halibut fisheries near Morro Bay and Monterrey Bay attributed to mortalities or serious injuries of this species (Carretta et al., 2023b). However, this fishery was banned from placing gillnets inshore of 110 m in 2002, which limited the interaction of harbor porpoises with this fishery. From 2015 through 2019, there have been no recorded strandings due to fishery interactions for the Morro Bay stock (Carretta et al., 2021a). During the same time period, there was one recorded stranding tied to fisheries for the Monterrey Bay stock (Carretta et al., 2021a).

Coastal gillnets have also been banned throughout the range of the San Francisco-Russian River stock; however, there have been 3 fishery-related strandings of this species from 2015 through 2019. It was determined that net fisheries were responsible for the strandings. From 2017 through 2021, there were no recorded strandings of the Northern California/Southern Oregon stock of harbor porpoises from fishery interactions (Carretta et al., 2023a).

Climate change has increasingly become a threat to marine mammals, including porpoises. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021). The northward expansion of bottlenose dolphins (due to climate change and sea temperature shifts) has increasingly overlapped with the range of the harbor porpoise. As a result, there have been increased instances of dolphin aggressions towards this species, which in turn could result in increased strandings (Gulland et al., 2022).

C.6.3.25 Cuvier's Beaked Whale (*Ziphius cavirostris*)

C.6.3.25.1 Status and Management

Cuvier's beaked whale is protected under the MMPA and is not listed under the ESA. There are three stocks of Cuvier's beaked whale recognized by NMFS: an Alaska stock, a California/Oregon/Washington stock, and a Hawaii stock (Carretta et al., 2023b). Animals in California or Hawaii are assigned to their respective stock.

C.6.3.25.2 Habitat and Geographic Range

Cuvier's beaked whales have an extensive range that includes all oceans, from the tropics to the polar waters of both hemispheres. Cuvier's beaked whales have been encountered in almost all areas of the

Pacific, including offshore areas of the central and eastern North Pacific, wherever surveys have occurred (Hamilton et al., 2009). Cuvier's beaked whales are generally sighted in waters with a bottom depth greater than 200 m and are frequently recorded in waters with bottom depths greater than 1,000 m (Bradford et al., 2013; Falcone et al., 2009; Jefferson et al., 2015). Acoustic sampling of bathymetrically featureless areas off Southern California detected many beaked whales over an abyssal plain, which counters a common misperception that beaked whales are primarily found over slope waters, in deep basins, or over seamounts (Griffiths & Barlow, 2016).

Cuvier's beaked whales are regularly found in waters surrounding the Hawaiian Islands (Baird et al., 2015d; Baird et al., 2009; Baird et al., 2013b; Barlow, 2006; Baumann-Pickering et al., 2010; Baumann-Pickering et al., 2014; Bradford et al., 2013; Lammers et al., 2015; Mobley, 2004; Oleson et al., 2013; Oleson et al., 2015; Shallenberger, 1981). In Hawaii, Cuvier's beaked whales have been occasionally observed breaching and this along with their large size and visible blows likely increases their detectability (Baird et al., 2013b). There was a total of 40 Cuvier's beaked whale sightings during systematic ship surveys within the Hawaiian Islands EEZ in 2002, 2010, and 2017, and none of the sightings were in waters within 140 km of the Main Hawaiian Islands (Bradford et al., 2021). Sightings have been reported off the Hawaiian Islands of Lanai, Maui, Hawaii, Niihau, and Kauai, providing strong evidence for both insular and offshore populations of Cuvier's beaked whales in waters of the Hawaiian Islands EEZ (Baird et al., 2015d; Baird et al., 2009Mobley, 2004 #986; Baird et al., 2013b; Oleson et al., 2013; Oleson et al., 2015; Shallenberger, 1981).

BIAs were redefined for a year-round Small and Resident Population area for Cuvier's beaked whales in Hawaiian waters (Kratofil et al., 2023). The parent BIA is 37,157 km^2 in size and the child BIA encompasses 5,400 km^2 within this region (Figure C-31) The child BIA was defined based on occurrence data that indicate that Cuvier's beaked whales spend the majority of their time between the 2,000 and 3,500 m isobaths off the leeward side of the Island of Hawaii.

Cuvier's beaked whale is the most commonly encountered beaked whale off the west coast of the U.S. (Carretta et al., 2023b). Research involving tagged Cuvier's beaked whales in the SOCAL Range Complex (Falcone & Schorr, 2011, 2012, 2013, 2014; Falcone et al., 2009) has documented movements in excess of hundreds of km. Schorr et al. (2014) reported that 5 out of 8 tagged whales journeyed approximately 250 km from their tag deployment location and one of these 5 made an extra-regional excursion over 450 km to the south to Mexico and back. During nine systematic ship surveys off the U.S. west coast between 1991 and 2018, there were a total of 91 Cuvier's beaked whale sightings, providing sufficient data to develop a habitat-based density model for this species (Fiedler et al., 2023). The model predicted highest numbers of Cuvier's beaked whales in deep, offshore waters of the study area. Repeated sightings of the same individuals have been reported off SCI in Southern California, which indicates some level of site fidelity (Falcone et al., 2009; Schorr et al., 2017). This species has also frequently been heard on passive acoustic recording devices in the southern portion of the California Study Area (Griffiths & Barlow, 2016; Širović et al., 2016). In a test of drifting passive acoustic recorders off California in the fall of 2014, Griffiths and Barlow (2016) reported beaked whale detections over slopes and seamounts, which was not unexpected, and also over deep ocean abyssal plains, which was a novel finding.

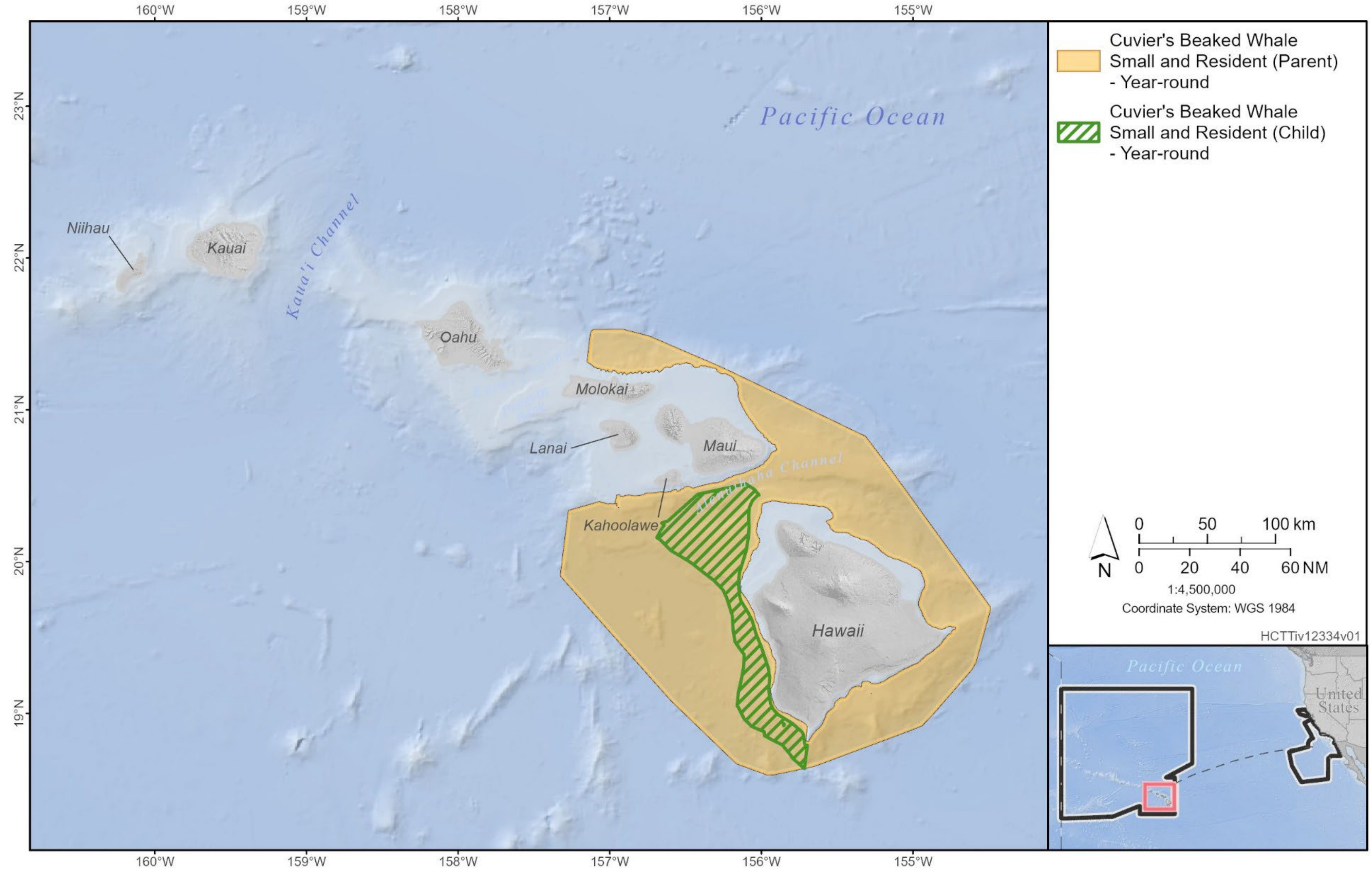


Figure C-31: Cuvier's Beaked Whale Small and Resident BIAs in the Hawaii Study Area

C.6.3.25.3 Population Trends

For the Hawaiian Islands, the currently available data precludes evaluation of population trends for Cuvier's beaked whales for the Hawaiian stock (Carretta et al., 2023b). Yearly abundance estimates for the Hawaiian Islands EEZ based on line-transect analyses are highly variable and have very broad confidence intervals, but this is likely due to the substantial variability in encounter rates during the individual survey years rather than a true change in population (Bradford et al., 2021).

A Bayesian trend analysis of systematic survey data collected from 1991 to 2008 suggested a decline in the abundance of beaked whales found in waters off California, Oregon, and Washington (Moore & Barlow, 2013). A more recent analysis that included additional survey data collected in 2014 indicated that the mean annual rate of population decline during this longer period was -3.0 percent per year (Moore & Barlow, 2017a). However, data from this study also indicated that while Cuvier's beaked whales along the entire U.S. west coast appear to have decreased in abundance from high values in 1991–1993, that decline now appears to have leveled off. Further, an acoustic-based estimate of Cuvier's beaked whale abundance in 2016 showed an increase from the previous estimates (Barlow et al., 2021). Unfortunately, this single estimate does not allow for a robust trend assessment given the difference in methodologies, but the greater precision in abundance estimates using acoustic-based methods highlights the potential of using this method for future trend assessments of this population (Barlow et al., 2021).

When considering beaked whales within the California Study Area, multiple studies have indicated that for waters surrounding the Navy training and testing areas in southern California the abundance of beaked whales remains high, including specifically where Navy has been training and testing for decades. Results from passive acoustic monitoring and other research have estimated regional Cuvier's beaked whale densities that were higher than indicated by the NMFS's broad-scale visual surveys for the U.S. west coast (Debich et al., 2015a; Debich et al., 2015b; Falcone & Schorr, 2012, 2014; Hildebrand et al., 2009; Moretti, 2016; Širović et al., 2016; Smultea & Jefferson, 2014). Research also indicates higher than expected residency in the Navy's instrumented Southern California Anti-Submarine Warfare Range in particular (Falcone & Schorr, 2012). Photo identification studies in the SOCAL Range Complex have identified approximately 100 individual Cuvier's beaked whale individuals with 40 percent having been seen in one or more prior years, with re-sightings up to 7 years apart (Falcone & Schorr, 2014). The documented residency by many Cuvier's beaked whales over multiple years indicate that a stable population may exist in that small portion of the stock's overall range (Falcone & Schorr, 2014; Falcone et al., 2009; Schorr et al., 2017). Based on Bayesian mark-recapture estimates of Cuvier's beaked whales in the San Nicolas Basin from photo-identification data collected from 2007 to 2018, an apparent decline of -0.8 percent of individuals per year was estimated (Curtis et al., 2020). However, this study also confirmed long-term site fidelity and high apparent annual survival rates, but the data did not support any definitive conclusions regarding population trends.

C.6.3.25.4 Population Threats

Cuvier's beaked whales are susceptible to interactions with fisheries and entanglements in fishing gear. Off the U.S. West Coast, the California swordfish fishery is the only fishery that has interacted with Cuvier's beaked whales in the region (Carretta et al., 2023b). However, the use of acoustic pingers in the fishery have resulted in no observed entanglements of this species since 1996 (Barlow & Cameron, 2003). Off Hawaii, Cuvier's beaked whales are threatened by the deep-set longline fishery and the shallow-set longline fishery. However, the most recent monitoring data from 2014 through 2018 indicate that there were no mortalities or serious injuries of this species in these fisheries (Bradford, 2018a; Bradford, 2018b; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019).

Anthropogenic noise such as sonar could disrupt their foraging behavior and echolocation activities of beaked whales and may result in unknown levels of injury or mortality (Carretta et al., 2023b; Tyack et al., 2011)There have been several stranding events of multiple beaked whale species over the years that may be associated with the use of sonar. It was found that tagged Cuvier's beaked whales demonstrated avoidance behaviors such as prolonged diving and cessation of echolocation clicks during sonar exposure to sonar (DeRuiter et al., 2013). In the Caribbean, monitoring of Blainville's beaked whales through hydrophones also demonstrated avoidance behaviors when exposed to sonar (Tyack et al., 2011). The absence of beaked whales in the California drift gillnet fisheries using pinging technology also provides evidence that this species may have increased sensitivity to anthropogenic noise (Carretta et al., 2023b).

Climate change has increasingly become a threat to marine mammals, including beaked whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.26 Baird's Beaked Whale (*Berardius bairdii*)

C.6.3.26.1 Status and Management

Baird's beaked whale is protected under the MMPA and is not listed under the ESA. Baird's beaked whale stocks are defined for two separate areas within Pacific U.S. waters where they are found: (1) Alaska and (2) California, Oregon, and Washington (Carretta et al., 2023b).

C.6.3.26.2 Habitat and Geographic Range

Baird's beaked whales are not present in the Hawaii Study Area.

Baird's beaked whale occurs mainly in deep waters over the continental slope, near oceanic seamounts, and areas with submarine escarpments, although they may be seen close to shore where deep water approaches the coast (Jefferson et al., 2008; Kasuya, 2009). This species is generally found throughout the colder waters of the North Pacific, ranging from off Baja California, Mexico, to the Aleutian Islands of Alaska (Jefferson et al., 2008; MacLeod & D'Amico, 2006), although they are found mainly north of 28° N in the eastern Pacific (Kasuya & Miyashita, 1997; Reeves et al., 2003).

Along the U.S. west coast, Baird's beaked whales are seen primarily along the continental slope, from late spring to early fall (Carretta et al., 2010; Green et al., 1992; Hamilton et al., 2009). Baird's beaked whales are sighted less frequently and are presumed to be farther offshore during the colder water months of November through April (Carretta et al., 2023b). Based on habitat-based density models developed using 1991–2018 survey data collected off the U.S. west coast during summer and fall (Becker et al., 2020), encounters of Baird's beaked whale increased in waters near the 2,000 m isobath. These patterns are consistent with previous habitat modeling efforts using a subset of the same data (Barlow et al., 2009b; Forney et al., 2012). Yearly density predictions from 1996 to 2018 showed relatively low variability in annual distribution patterns (Becker et al., 2020).

C.6.3.26.3 Population Trends

Bayesian trend analyses indicated that the abundance of Baird's beaked whales off the U.S. west coast has remained stable or increased slightly from 1991 to 2014 (Moore & Barlow, 2017a).

C.6.3.26.4 Population Threats

Baird's beaked whales are susceptible to entanglement in fishing gear and vessel strikes. Off the U.S. West Coast, the California large drift gillnet fishery is the only known fishery that has interacted with beaked whales in the region (Carretta et al., 2023b). However, the use of the use of acoustic pingers in this fishery has resulted in no observed entanglements of any beaked whale species since 1996 (Barlow & Cameron, 2003). Additionally, there was one recorded vessel strike of this species in 2016 that resulted in mortality (Carretta et al., 2021a)

Anthropogenic noise such as sonar could disrupt their foraging behavior and echolocation activities of beaked whales and may result in unknown levels of injury or mortality (Carretta et al., 2023b; Tyack et al., 2011). There have been several stranding events of multiple beaked whale species over the years that may be associated with the use of sonar. It was found that tagged Cuvier's beaked whales demonstrated avoidance behaviors such as prolonged diving and cessation of echolocation clicks during sonar exposure to sonar (DeRuiter et al., 2013). In the Caribbean, monitoring of Blainville's beaked whales through hydrophones also demonstrated avoidance behaviors when exposed to sonar (Tyack et al., 2011). The absence of beaked whales in the California drift gillnet fisheries using pinging technology also provides evidence that this species may have increased sensitivity to anthropogenic noise (Carretta et al., 2023b).

Climate change has increasingly become a threat to marine mammals, including beaked whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.27 Blainville's Beaked Whale (*Mesoplodon densirostris*)

C.6.3.27.1 Status and Management

Blainville's beaked whale is protected under the MMPA and is not listed under the ESA. In Hawaii and based on the number of sightings and genetic analysis of individuals around the Hawaiian Islands, NMFS recognizes a Hawaiian stock of Blainville's beaked whale (Carretta et al., 2023b). Due to the difficulty in distinguishing different *Mesoplodon* species from one another at sea during visual surveys, off the U.S. west coast NMFS designated a single management unit that includes all *Mesoplodon* species known to occur in these waters. This is the case in the California Study Area where six species of *Mesoplodon* beaked whales are represented by a single California/Oregon/Washington stock, including Blainville's beaked whale (Carretta et al., 2023b).

C.6.3.27.2 Habitat and Geographic Range

Blainville's beaked whales are one of the most widely distributed of the distinctive toothed whales within the *Mesoplodon* genus, found in both temperate and tropical waters (Jefferson et al., 2008; MacLeod & Mitchell, 2006). They are found mostly offshore in deeper waters along the California coast, Hawaii, Fiji, Japan, and Taiwan, as well as throughout the eastern tropical Pacific (Leslie et al., 2005; MacLeod & Mitchell, 2006; Mead, 1989).

Blainville's beaked whales are regularly sighted in Hawaiian waters (Baird et al., 2015a; Baird et al., 2003b; Baird et al., 2006; Barlow, 2006; Bradford et al., 2017; McSweeney et al., 2007), and their vocalizations have been routinely detected in acoustic monitoring in the Hawaiian Islands (Henderson et al., 2015; Klinck et al., 2015; Lammers et al., 2015; Manzano-Roth et al., 2016; Manzano-Roth et al., 2013; Rankin & Barlow, 2007). There were a total of 15 Blainville's beaked whale sightings during systematic ship surveys within the Hawaiian Islands EEZ in 2002, 2010, and 2017, and none of the sightings were in waters within 140 km of the Main Hawaiian Islands (Bradford et al., 2021).

Blainville's beaked whale has been detected off the coast of Oahu, Hawaii for prolonged periods annually, and this species is consistently observed in the same site off the west coast of the Island of Hawaii (Abecassis et al., 2015; Baird et al., 2006; McSweeney et al., 2007). Thirteen Blainville's beaked whales were satellite tagged off Hawaii Island between 2006 and 2012 with data records ranging from 15 to 159 days (Baird et al., 2015a; Baird et al., 2011). One tagged individual ranged from approximately 18 km to 573 km from land and moved a total of over 900 km from the initial tag location in 20 days. Similar data over an 8-day period for an individual tagged off Kauai showed movement on and off the Navy's instrument range at PMRF) three times before transiting to the southwest over a distance of approximately 100 km from the original tag location (Baird et al., 2015e).

BIAs were redefined for a year-round Small and Resident Population area for Blainville's beaked whales off the west coast and North Kohala portion of the Island of Hawaii, extending to the west and north to encompass waters off Maui Nui and Oahu (Kratofil et al., 2023). The parent BIA is 78,714 km^2 in size and the child BIA encompasses 4,214 km^2 within this region, representing an area of intensified use off the west coast and North Kohala portion of the Island of Hawaii (Figure C-32).

There are a handful of known records of Blainville's beaked whale from the coast of California and Baja California, Mexico, but the species does not appear to be common in the California portion of the Study Area (Hamilton et al., 2009; Mead, 1989; Pitman et al., 1988). *Mesoplodon* beaked whales were not detected during 15 aerial surveys conducted in the Southern California Range Complex from 2008 through 2012 (Smultea & Jefferson, 2014).

C.6.3.27.3 Population Trends

For the Hawaiian Islands, the currently available data precludes evaluation of population trends for Blainville's beaked whales in the Hawaiian stock (Carretta et al., 2023b). Acoustic monitoring using the Navy range hydrophones off Kauai from 2010 to 2014 suggest a low but stable abundance of *Mesoplodon* beaked whales at that location (Moretti, 2016).

A Bayesian trend analysis of systematic survey data collected from 1991 to 2008 suggested a decline in the abundance of beaked whales found in waters off California, Oregon, and Washington (Moore & Barlow, 2013). However, a more recent study that included data from an additional survey conducted in 2014 indicated that the pattern seen for the U.S. west coast from 1991 to 2014 indicates a reversal in that downward trend, and estimates a 0.87 percent probability of increase during this period (Moore & Barlow, 2017a).

C.6.3.27.4 Population Threats

Blainville's beaked whales are susceptible to fisheries interactions and entanglement in fishing gear. Off the U.S. West Coast, the California large drift gillnet fishery is the only known fishery that has interacted with beaked whales in the region (Carretta et al., 2023b). However, the use of the use of acoustic pingers has resulted in no observed entanglements of any beaked whale species in this fishery since 1996 (Barlow & Cameron, 2003).

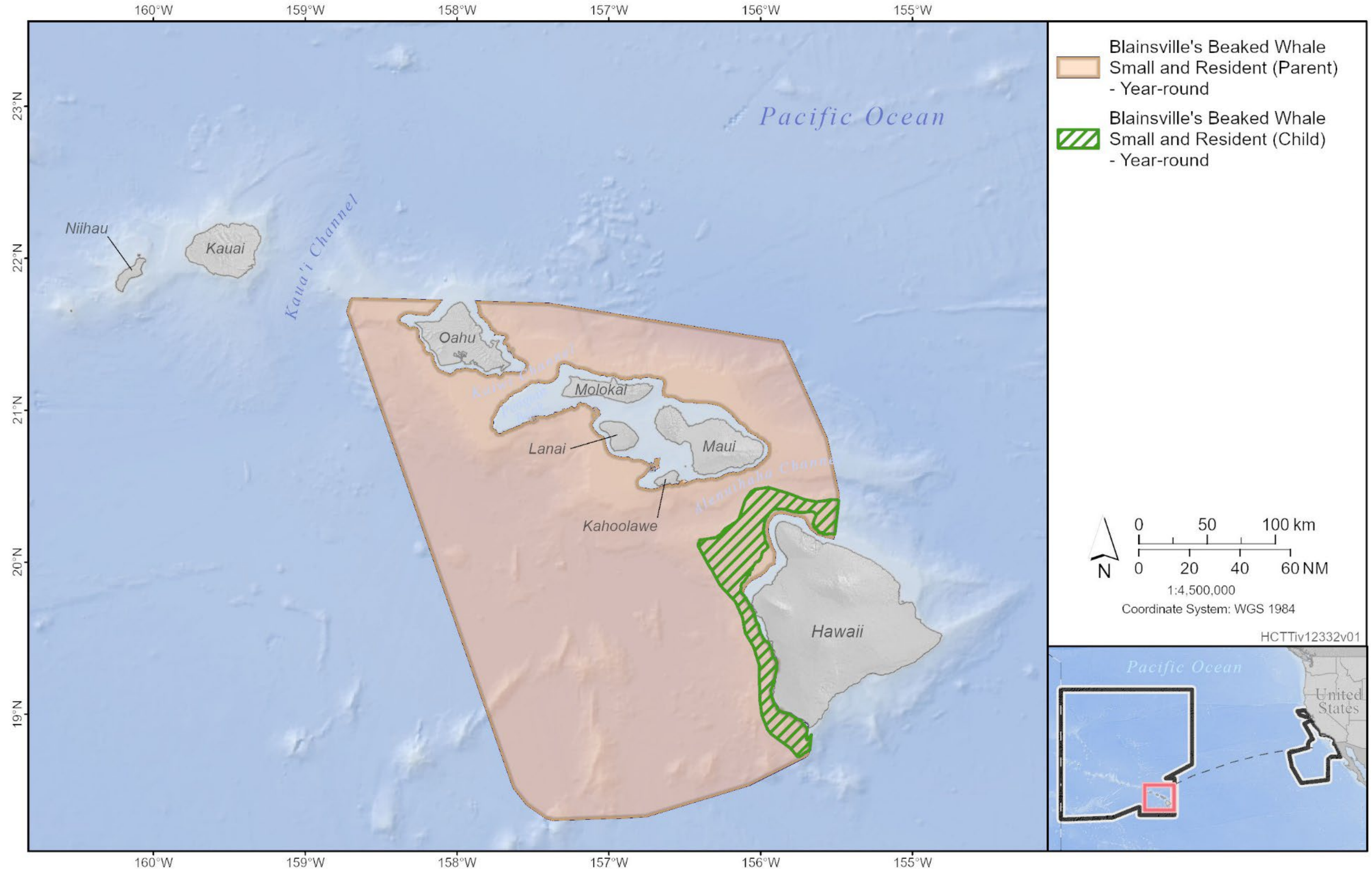


Figure C-32: Blainville's Beaked Whale Small and Resident BIAs in the Hawaii Study Area

Off Hawaii, Blainville's beaked whales are threatened by the deep-set longline fishery and the shallow-set longline fishery. However, the most recent monitoring data from 2014 through 2018 indicate that there were no mortalities or serious injuries of this species in these fisheries (Bradford, 2018a; Bradford, 2018b; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019). Additionally, there are no estimates of mortality or serious injury of this species in Hawaii nearshore gillnet or hook and line fisheries because nearshore fisheries are not monitored for protected species bycatch (Carretta et al., 2023b).

Anthropogenic noise such as sonar could disrupt their foraging behavior and echolocation activities of beaked whales and may result in unknown levels of injury or mortality (Carretta et al., 2023b; Tyack et al., 2011). There have been several stranding events of multiple beaked whale species over the years that may be associated with the use of sonar. It was found that tagged Cuvier's beaked whales demonstrated avoidance behaviors such as prolonged diving and cessation of echolocation clicks during sonar exposure to sonar (DeRuiter et al., 2013). In the Caribbean, monitoring of Blainville's beaked whales through hydrophones also demonstrated avoidance behaviors when exposed to sonar (Tyack et al., 2011). The absence of beaked whales in the California drift gillnet fisheries using pinging technology also provides evidence that this species may have increased sensitivity to anthropogenic noise (Carretta et al., 2023b).

Climate change has increasingly become a threat to marine mammals, including beaked whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.28 Longman's Beaked Whale (*Indopacetus pacificus*)

C.6.3.28.1 Status and Management

Longman's beaked whale is protected under the MMPA and is not listed under the ESA. Only one, the Hawaiian stock, is identified for the Pacific (Carretta et al., 2023b). This stock includes animals found within the Hawaiian Islands EEZ and adjacent high sea waters.

C.6.3.28.2 Habitat and Geographic Range

Longman's beaked whale is found in warm tropical waters, with most sightings occurring in waters with sea surface temperatures warmer than 78 °F (26°C) (Anderson et al., 2006; MacLeod & D'Amico, 2006; MacLeod et al., 2006). Although the full extent of this species' distribution is not fully understood, there have been many recorded sightings at various locations in tropical waters of the Pacific and Indian Oceans (Afsal et al., 2009; Dalebout et al., 2002; Dalebout et al., 2003; Moore, 1972). Sighting records of this species in the Indian Ocean showed that Longman's beaked whales are typically found in waters over deep bathymetric slopes of 200 to 2,000+ m (Anderson et al., 2006). In the Pacific, records of this species indicate presence in the eastern, central, and western Pacific, including waters off the coast of Mexico (Hamilton et al., 2009).

There was a total of 10 Longman's beaked whale sightings during systematic ship surveys within the Hawaiian Islands EEZ in 2002, 2010, and 2017 (Bradford et al., 2021). Observed group sizes during these three surveys were highly variable, ranging from approximately 7 to 99 individuals (Bradford et al., 2017). Longman's beaked whales have also been sighted off Kona (Cascadia Research, 2012) and there have been two known strandings of this species in the main Hawaiian Islands (Maldini et al., 2005; National Marine Fisheries Service, 2015c; West et al., 2012).

Longman's beaked whales are not present in the California Study Area.

C.6.3.28.3 Population Trends

The high uncertainty between the available design-based abundance estimates based on data collected during the 2002, 2010, and 2017 systematic surveys of the Hawaiian Islands EEZ precludes the evaluation of population trends for Longman's beaked whales (Carretta et al., 2023b).

C.6.3.28.4 Population Threats

The Longman's beaked whale is susceptible to fisheries interactions and entanglement in fishing gear. Longman's beaked whales are threatened by the Hawaiian deep-set longline fishery and the shallow-set longline fishery; however, the most recent monitoring data from 2014 through 2018 indicate that there were no mortalities or serious injuries of this species in these fisheries (Bradford, 2018a; Bradford, 2018b; Bradford et al., 2017; Carretta et al., 2023b; McCracken, 2019). Additionally, there are no estimates of mortality or serious injury of this species in Hawaii nearshore gillnet or hook and line fisheries because nearshore fisheries are not monitored for protected species bycatch (Carretta et al., 2023b).

Anthropogenic noise such as sonar could disrupt their foraging behavior and echolocation activities of beaked whales and may result in unknown levels of injury or mortality (Carretta et al., 2023b; Tyack et al., 2011). There have been several stranding events of multiple beaked whale species over the years that may be associated with the use of sonar. It was found that tagged Cuvier's beaked whales demonstrated avoidance behaviors such as prolonged diving and cessation of echolocation clicks during sonar exposure to sonar (DeRuiter et al., 2013). In the Caribbean, monitoring of Blainville's beaked whales through hydrophones also demonstrated avoidance behaviors when exposed to sonar (Tyack et al., 2011). The absence of beaked whales in the California drift gillnet fisheries using pinging technology also provides evidence that this species may have increased sensitivity to anthropogenic noise (Carretta et al., 2023b).

Climate change has increasingly become a threat to marine mammals, including beaked whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.29 Mesoplodont Beaked Whales (California, Washington Oregon stock)

C.6.3.29.1 Status and Management

The six species of Mesoplodont beaked whales known to occur off the U.S. west coast include Blainville's beaked whale (*M. densirostris*), Perrin's beaked whale (*M. perrini*), pygmy beaked whale (*M. peruvianus*), Stejneger's beaked whale (*M. stejnegeri*), Gingko-toothed beaked whale (*M. gingkodens*), and Hubbs' beaked whale (*M. carlhubbsi*). Due to the difficulty in distinguishing the different *Mesoplodon* species from one another at-sea, and thus due to the lack of species-specific abundance estimates, NMFS has combined six *Mesoplodon* species to make up the California, Oregon, and Washington stock of Mesoplodont beaked whales (Carretta et al., 2023b). None of the *Mesoplodon* species are listed under the ESA.

Of the six species included in this stock, Blainville's beaked whale occurs in Hawaiian waters and is addressed as an individual species in Section C.6.3.27. The other five beaked whale species are not expected to regularly occur in Hawaiian waters.

C.6.3.29.2 Habitat and Geographic Range

Worldwide, beaked whales normally inhabit continental slope and deep ocean waters (greater than 200 m) and are only occasionally reported in waters over the continental shelf (Canadas et al., 2002; Ferguson et al., 2006; MacLeod et al., 2006; Pitman, 2008; Waring et al., 2001). During eight systematic ship surveys off the U.S. west coast between 1991 and 2014, there were multiple Mesoplodont beaked whale sightings, although the majority were not identified to species (Carretta et al., 2023b).

Strandings along the U.S. west coast and elsewhere have provided some indication of marine mammal species range. Perrin's beaked whale is known only from five stranded specimens along the California coastline from 1975 to 1997 (Dalebout et al., 2002; MacLeod et al., 2006). These strandings include two at U.S. Marine Corps Base Camp Pendleton (33°15' N, 117°26' W), and one each at Carlsbad, (33°07' N, 117°20' W), Torrey Pines State Reserve (32°55' N, 117°15' W), and Monterey (36°37' N, 121°55' W) (Dalebout et al., 2002; Mead, 1981). Based on stranding data from the Pacific coast of Mexico, the pygmy beaked whale's range is thought to include deep waters off the Pacific coast of North America (Aurioles-Gamboa & Urban-Ramirez, 1993; Jefferson et al., 2008; Urban-Ramirez & Aurioles-Gamboa, 1992). This species was first described in 1991 from stranded specimens from Peru, and since then, strandings have been recorded along the coasts of both North and South America at Mexico, Peru, and Chile (Pitman & Lynn, 2001; Reyes et al., 1991; Sanino et al., 2007). MacLeod et al. (2006) suggested that the pygmy beaked whale occurs in the eastern Pacific from about 30° N to about 30° South (S). The handful of known records of the ginkgo-toothed beaked whale are from strandings, one of which occurred in California (Jefferson et al., 2015; MacLeod & D'Amico, 2006).

Acoustic monitoring has also provided information on the range for some *Mesoplodon* species in the California Study Area. Beaked whales produce species-specific frequency modulated echolocation pulses and acoustic monitoring devices located at seven sites in the Southern California Bight have recorded the presence of sounds identified as Stejneger's beaked whales and recorded other beaked whale-like frequency modulated pulse types that may possibly be produced by Perrin's beaked whale, Hubbs' beaked whale, and pygmy beaked whales (Baumann-Pickering et al., 2014; Baumann-Pickering et al., 2015; Debich et al., 2015a).

C.6.3.29.3 Population Trends

A Bayesian trend analysis of systematic survey data collected from 1991 to 2008 suggested a decline in the abundance of *Mesoplodon* beaked whales found in waters off California, Oregon, and Washington (Moore & Barlow, 2013). However, a more recent analysis that included additional survey data collected in 2014 indicated *Mesoplodon* beaked whales showed markedly higher abundance in 2014, reversing the declining trend from 1991 to 2008 that had been noted in the previous analysis (Moore & Barlow, 2017a). The increase may have been driven by an influx of tropical species of *Mesoplodon* during the unusually warm ocean conditions in 2014, and additional data are needed to better assess long-term population trends for this stock (Moore & Barlow, 2017a).

C.6.3.29.4 Population Threats

Mesoplodont beaked whales are susceptible to interactions with fisheries and entanglement in fishing gear. Off the U.S. West Coast, the California large drift gillnet fishery is the only known fishery that has interacted with beaked whales in the region (Carretta et al., 2023b).

Anthropogenic noise such as sonar could disrupt their foraging behavior and echolocation activities of beaked whales and may result in unknown levels of injury or mortality (Carretta et al., 2023b; Tyack et al., 2011). There have been several stranding events of multiple beaked whale species over the years that may be associated with the use of sonar. It was found that tagged Cuvier's beaked whales demonstrated avoidance behaviors such as prolonged diving and cessation of echolocation clicks during sonar exposure to sonar (DeRuiter et al., 2013). In the Caribbean, monitoring of Blainville's beaked whales through hydrophones also demonstrated avoidance behaviors when exposed to sonar (Tyack et al., 2011). The absence of beaked whales in the California drift gillnet fisheries using pinging technology also provides evidence that this species may have increased sensitivity to anthropogenic noise (Carretta et al., 2023b).

Climate change has increasingly become a threat to marine mammals, including beaked whales. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.30 California Sea Lion (*Zalophus californianus*)

C.6.3.30.1 Status and Management

The California sea lion is protected under the MMPA and is not listed under the ESA. The California sea lion is managed by NMFS as a single U.S. Stock (Carretta et al., 2017b).

C.6.3.30.2 Habitat and Geographic Range

California sea lions do not occur in Hawaii.

California sea lions are distributed in the eastern North Pacific from Puerto Vallarta, Mexico, through the Gulf of California and north along the west coast of North America to the Gulf of Alaska (Barlow et al., 2008; Jefferson et al., 2008; Maniscalco et al., 2004). During the summer breeding season, California sea lions congregate near rookery islands and nearby open-water areas. The primary rookeries off the U.S. West Coast are on San Nicolas, San Clemente, San Miguel, and Santa Barbara Islands (Carretta et al., 2023b; Lowry et al., 2021). This species is frequently found hauled out on human-made structures, including on docks, buoys, barges, and rip-rap or other tidal control structures.

California sea lions are the most commonly observed marine mammal in San Diego Bay (Graham & Saunders, 2015; U.S. Department of the Navy, 2015). There are two "bait" barges near the mouth of San Diego Bay that are haulout locations for California sea lions (U.S. Department of the Navy, 2015). During a pier replacement project at Point Loma from October 2014 to April 2015, an average of about 38 sea lions were observed hauled out with an additional 2–3 individuals in the water (U.S. Department of the Navy, 2015).

During the nonbreeding season, late summer through spring, adult and subadult males migrate north along the coast to Washington and into Puget Sound and return south the following spring (Jeffries, 2014; Lowry & Forney, 2005). Females and juveniles also disperse following the breeding season but tend to stay in waters off California to the north and west of the Channel Islands (Lowry & Forney, 2005; Melin & DeLong, 2000; Thomas et al., 2009).

California sea lions from the west coast of the Baja California Peninsula, Mexico, also migrate north and into waters off Southern California during fall and winter (Lowry & Forney, 2005). These sea lions are not part of the U.S. stock but are considered in the analysis of impacts because they are likely to occur in the Study Area.

California sea lions typically forage in waters over the continental shelf and slope; however, they are also known to occur farther offshore in deep, pelagic waters, particularly when prey is scarce (Jefferson et al., 2008; Melin et al., 2008; Urrutia & Dziendzielewski, 2012; Zavala-Gonzalez & Mellink, 2000).

Tagged California sea lions from Monterey Bay and SNI, California, demonstrated that adult males can travel more than 450 km from shore during longer foraging bouts (Weise et al., 2006; Weise et al., 2010); however, rehabilitated females and subadults normally stay mostly within 65 km of the coast (Thomas et al., 2009). Most individuals stay within 50 km of the rookery islands during the breeding season (Melin & DeLong, 2000). Females breeding and pupping on the Channel Islands typically feed over the continental shelf and generally remain within 150 km north and west of the islands (Kuhn & Costa, 2014; Melin & DeLong, 2000; Melin et al., 2008; Melin et al., 2012). Tagging results showed that lactating females foraging along the coast would travel as far north as Monterey Bay and offshore to the 1,000 m depth (Henkel & Harvey, 2008; Kuhn & Costa, 2014; Melin & DeLong, 2000; Melin et al., 2008). During the nonbreeding season, most locations of occurrence are over the slope or offshore; during the breeding season, most locations of occurrence are over the continental shelf (Melin & DeLong, 2000; Melin et al., 2008). Lowry and Forney (2005) estimated that 47 percent of sea lions would potentially be at-sea during the cold seasons.

Adult females alternate between nursing their pup on shore and foraging at sea, spending approximately 67–77 percent of time at sea (Kuhn & Costa, 2014; Melin & DeLong, 2000).

C.6.3.30.3 Population Trends

The U.S. Stock of California sea lions has an abundance of 257,606 (Carretta et al., 2022; Laake et al., 2018). The abundance estimate is based on a pup count from 2014; however, the mean pup count from 2016 through 2019 reported by Lowry et al. (2021) was nearly the same, suggesting that population growth may be leveling off. Furthermore, Laake et al. (2018) analyzed data from 1987 through 2015 and concluded that the population in 2014 was approaching carrying capacity.

In a comprehensive review of the status of the California sea lion, Hernández-Camacho et al. (2021) estimated the population abundance, including both the U.S. Stock and Mexico breeding population, to be between 327,157 and 334,205 individuals with 80 percent in the U.S. Stock; 14 percent off the Baja California Peninsula, Mexico; and 6 percent in the Gulf of California.

C.6.3.30.4 Population Threats

California sea lions are susceptible to interactions with fisheries and entanglements in fishing gear and marine debris. This species has been known to interact with the trawl, purse seine, and Gillnet fisheries off California. Both commercial and recreational hook and line fisheries have also attributed to injuries and mortalities of this species. From 2012 through 2016, there were 146 recorded mortalities of California sea lions from hook and line fisheries (Carretta et al., 2018; Carretta et al., 2023b).

Other human-caused threats include power plant entrainment, oil exposure, shootings, vessel strikes, dog attacks, and research activities (Carretta et al., 2018; Carretta et al., 2023b). Additionally, under the MMPA, individual California sea lions in the Columbia River have been removed from their environment or euthanized since 2008 due to their predation on the endangered salmon and steelhead fishes in the region (Carretta et al., 2023b).

Anthropogenic noise may also pose a threat to this species, as studies have shown changes in behavioral responses due to noise exposure. Houser et al. (2013) found that when California sea lions were exposed to sonar, they exhibited responses such as increases in respiration and submergence, lack of participation for food rewarding tasks, and evasive hauling out behaviors (Carretta et al., 2023b).

Climate change has increasingly become a threat to marine mammals, including pinnipeds. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021). Changes in the availability of prey due to changing ocean temperatures has already posed a threat to this species. From 2013 through 2017, an unusually high mortality event of pup and juvenile sea lions was attributed to the lack of availability of sardines (Carretta et al., 2023b). Additionally, there have been reports dating to 1998 of California sea lion mortalities due to domoic acid toxicity. Mortalities from domoic acid exposure could increase as blooms of the Psuedo-nizschia have been increasing in duration and extent partially due to climate change effects (Gulland et al., 2022).

C.6.3.31 Northern Fur Seal (*Callorhinus ursinus*)

C.6.3.31.1 Status and Management

Two stocks of northern fur seals (*Callorhinus ursinus*) are recognized in U.S. waters: the Eastern Pacific stock and the California stock (Carretta et al., 2023b; Young, 2023). The California stock breeds on San Miguel Island and the Farallon Islands, and the larger Eastern Pacific stock breeds on islands in the Bering Sea. Both stocks are protected under the MMPA, but neither stock is considered depleted or is listed under the ESA (Carretta et al., 2023b; Young, 2023).

C.6.3.31.2 Habitat and Geographic Range

Northern fur seals do not occur in the Hawaii Study Area.

Northern fur seals range from the northern Channel Islands off California, north along the coast of North America to the Bering Sea, and west to Japan (Carretta et al., 2022). They are typically found over the edge of the continental shelf and slope (Gentry, 2009; Sterling & Ream, 2004).

Northern fur seal breeding colonies are present at Adams Cove on San Miguel Island and on Castle Rock, an offshore island 1.1 km northwest of San Miguel Island (Baird & Hanson, 1997; Melin et al., 2012; Pyle et al., 2001; Stewart & Huber, 1993). Northern fur seal can occasionally haulout on SNI during summer (Baird & Hanson, 1997; Melin et al., 2012; Pyle et al., 2001). Animals from the California stock may remain in or near the area throughout the year but generally move to the North Pacific in waters off Washington, Oregon, and northern California to forage (Carretta et al., 2017b; Koski et al., 1998; Melin et al., 2012; Sterling et al., 2014).

Adult female Northern fur seals and juvenile of both sexes from the Eastern Pacific stock migrate along continental margins northern breeding islands from low-latitude winter foraging areas to (Gentry, 2009; Ragen et al., 1995). They leave the breeding islands in November and concentrate around the continental margins of the north Pacific Ocean in January and February, where they have access to vast, predictable food supplies (Gentry, 2009; Ream et al., 2005). Juveniles have been known to conduct trips between 8 and 29 days in duration, ranging from 171 to 680 km (Sterling & Ream, 2004). Adult female fur seals equipped with radio transmitters have been recorded conducting roundtrip foraging trips of up to 740 km (National Marine Fisheries Service, 2007a; Robson et al., 2004).

C.6.3.31.3 Population Trends

The abundance of the Eastern Pacific stock of northern fur seals is estimated to be 626,618 (CV = 0.2) fur seals, and the abundance of the California Stock is estimated at 14,050 fur seals .

C.6.3.31.4 Population Threats

Northern fur seals are likely threatened by interactions with fisheries, entanglement in fishing gear, and research activities. Stranding records off the U.S. West coast from 2009 through 2013 found that fishery-related interactions attributed to 4 mortalities (Carretta et al., 2023b; Carretta et al., 2014). Northern fur seals have also been killed due to research activities, particularly trawling operations. The rate of mortality or serious injury from research activities from 2009 through 2013 is 0.8 individuals annually (Carretta et al., 2023b). Other threats may include entanglement in marine debris, power plant entrainment, and oil exposure.

Climate change has increasingly become a threat to marine mammals, including pinnipeds. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.32 Northern Elephant Seal (*Mirounga angustirostris*)

C.6.3.32.1 Status and Management

The northern elephant seal is protected under the MMPA and is not listed under the ESA. The northern elephant seal population has recovered dramatically after being reduced to perhaps no more than 10 to 100 animals surviving in Mexico in the 1890s (Carretta et al., 2010; Hoelzel, 1999; Stewart et al., 1994). Movement and some genetic interchange occur among rookeries, but most elephant seals return to the rookeries where they were born to breed and thus may have limited genetic differentiation (Carretta et al., 2010). There are two DPS of northern elephant seals: one that breeds in Baja, Mexico, and a population that breeds in California. NMFS stock assessment report considers northern elephant seals in the Study Area to be from the California Breeding Stock, although elephant seals from Baja Mexico frequently migrate north through the California Study Area (Aurioles-Gamboa & Camacho-Rios, 2007).

C.6.3.32.2 Habitat and Geographic Range

Northern elephant seals are found in both coastal and deep waters of the eastern and central north Pacific. Elephant seals spend more than 80 percent of their annual cycle at sea, making long migrations to offshore foraging areas and feeding intensively to build up the blubber stores required to support them during breeding and molting haulouts (Hindell & Perrin, 2009; Le Boeuf & Laws, 1994; Worthy et al., 1992). Breeding and pupping take place on offshore islands and mainland rookeries (Carretta et al., 2010; Le Boeuf & Laws, 1994). Small colonies of northern elephant seals breed and haul-out on Santa Barbara Island and SCI with large colonies on SNI and San Miguel Island (Stewart et al., 1993a; Stewart et al., 1994). Aerial survey that included all the Channel Islands in July 2015 found the majority (approximately 61 percent) of elephant seals at San Miguel Island, approximately 21 percent at SNI, and 18 percent at Santa Rosa Island (Lowry et al., 2017). Elephant seals use these islands as rookeries from late December to February, and to molt from April to July. Northern elephant seals spend little time nearshore, and migrate through offshore waters four times a year as they travel to and from breeding/pupping and molting areas on various islands and mainland sites along the Mexico and California coasts.

With most of their prey found in open oceans, northern elephant seal juveniles and females are often found in deepwater zones while males also engage in benthic foraging and travel as far north as seamounts in the Gulf of Alaska (Le Boeuf et al., 2000; Le Boeuf et al., 1996; Robinson et al., 2012; Simmons et al., 2010; Simmons et al., 2007; Stewart & DeLong, 1995).

There are records of three northern elephant seals being present in the Hawaiian Islands, indicating that movements beyond their normal range do occur, but are very rare. A female, an immature male, and mature male were sighted on Midway Island in the northwestern Hawaiian Islands in 1978 (Tomich, 1986). On January 2, 2002, a juvenile male elephant seal was discovered on Molokai and reported to be the second confirmed sighting in the Main Hawaiian Islands since 2001 (National Marine Fisheries Service, 2006). This same elephant seal was next encountered on January 11, 2002 on the Kona coast of Hawaii at Kawaihae Beach and later at the Kona Village Resort where it was captured and returned to California by NMFS (Fujimori, 2002). These occurrences in the Hawaiian Islands are considered extralimital and northern elephant seals are not expected to be present in Hawaii Study Area.

Northern elephant seals are found in both coastal areas and deeper waters off Southern California (Carretta et al., 2010; Jefferson et al., 2008; Robinson et al., 2012). The foraging range of northern elephant seals extends thousands of km offshore from the breeding range into the central North Pacific Transition Zone well to the north of Hawaii; however, their range is not considered to be continuous across the Pacific (Simmons et al., 2010; Stewart & Huber, 1993). Adult males and females segregate while foraging and migrating (Simmons et al., 2010; Stewart, 1997; Stewart & DeLong, 1995). Adult females mostly range west to about 173° W, between the latitudes of 40° N and 45° N, whereas adult males range farther north into the Gulf of Alaska and along the Aleutian Islands to between 47° N and 58° N (Le Boeuf et al., 2000; Robinson et al., 2012; Stewart & DeLong, 1995; Stewart et al., 1993a). Adults stay offshore during migration, while juveniles are often seen along the coasts of Oregon, Washington, and British Columbia (Le Boeuf et al., 1996; Stewart & Huber, 1993). The most far-ranging individual appeared on Nijima Island off the Pacific coast of Japan in 1989 (Kiyota et al., 1992). This demonstrates the great distances that these animals are capable of covering.

C.6.3.32.3 Population Trends

The population in California continues to increase, but the Mexican stock appears to be stable or slowly decreasing (Carretta et al., 2015; Lowry et al., 2014; Stewart & DeLong, 1994). Some evidence indicates that elephant seals may be expanding their pupping range northward, possibly in response to continued population growth (Hodder et al., 1998). Hodder et al. (1998) noted a possible emerging breeding colony at Shell Island off Cape Arago in southern Oregon. Other northern mainland breeding rookeries include Ano Nuevo, Point Reyes and Cape San Martin (Stewart et al., 1994).

C.6.3.32.4 Population Threats

Northern elephant seals are susceptible to interactions with fisheries and entanglement in fishing gear. The total estimated annual mortalities of this species in commercial fisheries off California is 5.3 individuals annually (Carretta et al., 2021a).

Additionally, the several recorded mortalities or injuries of northern elephant seals have been attributed to shootings, hook and line fisheries, marine debris, dog attacks, vehicle strikes, harassment, oil spills, and vessel strikes (Carretta et al., 2021a) From 2015 through 2019, there were approximately 42 mortalities or serious injuries resulting from the threats listed above (Carretta et al., 2021a).

Climate change has increasingly become a threat to marine mammals, including pinnipeds. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.33 Harbor Seal (*Phoca vitulina*)

C.6.3.33.1 Status and Management

The harbor seal is protected under the MMPA and is not listed under the ESA. Harbor seals are distributed in temperate to cold water regions in the north Pacific. The Society of Marine Mammalogy's Committee on Taxonomy (2016) has determined that all harbor seals in the north Pacific should be recognized as a single subspecies (*Phoca vitulina richardii*) until the subspecies limits of various populations are better known. There are 17 stocks of harbor seal along the U.S. west coast (Carretta et al., 2017b; Muto & Angliss, 2016); there is a single California stock occurring within the southern portion of the California Study Area.

C.6.3.33.2 Habitat and Geographic Range

The harbor seal is one of the most widely-distributed seals, found in nearly all temperate coastal waters of the northern hemisphere (Jefferson et al., 2008). Harbor seals are generally not present in the open ocean. Harbor seals are not present in the Hawaii Study Area.

Harbor seals, while primarily aquatic, also use the coastal terrestrial environment, where they haul out of the water periodically. Harbor seals are a coastal species, rarely found more than 20 km from shore, and frequently occupy bays, estuaries, and inlets (Baird, 2001). Individual seals have been observed several kilometers upstream in coastal rivers (Baird, 2001). Harbor seals are not considered migratory (Burns, 2008; Jefferson et al., 2008).

Ideal harbor seal habitat includes suitable haulout sites, shelter from high surf during the breeding periods, and sufficient food near haulout sites to sustain the population throughout the year (Bjorge, 2002). Haulout sites vary, but include intertidal and subtidal rock outcrops, sandbars, sandy beaches, estuaries, and even peat banks in salt marshes (Burns, 2008; Gilbert & Guldager, 1998; Prescott, 1982; Schneider & Payne, 1983; Wilson, 1978).

Small numbers of harbor seals are found hauled out on coastal and island sites and forage in the nearshore waters of the SOCAL Range Complex, but are found in only moderate numbers compared to sea lions and elephant seals. In California, approximately 400 to 600 harbor seal haulout sites are widely distributed along the mainland and on offshore islands of the state (Lowry et al., 2008). The harbor seal haul-out sites in the San Diego area include mainland beaches and all of the Channel Islands, including Santa Barbara, Santa Catalina, and SNI (Lowry et al., 2008). There were for instance 1,367 harbor seals counted in the Channel Islands during aerial surveys in July 2015 (Lowry et al., 2017). Individuals have also been observed hauled out at La Jolla Cove, and within the channel of San Diego Bay at Ballast Point and Navy Base Point Loma. Monitoring during a pier replacement project in at Point Loma (October 2014–April 2015) encountered a mean number of three harbor seals hauled out and 2.00 to 2.48 per day in the water (U.S. Department of the Navy, 2015). A total of 15 harbor seals were sighted off the coast during 18 aerial surveys conducted between 2008 and 2013 in the southern portion of the California Study Area (Jefferson et al., 2014). There were no harbor seals detected in the 17 days of surveys (between October 2013 and September 2014) nearshore off the Silver Strand Training Complex and San Diego Bay (Graham & Saunders, 2015).

C.6.3.33.3 Population Trends

The most recent (2011) survey of California harbor seal rookeries resulted in the highest recorded pup count since 1975 (Carretta et al., 2015). In the short term, this trend may be affected by the pinniped UME that has been ongoing on the U.S. west coast since 2013.

C.6.3.33.4 Population Threats

Historically, harbor seals were threatened by hunting in the region; however, the population has since made dramatically increased.

Harbor seals are susceptible to interactions with fisheries and entanglement in fishing gear. Off the U.S. West Coast, this species has historically interacted most with the set gillnet fisheries for halibut and sea bass (Carretta et al., 2023a). From 1990 through 1994, mortality estimates ranged from 227 to as much as 1,204 harbor seals in the region; however, these fisheries are no longer observed as frequently as they previously were (Carretta et al., 2023b; Carretta et al., 2014; Julian & Beeson, 1998). They also were often observed entangled in salmon gillnet fisheries in Puget Sound; however, this fishery no longer exists.

There are also recorded observations of harbor seal interactions with tribal fisheries in Washington, mainly tribal gillnet fisheries. The Northwest Indian Fisheries Commission reported a total of 166 serious injuries or mortalities of harbor seals from the Washington inland waters stocks (Carretta et al., 2024). Takes of harbor seals for tribal subsistence purposes are not currently reported.

Other recorded anthropogenic mortalities or injuries of harbor seals have been attributed to shootings, stabbings/wounds, harassment, research, dog attacks, marine debris, fur traps, vehicle collisions, and vessel strikes. From 2017 through 2021, there were approximately 82 mortalities or serious injuries of harbor seals resulting from the threats listed above (Carretta et al., 2023a).

Climate change has increasingly become a threat to marine mammals, including pinnipeds. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.6.3.34 Steller Sea Lion (*Eumetopias jubatus*)

C.6.3.34.1 Status and Management

Information on stocks, status, and abundance is provided in Table 3.7-2 of Section 3.7 (Marine Mammals).

C.6.3.34.2 Habitat and Geographic Range

Steller sea lions range along the North Pacific Rim from northern Japan to California, with centers of abundance and distribution in the Gulf of Alaska and Aleutian Islands. The species is not known to migrate, but many individuals disperse widely outside of the breeding season. NMFS has designated two Steller sea lion stocks in the North Pacific corresponding to two DPSs (Muto et al., 2020). The Eastern U.S. Stock (or DPS) is defined as the population occurring east of 144°W longitude, and the Western U.S. Stock (or DPS) consists of sea lions occurring west of 144°W longitude. Although the distribution of individuals from the two stocks overlaps outside of the breeding season (May–July), Steller sea lions typically return to their natal rookeries and haulouts in each DPS area prior to the breeding season (Fritz et al., 2016; Jemison et al., 2013; Muto et al., 2017; Muto et al., 2018; Muto et al., 2020; National Marine Fisheries Service, 2013a; Raum-Suryan et al., 2004; Sigler et al., 2017).

Only Steller sea lions from the Eastern U.S. Stock are expected to occur in waters off California, with highest levels of occurrence in the northern part of the Study Area and fewer occurring in the Channel Islands and Southern California waters. Important haulouts along the California coastline include Año Nuevo Island and the Farallon Islands in Central California and the Saint George Reef rookery and the Sugarloaf Island rookery at Cape Mendocino in northern California Lowry et al. (2021).

Detailed information on the distribution of this species in the Study Area is provided in the *U.S. Navy Marine Species Density Database Phase IV for the Hawaii-California Training and Testing Study Area* technical report (U.S. Department of the Navy, 2024).

C.6.3.34.3 Population Trends

Information on population trends is provided in Table 3.7-2 of Section 3.7 (Marine Mammals).

C.6.3.34.4 Population Threats

Stellar sea lions are susceptible to entanglement in fishing gear. In Alaska, stellar sea lions have been documented interacting with the halibut longline fishery, sablefish longline fishery, and the salmon drift gillnet fishery (Delean et al., 2020). Off the U.S. West Coast, this species interacts with several groundfish fisheries, the sablefish hook and line fishery, and the California halibut bottom trawl (Carretta et al., 2023b; Jannot et al., 2018). The total estimated mean mortality or serious injury of this species due to U.S. commercial fisheries from 2013 through 2017 is 24 individuals annually (Carretta et al., 2023b). In Alaska, stellar sea lions are hunted by native populations for subsistence harvests. Available data from 2005 through 2008 and 2012 indicate that approximately 11 individuals were harvested or struck and lost annually (Carretta et al., 2023b; Delean et al., 2020). Stellar sea lions are also hunted in Canada; however, the amount of stellar sea lions harvested is likely minimal.

Other recorded human-caused mortalities or injuries of stellar sea lions have been attributed to recreational fisheries, marine debris, illegal shootings, vessel strikes, and explosives (Carretta et al., 2023b).

Climate change has increasingly become a threat to marine mammals, including pinnipeds. The predicted effects of climate change on marine mammals include habitat loss, shifts in range, changes in competition, changes in prey availability and abundance, altered foraging and reproductive behavior, and increased susceptibility to diseases and harmful algae (Gulland et al., 2022; Nelms et al., 2021).

C.7 Reptiles

C.7.1 General Background

There are two types of marine reptiles analyzed in this EIS/OEIS—sea turtles (including five species of sea turtles, all of which are listed under the ESA) and sea snakes (one species of sea snake that is not currently listed under the ESA).

Sea turtles are highly migratory, long-lived reptiles that occur throughout the open-ocean and coastal regions of the Study Area. Generally, sea turtles are distributed throughout tropical to subtropical latitudes, with some species extending into temperate seasonal foraging grounds. Leatherback sea turtles are partially endothermic, where they can tolerate colder waters relative to other sea turtle species. Leatherback sea turtles are partially endothermic, where they can tolerate colder waters relative to other sea turtle species. This allows for a much greater range at higher latitudes than other sea turtles, which are generally exothermic and therefore less tolerant of colder waters. In general, sea turtles spend most of their time at sea, with female turtles returning to land to nest. Green sea turtles in

Hawaii, however, will also bask on the shore, particularly on beaches with lower levels of human disturbance. Green sea turtles bask on the shore for purposes of resting, thermoregulation (increasing their temperature by laying in the sun), and predator avoidance. Females also haul out onshore to avoid the advances of males during reproductive season (Spotila, 2004).

Sea snakes, also known as coral reef snakes, form a subfamily of venomous snakes closely related to the cobra and other terrestrial venomous snakes of Australia (Heatwole, 1999). Most species of sea snakes are adapted to a fully aquatic life, with few records on land (Udyawer et al., 2013). Only the yellow-bellied sea snake is thought to occur within the HCTT Study Area. Because of this species' passive drifting ecology, yellow-bellied sea snake sightings are reported in nearshore waters of Hawaii and California where they do not maintain resident breeding populations.

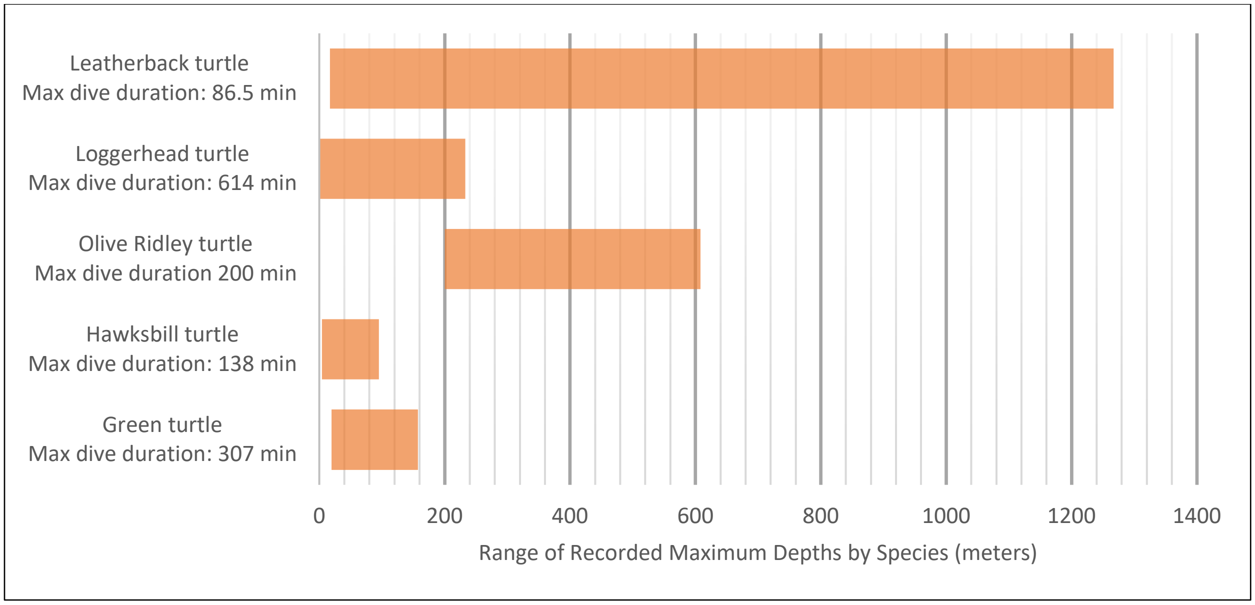
Additional species profiles and information on the biology, life history, species distribution, and conservation of reptile species can also be found on the following organizations:

- NMFS Office of Protected Resources (includes sea turtle species distribution maps),
- USFWS Ecological Services Field Office and Region Offices (for sea turtle nesting habitat and general locations of nesting beaches),
- Ocean Biogeographic Information System-Spatial Ecological Analysis of Megavertebrate Populations (known as OBIS-SEAMAP) species profiles,
- International Union for Conservation of Nature, Marine Turtle Specialist Group, and
- State resource agencies (specifically, Hawaii Division of Land and Natural Resources).

Detailed information about threats to sea turtles and life history information can be found in the ESA listing documentation and their recovery plans (44 *FR* 75074; 52 *FR* 21059; 72 *FR* 13027; (U.S. Fish and Wildlife Service, 1999).

C.7.1.1 Dive Behavior

Sea turtle dive depth and duration varies by species, the age of the animal, the location of the animal, and the activity (e.g., foraging, resting, and migrating). Regional differences in behavior among the same species may also occur. Hochscheid (2014) collected information on generalized dive profiles, with correlations to specific activities, such as bottom resting, bottom feeding, orientation and exploration, pelagic foraging and feeding, mid-water resting, and traveling during migrations. Dive durations are often a function of turtle size, with larger turtles being capable of diving to greater depths and for longer periods (Asada et al., 2022). Fukuoka et al. (2022) noted that olive ridley sea turtles exhibit longer dive times in warmer ocean temperatures. The diving behavior of a particular species or individual has implications for mitigation, monitoring, and developing sound conservation strategies. Figure C-33 presents the ranges of maximum dive depths for each sea turtle species found in the Study Area.

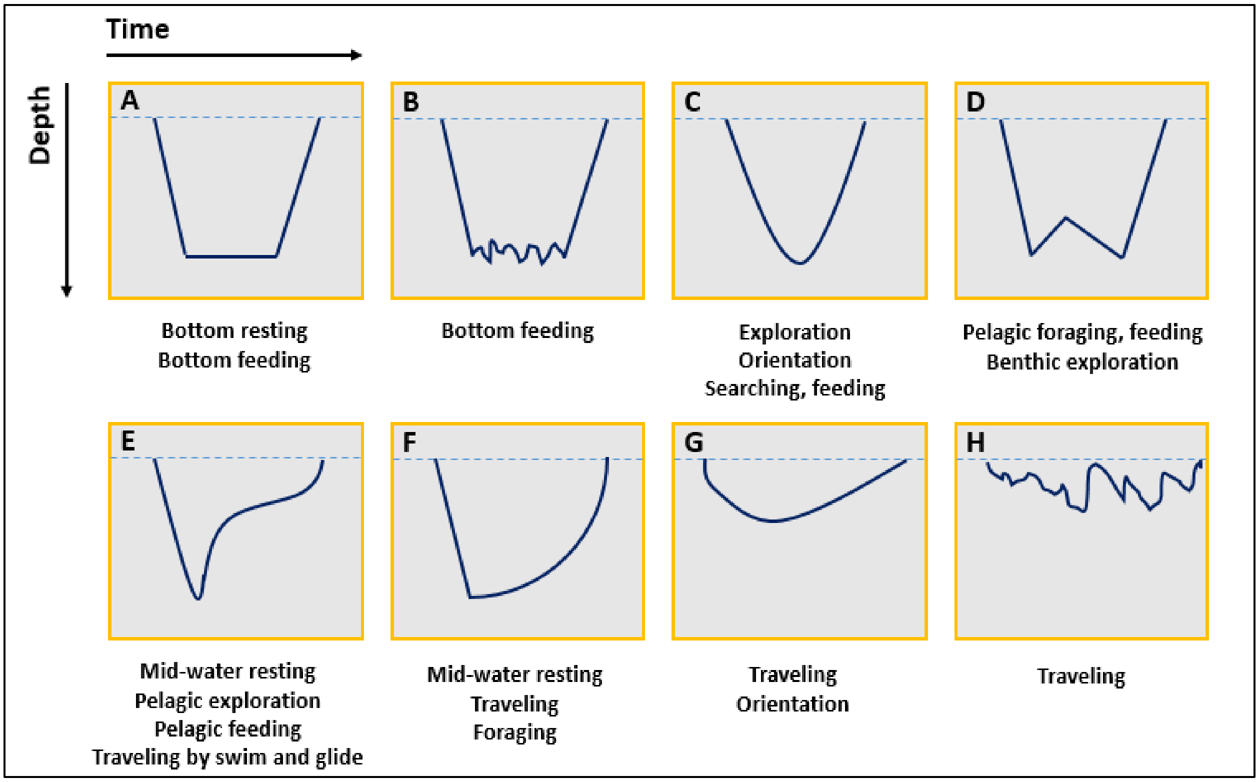


Sources: Hochscheid (2014); Sakamoto et al. (1993); Rice and Balazs (2008); Gitschlag (1996); Salmon et al. (2004).

Figure C-33: Dive Depth and Duration Summaries for Sea Turtle Species

Hochscheid (2014) also collected information on generalized dive profiles, with correlations to specific activities, such as bottom resting, bottom feeding, orientation and exploration, pelagic foraging and feeding, mid-water resting, and traveling during migrations. Generalized dive profiles compiled from 11 different studies by Hochscheid (2014) show eight distinct profiles tied to specific activities (Hochscheid, 2014). These profiles and activities are shown in Figure C-34.

Little is known about yellow-bellied sea snake diving behavior. Yellow-bellied sea snakes likely forage only in pelagic environments, and are believed to forage on the surface to a depth of 10 m (Brischoux et al., 2016; (Goiran et al., 2020). Cook et al. (2015) implanted temperature-depth loggers on three other sea snake species in New Caledonia. Logging 1,850 dives, nearly all dives were less than 30 m deep, with an average dive depth of approximately 11 m. A maximum dive duration was approximately 124 minutes.



Sources: Hochscheid (2014); Rice and Balazs (2008), Sakamoto et al. (1993), Houghton et al. (2003), Fossette et al. (2007), Salmon et al. (2004), Hays et al. (2004); Southwood et al. (1999).

Notes: Profiles A-H, as reported in the literature and compiled by Hochscheid (2014). The depth and time arrows indicate the axis variables, but the figure does not represent true proportions of depths and durations for the various profiles. In other words, the depths can vary greatly, but behavioral activity seems to dictate the shape of the profile. Profiles G and H have only been described for shallow dives (less than 5 m).

Figure C-34: Generalized Dive Profiles and Activities Described for Sea Turtles

C.7.1.2 Hearing and Vocalization

Refer to Appendix D for a summary and details regarding the hearing and vocalization of reptiles.

C.7.1.3 General Threats

C.7.1.3.1 Water Quality

Water quality in sea turtle habitats can be affected by a wide range of activities. The potential for energy exploration and extraction activities to degrade nearshore and off-shore habitats are discussed in Section C.8.1.5.2 (Commercial Industries). Marine debris in sea turtle habitats is discussed in Section C.8.1.5.6 (Marine Debris). Chemical pollution and impacts on water quality is also of great concern, although its effects on reptiles are just starting to be understood in marine organisms (Aguilar de Soto et al., 2008; Jepson et al., 2016; Law et al., 2014; National Marine Fisheries Service, 2011e, 2014b; Ortmann et al., 2012; Peterson et al., 2015). Oil and other chemical spills are a specific type of ocean contamination that can have damaging effects on some sea turtle and other marine reptile species directly through exposure to oil or chemicals and indirectly due to pollutants' impacts on prey and habitat quality. Ingested plastics, discussed in more detail in Section C.8.1.5.6 (Marine Debris), can also release toxins, such as bisphenol-A (commonly known as "BPA") and phthalates, and organisms may absorb heavy metals from the ocean and release those into tissues (Fukuoka et al., 2016; Teuten et al., 2007). Life stage, geographic location relative to concentrations of pollutants, and feeding preference affects the severity of impacts on reptiles associated with chemical pollution in the marine environment. Exposure to pollutants may reduce turtle immune system responses, making them more susceptible to disease (Spotila, 2004).

Within the Study Area, sea snakes are primarily pelagic, and only occur close to shore in more tropical environments outside of the Study Area. In these locations, sea snakes are likely more susceptible to water quality degradation, which may decrease prey availability.

C.7.1.3.2 Commercial and Recreational Industries

One comprehensive study estimates that worldwide, 447,000 sea turtles are killed each year from bycatch in commercial fisheries around the world (Wallace et al., 2010). Lewison et al. (2014) compared bycatch using three different gear types (longline, gillnet, and trawling nets) for sea turtles, marine mammals, and seabirds. Sea turtles were most susceptible to bycatch, with the Mediterranean and waters off the Atlantic coast of South America as the two fisheries reporting the highest number of sea turtle mortalities (primarily through trawling) (Lewison et al., 2014). In U.S. fisheries, Finkbeiner et al. (2011) estimate that bycatch resulted in 71,000 sea turtle deaths per year prior to effective regulations that protect sea turtles (e.g., regulations adopted since the mid-1990s in different U.S. fisheries for turtle exclusion devices). Current mortality estimates are 94 percent lower (4,600 deaths) than pre-regulation estimates (Finkbeiner et al., 2011). The trend in bycatch reductions continues throughout the Study Area. For example, Eguchi et al. (2018) determined that current restrictions in West Coast fisheries (e.g., time-area closures for West Coast drift gill net fishery) have been effective and suggested that if the fixed time-area closure regulation existed in the 1990s, 18 of 19 observed bycatch events in this fishery could have been avoided (Eguchi et al., 2018).

Large-scale commercial exploitation also contributes to global decline in marine turtle populations. Currently, 42 countries and territories allow direct take of turtles and collectively take in excess of 42,000 turtles per year, the majority of which (greater than 80 percent) are green sea turtles (Humber et al., 2014). Illegal fishing for turtles and nest harvesting also continues to be a major cause of sea turtle mortality, both in countries that allow sea turtle take and in countries that outlaw the practice (Lam et al., 2011; Maison et al., 2010). For example, Humber et al. (2014) estimated that in Mexico 65,000 sea turtles have been illegally harvested since 2000. The authors, however, noted a downward trend of legal and illegal direct takes of sea turtles over the past three decades—citing a greater than 40 percent decline in green sea turtle take since the 1980s, a greater than 60 percent decline in hawksbill and leatherback take, and a greater than 30 percent decline in loggerhead take (Humber et al., 2014).

Boat strike has been identified as one of the important mortality factors in several nearshore turtle habitats worldwide. Precise data are lacking for sea turtle mortalities directly caused by ship strikes; however, live and dead turtles are often found with deep cuts and fractures indicative of collision with a boat hull or propeller (Hazel et al., 2007; Lutcavage & Lutz, 1997). For example, scientists in Hawaii reported that 2.5 percent of green sea turtles found dead on the beaches between 1982 and 2003 had been killed by boat strike (Chaloupka et al., 2008), and in the Canary Islands, 23 percent of stranded sea turtles showed lesions from boat strikes or fishing gear (Oros et al., 2005). Denkinger et al. (2013) reports that boat strikes in the Galapagos Islands were most frequent at foraging sites close to a commercial and tourism port.

Onshore development can lead to nesting habitat loss or habitat degradation. Construction activities can facilitate erosion or inhibit natural sediment deposition to form beaches. Once facilities are operational, artificial lighting, noise, and other stressors can degrade nesting habitats (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2011; Seminoff et al., 2015). Two utility-grade offshore wind projects are in the early planning stages for Hawaii (Smith et al., 2015). Projects generating electricity in offshore areas may also use wave generation technologies. While no projects are planned for West Coast states, waters off of Oregon and Washington have the most potential for wave generation, with a targeted installed capacity of 500 megawatts by 2025 (Parkinson et al., 2015). These early individual projects will not likely harm sea turtles or disrupt behaviors because of their northern location, but an increasing trend in offshore energy development may present a cumulative threat to sea turtles in nearshore environments with higher sea turtle concentrations. The anticipated increase in renewable energy development in coastal waters and deeper sites on the continental shelf will require increased vessel traffic, seismic surveys, and possibly pile driving activities for the turbine footings (Pacific Fishery Management Council, 2011), all of which may potentially stress sea turtles and their habitats.

The main threat to sea snakes globally is fisheries bycatch. Milton (2001) determined that the impact is relatively low, with prawn fisheries presenting the highest risk to sea snakes.

C.7.1.3.3 Disease and Parasites

Fibropapillomatosis is a disease of sea turtles that results in the production of tumors, both external and internal, that are considered benign, but may obstruct crucial functions, such as swimming, feeding, sight, and buoyancy, and can lead to death (Balazs, 1986; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 1991; Patrício et al., 2016; Work & Balazs, 2013). The disease was first noticed in 1928, and was not observed again until the 1970s (McCorkle, 2016). The disease shows the highest prevalence among green sea turtles (Patrício et al., 2016), with rapid spread of the disease through the 1980s, becoming an epidemic in both Florida and Hawaii in green sea turtle populations (McCorkle, 2016; Work & Balazs, 2013). By 1995 the concentration of disease in the population reached its climax and has showed a decline in prevalence since (Patrício et al., 2016).

Edmonds et al. (2016) lists 16 parasites known to occur in sea turtles, with the most common and significant (in terms of impacts on health) being blood flukes and flatworms (Watson et al., 2017). Some of the common external parasites found on sea turtles include leeches and a number of different species that reside on the shell called epibiota (Suzuki et al., 2014). Leeches are usually seen around where the flippers attach to the rest of the body. Parasitic isopods (e.g., sea lice) can attach themselves to sea turtle soft tissue on the outside and within the mouth (Júnior et al., 2015).

There is no available information regarding disease of sea snakes and parasites that infect internal organs or external surfaces of sea snakes.

C.7.1.3.4 Invasive Species

Invasive species have been shown to have both harmful and beneficial impacts on sea turtles. Impacts on sea turtles associated with invasive species primarily concern nest predation and prey base (Stokes et al., 2024). Nests and eggs in the Northwestern Hawaiian Islands are at low risk of predation, but eggs deposited on beaches in the main Hawaiian Islands may be consumed by a variety of introduced species (e.g., mongooses, rats, feral dogs and cats, pigs, ants). In foraging grounds, sea turtles have been shown to adapt their foraging preferences for invasive seagrass and algae. Becking et al. (2014) showed green sea turtle foraging behavior shift to consumption of *Halophila stipulacea*, a rapidly spreading seagrass in the Caribbean. In Hawaii, green sea turtles in Kaneohe Bay have modified their diets over several decades to include seven non-native species (*Acanthophora spicifera, Hypnea musciformis, Gracilaria salicornia, Eucheuma denticulatum, Gracilaria tikvahiae, Kappaphycus striatum,* and *Kappaphycus* *alvarezii*), with non-native algae accounting for over 60 percent of sea turtle diet (Russell & Balazs, 2015).

There is no information available on the potential impact of invasive species on sea snakes.

C.7.1.3.5 Climate Change

Sea turtles are particularly susceptible to climate change effects because their life history, physiology, and behavior are extremely sensitive to environmental temperatures (Blechschmidt et al., 2020; Fuentes et al., 2013; Patel et al., 2021; São Miguel et al., 2022). The rate of climate change experienced globally in recent decades may compromise sea turtles' survival. Most notable factors include ocean temperature increases and storm surge affecting nesting beaches. Oceanic temperature increases impact life history characteristics, such as temperature-dependent sex determination, late maturity and migratory abilities of sea turtles (São Miguel et al., 2022). Climate change models predict sea level rise and increased intensity of storms and hurricanes in tropical sea turtle nesting areas (Patino-Martinez et al., 2008). These factors could significantly increase beach inundation and erosion, thus affecting water content of sea turtle nesting beaches and potentially inundating nests (Pike et al., 2015). Climate change may negatively impact turtles in multiple ways and at all life stages. While rising temperatures may initially result in increased female population sizes, the lack of male turtles will likely impact the overall fertility of females in the population (Jensen et al., 2018). For example, breeding male sea turtles show strong natal philopatry (the tendency for animals to return to their birth places to mate) (Roden et al., 2017; Shamblin et al., 2015). With fewer available breeding males, it is unlikely that available males from other locations would interact with females in male-depleted breeding areas (Jensen et al., 2018).

Adaption strategies to protect coastal infrastructure are an anticipated response to rising sea levels. These activities may include shoreline stabilization projects and infrastructure hardening, which could contribute to the loss of nesting habitat. Shoreline stabilization may hold in place beach sediments in a specific location; however, the disruption of onshore currents can reduce the beach replenishment of sediments further away (Boyer et al., 1999; Fish et al., 2008).

Climate change may increase the likelihood of sea snakes moving into locations outside of their normal range. Although recent sightings of sea snakes appear to be correlated with El Niño events, it is reasonable to assume that warming oceanic trends may facilitate range expansion Brischoux et al. (2016).

C.7.1.3.6 Marine Debris

Ingestion of marine debris can cause mortality or injury to sea turtles, with adverse effects resulting from blocking of gastro-intestinal tracts and succumbing to toxicity from harmful chemicals (Sinaei et al., 2021). The United Nations Environment Program estimates that approximately 6.4 million tons of anthropogenic debris enters the marine environment every year (United Nations Environmental Program, 2005). This estimate, however, does not account for cataclysmic events, such as the 2011 Japanese tsunami estimated to have generated 1.5 million tons of floating debris (Murray et al., 2015). Plastic is the primary type of debris found in marine and coastal environments, and plastics are the most common type of marine debris ingested by sea turtles (Schuyler et al., 2014). Sea turtles can mistake debris for prey; one study found 37 percent of dead leatherback sea turtles to have ingested various types of plastic (Mrosovsky et al., 2009), and Narazaki et al. (2013) noted an observation of a loggerhead exhibiting hunting behavior on approach to a plastic bag, possibly mistaking the bag for a jelly fish. Even small amounts of plastic ingestion can cause an obstruction in a sea turtle's digestive track and mortality (Bjorndal, 1997; Bjorndal et al., 1994), and hatchlings are at risk for ingesting small plastic fragments.

Ingested plastics can also release toxins, such as BPA and phthalates, or absorb heavy metals from the ocean and release those into tissues (Fukuoka et al., 2016; Teuten et al., 2007). Life stage and feeding preference affects the likelihood of ingestion. Sea turtles living in oceanic or coastal environments and feeding in the open ocean or on the seafloor may encounter different types and densities of debris and may therefore have different probabilities of ingesting debris. In 2014, Schuyler et al. (2014) reviewed 37 studies of debris ingestion by sea turtles, showing that young oceanic sea turtles are more likely to ingest debris (particularly plastic), and that green and loggerhead sea turtles were significantly more likely to ingest debris than other sea turtle species.

Within the Study Area, sea snakes are primarily pelagic, with fish as their primary diet. Further, sea snakes rely on visual cues from fish during hunting activities. With fish as their primary dietary component, mistaking marine debris for a prey item is not likely.

C.7.2 Endangered Species Act-Listed Species

C.7.2.1 Green Turtle (*Chelonia mydas*)

C.7.2.1.1 Status and Management

The green sea turtle was first listed under the ESA in 1978 (43 FR 32800 published July 28, 1978). In 2016, NMFS and USFWS reclassified the species into 11 “DPSs,” which maintains federal protections while providing a more tailored approach for managers to address specific threats facing different populations (see 81 FR 20058 published April 6, 2016). The geographic areas that include these DPSs are: (1) North Atlantic Ocean, (2) Mediterranean Sea, (3) South Atlantic Ocean, (4) Southwest Indian Ocean, (5) North Indian Ocean, (6) East Indian Ocean – West Pacific Ocean, (7) Central West Pacific Ocean, (8) Southwest Pacific Ocean, (9) Central South Pacific Ocean, (10) Central North Pacific Ocean, and (11) East Pacific Ocean.

The Central North Pacific DPS is defined as green turtles originating from the Central North Pacific Ocean, including those hatching from nests on the beaches within the Hawaiian Archipelago and those occurring at Johnston Atoll. The Central North Pacific DPS is listed as threatened. The East Pacific DPS, listed as threatened, is defined as green turtles originating from the eastern Pacific Ocean, including those hatching from nests on the beaches in Mexico, Costa Rica, and Ecuador and foraging off the coast of California (88 FR 46376, July 19, 2023). Only the Central North Pacific and East Pacific Ocean DPSs occur within the Study Area. Four regional genetic stocks have been identified for the East Pacific DPS; however, stocks likely mix at foraging areas (81 FR 20058, April 6, 2016).

Critical Habitat

On July 19, 2023, NMFS issued a proposed rule for critical habitat designation (88 FR 46376) containing four essential features, three of which, overlap with the Central North Pacific DPS and East Pacific, and Navy training and testing activities. The reproductive essential feature proposed by NMFS for the Central North Pacific DPS extends “from the MHW to 20 m depth, sufficiently dark and unobstructed nearshore waters adjacent to nesting beaches proposed as critical habitat by USFWS… to allow for the transit, mating, and interesting of reproductive individuals and the transit of post-hatchlings.” Navy activities also overlap with the proposed migratory essential feature for the East Pacific DPS, which ranges between Mexico and San Diego Bay “from the MHW line to 10 km offshore, sufficiently unobstructed corridors that allow for unrestricted transit between foraging and nesting areas for reproductive individuals.” The proposed benthic foraging/resting essential features for the Central North Pacific DPS and East Pacific DPS also overlap Navy training and testing activities and were proposed “from the MHW to 20 m depth, underwater refugia (rocks, reefs, and troughs) and food resources (i.e., seagrasses, macroalgae, and/or invertebrates) of sufficient condition, distribution, diversity, abundance, and density necessary to support survival, development, growth, and/or reproduction" (88 FR 46376 July 19, 2023).

Within the Study Area, NMFS is proposing critical habitat for the green sea turtle within waters under the jurisdiction of JBPHH (Pearl Harbor Naval Defensive Sea Area, Naval Special Warfare training areas at Pearl City Peninsula and Lima Landing, Puuloa Underwater Range, And Barbers Point Underwater Range and Ewa Training Minefield), PMRF, and Navy Training Ranges in Southern California (Silver Strand Training Complex and Coastal San Diego Training Ranges, select areas within San Diego Bay, and Naval Weapons Station Seal Beach).

Recovery Goals

See the 1998 and 1991 recovery plans for the Pacific, East Pacific and Atlantic populations of green sea turtles for complete down-listing/delisting criteria for recovery goals for the species (NMFS and USFWS 1991; NMFS and USFWS 1998b). Broadly, recovery plan goals emphasize the need to protect and manage nesting and marine habitat, protect and manage populations on nesting beaches and in the marine environment, increase public education, and promote international cooperation on sea turtle conservation topics.

C.7.2.1.2 Habitat and Distribution

Green sea turtles are found throughout the world, nesting in 80 countries and living in the coastal areas of 140 countries (National Marine Fisheries Service, 2024f). After emerging from the nest, green sea turtle hatchlings swim to offshore areas where they float passively in major current systems; however, laboratory and modeling studies suggest that dispersal trajectories might also be shaped by active swimming (Putman & Mansfield, 2015). Post-hatchling green sea turtles forage and develop in floating algal mats habitats of the open ocean. At the juvenile stage (estimated at five to six years), they leave the open-ocean habitat and retreat to protected lagoons and open coastal areas that are rich in seagrass or marine algae (Bresette et al., 2006), where they will spend most of their lives (Bjorndal & Bolten, 1988). The optimal developmental habitats for late juveniles and foraging habitats for adults are warm shallow waters (3–5 m), with abundant submerged aquatic vegetation and close to nearshore reefs or rocky areas (Holloway-Adkins, 2006; Seminoff et al., 2003a). Climate change and ocean warming trends may impact the habitat and range of this species over time (Fuentes et al., 2013). These impacts apply to all sea turtle species and are discussed in Section C.8.1.5.5 (Climate Change).

Green sea turtles nest on beaches within the Hawaii Study Area, while they feed and migrate throughout all waters of the Study Area. Green sea turtles likely to occur in the Study Area come from eastern Pacific Ocean and Hawaiian nesting populations. There are very few reports of turtles from southern Pacific Ocean populations occurring in the northern Pacific Ocean (Limpus et al., 2009; Seminoff et al., 2015).

Migratory routes within the open ocean are unknown. The main source of information on distribution in the Study Area comes from catches in U.S. fisheries. About 57 percent of green sea turtles (primarily adults) captured in longline fisheries in the North Pacific Subtropical Gyre and North Pacific Transition Zone come from the Eastern Pacific DPS, while 43 percent are from the North Central Pacific DPS. These findings suggest that green sea turtles found on the high seas of the western and central Pacific Ocean are from these two populations. Though few observations of green sea turtles in the offshore waters along the U.S. Pacific coast have been verified, their occurrence within the nearshore waters from Baja

California to Alaska indicates a presence in waters off of California (Stinson, 1984), including San Diego Bay (Turner-Tomaszewicz & Seminoff, 2012; U.S. Department of the Navy, 2013d).

In Hawaii, green sea turtles enter the reproductive stage upon achieving sizes of 36 in. in carapace length at about 30–35 years of age (Spotila, 2004). Male green sea turtles may mate with females on foraging grounds, along migratory pathways, and off nesting beaches (National Marine Fisheries Service, 2024f). Mating occurs from March to June, and nesting occurs from May to September throughout the Hawaiian Archipelago. In captive green turtles, the average time from mating to nesting is 34.7 days (Wood & Wood, 1980). Green sea turtle clutches may contain 110 eggs (Spotila, 2004). Egg deposition takes place at night prior to sunrise (one reason to limit artificial lighting), but females may begin excavating a nest site as early as 2 hours prior to sunset (Balazs, 1980). During a season, females lay up to nine clutches (mean = 1.8 clutches) with an inter-nesting interval of 11 to 18 days (mean = 13 days) (Balazs et al., 2015). During the internesting interval, males and females regularly occupy neritic waters adjacent to nesting beaches and the undersides of reefs as refugia (Balazs, 1980). Females appear to stay relatively close to shore during the internesting interval; the maximum diving depth recorded during that time was 12.8 m (Balazs, 1980). Nesting trends over the last 40 years have increased approximately 4.8 to 5.4 percent per year (depending on the study) (Balazs et al., 2015; Seminoff et al., 2015).

Green sea turtle distribution is inconsistent and patchy throughout Pearl Harbor, however turtles occur in higher numbers around the entrance channel and within the West Loch channel of the harbor and use “caves” in benthic habitat throughout Pearl Harbor (Teresa, 2021). (Teresa, 2021) In the spring of 2010, two green sea turtles nested at PMRF for the first time in more than a decade. The number of nests observed at this location has increased over the years with six successful nests producing 468 hatchlings (Hanser et al., In Prep.). Green sea turtles are also common at all three landing beaches of U.S. Marine Corps Base Hawaii in Kaneohe Bay, where they forage in the shallow water seagrass beds (Marine Corps Base Hawaii, 2011; Martínez-Abraín, 2008), with successful the first known successful hatching occurring in August 2010 (Marine Corps Base Hawaii, 2011).

The Navy conducts aerial surveys for marine mammals in Hawaii annually as a requirement under the Navy’s LOA in compliance with MMPA for at-sea training in the Hawaii Range Complex. Sea turtles are observed and recorded opportunistically while surveying for marine mammals. Turtles can be spotted from a plane or helicopter during surveys. Based on these methods, sea turtle densities were calculated for each island that was surveyed. In 88 FR 46376 (July 19, 2023), NMFS discussed foraging green sea turtle density estimates for each island. The lowest density estimates for those islands was 0.10 green turtles/km at Lanaʻi (Becker et al., 2019). Therefore, estimates greater than or equal to 0.10 green turtles/km (Becker et al., 2019) constitute high density within the Hawaiian Archipelago. Based on this threshold, high densities of foraging/resting green turtles occur in waters off the Island of Hawaiʻi (0.27 green turtles/km), Maui (0.24), Molokaʻi (0.13), LanaʻI (0.10), Oʻahu (0.11), and Kauaʻi (0.18). Low densities (less than 0.10 green turtles/km) of foraging/resting green turtles occur in waters off Niʻihau and throughout the PMNM (Becker et al., 2019).

The green sea turtle is not known to nest anywhere on the U.S. West Coast, but ranges widely in nearshore waters as far as British Columbia (National Marine Fisheries Service & U. S. Fish and Wildlife Service, 2007) with high concentrations in the subtropical coastal waters of southern Baja California, Mexico, and Central America (Chaloupka et al., 2004).

In the proposed rule (88 FR 46376, July 19, 2023), NMFS included a summary of research known about green sea turtles within the East Pacific DPS foraging in southern Californian waters, including inside San Diego Bay and outside along the coastal ranges of Silver Strand Training Complex. These areas are known to support a resident population of benthic foraging juvenile and adult green turtles reported by MacDonald et al. (2013). Juveniles comprise the majority of the California population of the East Pacific DPS, which is expected due to recent increases in abundance (Tomaszewicz et al., 2022). Turtles forage on seagrass in the South and Central Bays (MacDonald et al., 2012; MacDonald et al., 2013), which have dense seagrass beds that have expanded to several thousand acres during the past several years; however, the heavily developed jetties on the eastern shores of the central portion of San Diego Bay do not appear to be used, likely attributed to the heavy boat traffic.

Green sea turtles rest after in underwater refugia after foraging. In the winter and in some locations, turtles use refugia during the day, suggesting resting between diurnal foraging activity (Crear et al., 2017; MacDonald et al., 2013). Generally, adults and benthic-foraging juveniles occupy small home ranges that include foraging resources and underwater refugia. For example, green turtles acoustically tracked in San Diego Bay occupied areas of 2.09 to 8.70 km^2, remaining in one or two core areas more than half the time (MacDonald et al., 2012). Within the action area, Naval Weapons Station Seal Beach is considered one of the most important foraging and resting areas for green turtles in California, given that use of the area is frequent, and many turtles use the areas (National Marine Fisheries Service, 2024f). In 88 FR 46376 (July 19, 2023), NMFS discusses that food resources and refugia are typically found located in adjacent areas of San Diego Bay, and turtles must move between these areas. Refugia are likely to be found nearby the areas noted for sea turtle foraging within San Diego Bay.

In a satellite tracking study involving 25 green turtles within San Diego Bay, juveniles remained within the bay foraging, while four of the five adult turtles that left the bay migrated south to Mexico, and the fifth turtle migrated north to other foraging areas. Three were tracked to nesting beaches within Mexico, with one returning to San Diego Bay after nesting; the fourth turtle was a male that likely migrated to Mexican waters to mate (Eguchi et al., 2020).

The East Pacific DPS of the green sea turtle exhibits moderate levels of green turtle nesting abundance (>20,000 nesting females) occurring in three primary regions, with Mexico having the largest number of nesting females at several sites (13,664 nesting females), followed by the Galápagos, Ecuador (3,603 nesting females), and Costa Rica (2,826 nesting females distributed among 26 nesting sites). As discussed, some green turtles nesting on beaches in Mexico forage in the waters of California, thus requiring migration to complete their life cycle. The foraging population in California is small and has been increasing since the early 2000s, likely as a result of increases in nesting observed at Mexico nesting beaches, which may be attributed to nesting beach protections (National Marine Fisheries Service, 2024f). Twenty-five years of monitoring at the DPS’s largest nesting aggregation, Michoacán, Mexico, suggest an increasing trend since the population’s low point in the mid-1980s. The broad latitudinal range may provide a benefit to the East Pacific DPS of the green sea turtle in the face of climate change.

In 88 FR 46376 (July 19, 2023), NMFS discussed that reproductive individuals are agile and able to move around minor structures within migratory corridors without using excessive time or energy. However, obstructions may impede their migration in narrow, coastal corridors.

Ocean waters off Southern California and northern Baja California are also designated as areas of occurrence because of the presence of rocky ridges and channels and floating kelp habitats suitable for green sea turtle foraging and resting (Stinson, 1984); however, these waters are often at temperatures below the thermal preferences of this primarily tropical species and turtles found in these waters are likely transiting.

C.7.2.1.3 Population Trends

The Central North Pacific DPS populations have increased since Federal and State protections were implemented in the mid-1970s (Dentlinger, 2023). The Central North Pacific DPS has seen an estimated 4.8 percent annual increase in nesting activity over the last 40 years (Seminoff et al., 2015). In-water abundance trends appear to also be increasing. A significant increase in catch per unit effort of green sea turtles was seen from 1982 to 1999 during bull-pen fishing conducted at Pala'au, Molokai, with anecdotal indications of increased abundance with more green sea turtle basking activity observed in the main Hawaiian Islands (Balazs & Chaloupka, 2006).

The East Pacific DPS also shows an increasing population trend (Massey et al., 2023; Tomaszewicz et al., 2022). This observed increase may have resulted from the onset of nesting beach protection in 1979—as is suggested by the similarity in timing between the onset of beach conservation and the age to maturity for green sea turtles along Pacific nesting beaches of Mexico (Seminoff et al., 2015).

C.7.2.1.4 Population Threats

In addition to the general threats, damaged seagrass beds and declines in seagrass distribution can reduce foraging habitat for green sea turtles (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 1991; Seminoff et al., 2015; Williams, 1988). Major threats to green sea turtle recovery includes harvesting of eggs and the killing of green turtles for their meat or to shells for the wildlife trafficking trade. Bycatch in commercial and recreational fishing gear, vessel strike, coastal development impacts to nesting habitat, are also threats to green turtle recovery (National Marine Fisheries Service, 2024f). Green sea turtles are susceptible to the disease fibropapillomatosis, which causes tumor-like growths (fibropapillomas) resulting in reduced vision, disorientation, blindness, physical obstruction to swimming and feeding, increased susceptibility to parasites, and increased susceptibility to entanglement (Balazs, 1986; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 1991; Patrício et al., 2016; Work & Balazs, 2013). Some populations (e.g., the Florida population) have begun to show resistance to the disease, but it remains an issue for others, such as Pacific populations, and Hawaii's green sea turtles in particular (Chaloupka et al., 2009; Seminoff et al., 2015). Other factors, such as increased stressors and selection of healthy turtles during illegal poaching activities, may increase susceptibility of turtles (Patrício et al., 2016).

C.7.2.2 Hawksbill Sea Turtle (*Eretmochelys imbricata*)

C.7.2.2.1 Status and Management

The hawksbill sea turtle is listed as endangered under the ESA (35 FR 8491). While the current listing as a single global population remains valid, data may support separating populations at least by ocean basin under the DPS policy (Seminoff et al., 2015). The most recent status review document was released in 2013 by the NMFS and USFWS (Hill et al., 2017).

There is no critical habitat designated for hawksbill sea turtles in the Study Area.

Recovery Goals

See the 1992 Recovery Plan for the U.S. Caribbean, Atlantic and Gulf of Mexico (NMFS and USFWS 1993) and the 1998 Recovery Plan for the U.S. Pacific populations (NMFS and USFWS 1998c) of hawksbill sea turtles, for complete down listing/delisting criteria for each of their respective recovery goals. The following items were the top recovery actions identified to support in the recovery plans:

1. Identify important nesting beaches.
2. Ensure long-term protection and management of important nesting beaches.
3. Protect and manage nesting habitat; prevent the degradation of nesting habitat caused by seawalls, revetments, sand bags, other erosion-control measures, jetties and breakwaters.
4. Identify important marine habitats; protect and manage populations in marine habitat.
5. Protect and manage marine habitat; prevent the degradation or destruction of important [marine] habitats caused by upland and coastal erosion.
6. Prevent the degradation of reef habitat caused by sewage and other pollutants.
7. Monitor nesting activity on important nesting beaches with standardized index surveys.
8. Evaluate nest success and implement appropriate nest-protection on important nesting beaches.
9. Ensure that law-enforcement activities prevent the illegal exploitation and harassment of sea turtles and increase law-enforcement efforts to reduce illegal exploitation.
10. Determine nesting beach origins for juveniles and subadult populations.

C.7.2.2.2 Habitat and Distribution

The hawksbill is the most tropical of the world's sea turtles, rarely occurring above 35° N or below 30° south (Witzell, 1983). After hatching, hawksbill sea turtles may spend 1 to 3 years at sea where they may often be associated with floating vegetation. Upon reaching a size of about 15 in. in length (in the Indo-Pacific), they recruit to nearshore foraging areas for foraging areas. Approximately once every three years, reproductive adults migrate hundreds to thousands of miles between foraging grounds and nesting beaches, and within feeding ground, typically maintain small ranges and return to the same nesting locations at night (Spotila, 2004) . Van Houtan et al. (2016) suggest that hatchlings within the HCTT Study Area may move to coastal habitats and nearshore foraging grounds more quickly. Within the Study Area, nesting occurs only in the Hawaiian Islands, with known nesting activities only at Hawaii, Maui, and Molokai Islands (Brunson et al., 2022).

Less is known about the hawksbill's oceanic stage, but it is thought that neonates live in the oceanic zone where water depths are greater than 200 m. Distribution in the oceanic zone may be influenced by surface gyres (Gaos, 2011; Leon & Bjorndal, 2002).

Juveniles and adults share the same foraging areas, including tropical nearshore waters associated with coral reefs, hard bottoms, or estuaries with mangroves (Musick & Limpus, 1997). In nearshore habitats, resting areas for late juvenile and adult hawksbills are typically in deeper waters, such as sandy bottoms at the base of a reef flat (Houghton et al., 2003). As they mature into adults, hawksbills move to deeper habitats and may forage to depths greater than 90 m. During this stage, hawksbills are seldom found in waters beyond the continental or insular shelf unless they are in transit between distant foraging and nesting grounds (Renaud et al., 1996). Ledges and caves of coral reefs provide shelter for resting hawksbills during both day and night, where an individual often inhabits the same resting spot. Hawksbills are also found around rocky outcrops and high-energy shoals, where sponges are abundant, and in mangrove-fringed bays and estuaries. Adults migrate hundreds to thousands of miles between foraging grounds and nesting beaches. Females may lay 3 to 5 clutches of eggs within a nesting season, laying one clutch every 13 to 16 days (Spotila, 2004).

Hawksbill sea turtles have a varying diet and feeding habitat preference throughout different lifestages. Post-hatchling hawksbills feed on algae in floating habitats (e.g., *Sargassum*) in the open ocean (Plotkin & Amos, 1998; Van Houtan et al., 2016). During the later juvenile stage, hawksbills are considered omnivorous, feeding on sponges, sea squirts, algae, molluscs, crustaceans, jellyfish, and other aquatic invertebrates (Bjorndal, 1997). Older juveniles and adults are more specialized, feeding primarily on sponges, which compose as much as 95 percent of their diet in some locations (Meylan, 1988; Witzell, 1983). As adults, Hawksbill sea turtles fill a unique ecological niche in marine and coastal ecosystems, supporting the natural functions of coral reefs by keeping sponge populations in check, which may otherwise compete for space with reef-building corals (Hill, 1998; Leon & Bjorndal, 2002).

C.7.2.2.3 Population Trends

Gaos et al. (2021) analyzed 30 years of nesting data within the Hawaiian Islands (between 1998 and 2018) and determined that nesting trends had historic decreases through 2006, with slight annual increases occurring for the remainder of the monitoring period. Van Houtan et al. (2016) also noted increases around the same time as observed by Gaos et al. (2021). Hawksbills in the eastern Pacific Ocean are probably the most endangered sea turtle population in the world (Gaos & Yañez, 2008). A lack of nesting beach surveys for hawksbill sea turtles in the Pacific Ocean and the poorly understood nature of this species' nesting have made it difficult for scientists to assess the population status of hawksbills in the Pacific (Gaos & Yañez, 2008; Seminoff et al., 2003b). The largest of these regional populations is in the South Pacific Ocean, where 6,000–8,000 hawksbills nest off the Great Barrier Reef (Limpus, 1992).

C.7.2.2.4 Population Threats

In addition to the general threats described in Section C.8.1.5 (General Threats), the greatest threat to hawksbills is harvest for commercial and subsistence use (Van Houtan et al., 2016). Direct harvest of eggs and nesting adult females from beaches, as well as direct hunting of turtles in foraging areas, continues in many countries. International trade of tortoise shells is thought to be the most important factor endangering the species worldwide. The second-most significant threat to hawksbill sea turtles is loss of nesting habitat caused by the expansion of human populations in coastal areas of the world, as well as the increased destruction or modification of coastal ecosystems to support tourism (National Oceanic and Atmospheric Administration, 2024a). Coastal pollution as a result of increased development degrades water quality, particularly coral reefs, which are primary foraging areas for hawksbills. Due to their preference for nearshore areas, hawksbills are particularly susceptible to nearshore fisheries gear such as drift nets, entanglement in gill nets, and capture on fish hooks of fishermen (Gaos, 2011; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 1993). Hawksbills in the North Pacific may occupy a variety of ecosystems, including coastal pelagic waters and shallow reefs in remote atolls, and therefore be exposed to threats specific to these environments (Van Houtan et al., 2016).

C.7.2.3 Olive Ridley Sea Turtle (*Lepidochelys olivacea*)

C.7.2.3.1 Status and Management

Olive ridley sea turtles that nest along the Pacific coast of Mexico are listed as endangered under the ESA, while all other populations are listed under the ESA as threatened (43 FR 32800). Based on genetic data, the worldwide olive ridley population is composed of four main lineages: east India, Indo-Western Pacific, Atlantic, and eastern Pacific Ocean (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2014; Shankar et al., 2004). Most olive ridley sea turtles found in Hawaiian waters are of the eastern Pacific Ocean lineage, with about a third from the Indo-Western Pacific lineage. Off of California, olive ridleys are thought to be within the eastern Pacific Ocean lineage (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2014). There is no critical habitat designated for this species in the Study Area.

Recovery Goals

There has not been a Recovery Plan prepared specifically for olive ridley sea turtles of the breeding populations of the Pacific coast of Mexico. The 1998 Recovery Plan was prepared for olive ridleys found in the U.S. Pacific. Olive ridley sea turtles found in the Pacific could originate from the Pacific coast of Mexico or from another nesting population. As such, the recovery goals in the 1998 Recovery Plan for the U.S. Pacific olive ridley sea turtle can apply to both listed populations. See the 1998 Recovery Plan for the U.S. Pacific olive ridley sea turtles for complete down listing/delisting criteria for their recovery goals. The following items were the recovery criteria identified to consider delisting:

(1) All regional stocks that use U.S. waters have been identified to source beaches based on reasonable geographic parameters.
(2) Foraging populations are statistically significantly increasing at several key foraging grounds within each stock region.
(3) All females estimated to nest annually at source beaches are either stable or increasing for over ten years.
(4) Management plan based on maintaining sustained populations for turtles is in effect.
(5) International agreements in place to protect shared stocks.

C.7.2.3.2 Habitat and Distribution

The olive ridley has a circumtropical distribution, occurring in the Atlantic, Pacific, and Indian Oceans (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2014). In the eastern Pacific, olive ridleys typically occur in tropical and subtropical waters, as far south as Peru and as far north as California, but occasionally have been documented as far north as Alaska. Key arribada beaches include La Flor in Nicaragua, Nancite and Ostinal in Costa Rica, La Marinera and Isla Cañas in Panama, Gahirmatha, Rushikulya, and Devi River in India, and Eilanti in Suriname. Arribada is the common term for large concentrations of nesting activity.

Olive ridley sea turtles are primarily carnivorous. They consume a variety of prey in the water column and on the seafloor, including snails, clams, tunicates, fish, fish eggs, crabs, oysters, sea urchins, shrimp, and jellyfish (Polovina et al., 2004), though olive ridleys may also consume algae (Spotila, 2004). Like other sea turtle species olive ridleys are subject to predation by sharks, fish and sharks on hatchlings, and various land predators on hatchlings (e.g., ants, crabs, birds, and mammals)(Ariano-Sánchez et al., 2020; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2014; Valverde et al., 2012).

Studies from different populations of olive ridley sea turtles show a strong preference for neritic areas (shallow part of the sea near a coast and overlying the continental shelf) (Plot et al., 2015; Polovina et al., 2004; Rees et al., 2016); however, deep water foraging has been documented in the north Pacific, where prey items are scattered and less predictable and migrate widely from nesting locations (Polovina et al., 2004). Comparing olive ridley habitat use in different regions, Plot et al. (2015) suggest that the differing migration patterns observed (i.e., oceanic migrations versus neritic movements) may be attributed to specific environmental conditions of the areas in close proximity to nesting sites.

Olive ridley sea turtles can dive and feed at considerable depths from 80 to 300 m (Chambault et al., 2016; Montero et al., 2016), although only about 10 percent of their foraging time is spent at depths greater than 100 m (Polovina et al., 2002). In the eastern tropical Pacific Ocean, at least 25 percent of their total dive time is spent between 20 and 100 m (Parker et al., 2003). While olive ridley sea turtles are known to forage to great depths, Polovina et al. (2002) found that most dives (approximately 70 percent) were no deeper than 15 m.

Rare instances of nesting occur in the Hawaiian Islands, with the first olive ridley nest documented in 1985 at Paia, Maui. A second nest was recorded in Hilo, Hawaii, in 2002, and a third olive ridley nest was recorded at Marine Corps Base Hawaii in Kaneohe Bay in 2009 (Marine Corps Base Hawaii, 2011).

C.7.2.3.3 Population Trends

The olive ridley is the most abundant sea turtle in the world, with the most recent at-sea estimates of density and abundance providing a population range of 1.15–1.62 million olive ridley sea turtles (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2014). Although this is a dramatic decrease over the past 50 years, where the population from the five Mexican Pacific Ocean beaches was previously estimated at 10 million adults, short-term population trends appear to be increasing overall. The number of olive ridley sea turtles occurring in U.S. territorial waters is believed to be small (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 1998, 2014). At-sea abundance surveys conducted along the Mexican and Central American coasts between 1992 and 2006 provided an estimate of 1.39 million turtles in the region, which was consistent with the increases seen on the eastern Pacific Ocean nesting beaches between 1997 and 2006.

(Ariano-Sánchez et al., 2020; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2014; Valverde et al., 2012)

C.7.2.3.4 Population Threats

Besides the array of threats to sea turtles in general, most of the species-specific threats for olive ridleys in the east Pacific coast population are associated with nesting habitats along the eastern Pacific coast. Lutcavage et al. (1997) note that impacts on nesting habitats for olive ridley sea turtles include construction of buildings and pilings, beach armoring and nourishment, and sand extraction. These activities have increased in many parts of the olive ridley's range and pose threats to major nesting sites in Central America (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2014).

C.7.2.4 Loggerhead Turtle (*Caretta caretta*)

C.7.2.4.1 Status and Management

In 2009, a status review conducted for the loggerhead (the first turtle species subjected to a complete stock analysis) identified nine DPS within the global population (Conant et al., 2009). In a September 2011 rulemaking, the NMFS and USFWS listed five of these DPS as endangered and kept four as threatened under the ESA, effective as of October 24, 2011 (76 FR 58868). The North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Northeast Atlantic Ocean, and Mediterranean Sea DPSs of the loggerhead sea turtle are classified as endangered under the ESA, and the Southeast Indo-Pacific Ocean, Southwest Indian Ocean, Northwest Atlantic Ocean, and South Atlantic Ocean DPS are classified as threatened. Only the North Pacific Ocean DPS occurs within the Study Area; however, mixing is known to occur between other populations in the Pacific and Indian Oceans, enabling a limited amount of gene flow with other DPSs (Gaos, 2011). NMFS and USFWS completed the most recent status review for the North Pacific Ocean DPS in 2020 (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020a).

There is no critical habitat designated for loggerhead sea turtles within the Study Area.

C.7.2.4.2 Habitat and Distribution

Loggerhead sea turtles occur in U.S. waters in habitats ranging from coastal estuaries to waters far beyond the continental shelf (Dodd, 1988; Martin et al., 2020); however, loggerheads are not found in high numbers within the HCTT Study Area. Loggerheads typically nest on beaches close to reef formations and in close proximity to warm currents (Dodd, 1988), preferring beaches facing the ocean or along narrow bays (National Marine Fisheries Service & U. S. Fish and Wildlife Service, 1998; Rice et al., 1984). Most of the loggerheads observed in the eastern North Pacific Ocean are believed to come from beaches in Japan where the nesting season is late May to August. Aschettino et al. (2015) found that most loggerheads that use the Southern California Bight are more similar, using stable isotope analysis, to loggerheads in the Central North Pacific, as opposed to loggerheads that nest in Baja. Migratory routes can be coastal or can involve crossing deep ocean waters (Schroeder et al., 2003). The species can be found hundreds of km out to sea, as well as in inshore areas, such as bays, lagoons, salt marshes, creeks, ship channels, and the mouths of large rivers. Coral reefs, rocky areas, and shipwrecks are often used as feeding areas. The nearshore zone provides crucial foraging habitat, as well as habitat during nesting season and overwintering habitat.

Loggerhead sea turtles are primarily carnivorous in both open ocean and nearshore habitats, although they also consume some algae (Bjorndal, 1997). Diet varies by age class (Godley et al., 1998) and by specializing in specific prey groups dependent on location (Besseling et al., 2015; Biggs et al., 2000). For post hatchlings that tend to be grouped in masses of floating habitats, various diet analyses of gut contents show parts of marine algae, zooplankton, jellyfish, larval shrimp and crabs, and gastropods (Browlow et al., 2016; Burkholder et al., 2004; Carr & Meylan, 1980; Harrison et al., 2021; Morales-Zárate et al., 2021; Richardson & McGillivary, 1991). Both juveniles and adults forage in coastal habitats, where they feed primarily on the bottom, although they also capture prey throughout the water column (Bjorndal, 2003; DiMatteo et al., 2022; Harrison et al., 2021; Morales-Zárate et al., 2021). Adult loggerheads feed on a variety of bottom-dwelling animals, such as crabs, shrimp, sea urchins, sponges, and fish. They have powerful jaws that enable them to feed on hard-shelled prey, such as whelks and conch. During migration through the open sea, they eat jellyfish, molluscs, flying fish, and squid (Besseling et al., 2015; Harrison et al., 2021; Martin et al., 2020; Rice et al., 1984).

Pacific Ocean loggerheads appear to use the entire North Pacific Ocean during development. There is substantial evidence that the North Pacific Ocean stock makes two transoceanic crossings. The first crossing (west to east) is made immediately after they hatch from the nesting beach in Japan, while the second (east to west) is made when they reach either the late juvenile or adult life stage at the foraging grounds in Mexico. Offshore, juvenile loggerheads forage in or migrate through the North Pacific Subtropical Gyre as they move between North American developmental habitats and nesting beaches in Japan. The highest densities of loggerheads can be found just north of Hawaii in the North Pacific Transition Zone (Briscoe et al., 2021; Polovina et al., 2000).

The North Pacific Transition Zone is defined by convergence zones of high productivity that stretch across the entire northern Pacific Ocean from Japan to California (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020a). Within this gyre, the Kuroshio Extension Bifurcation Region is an important habitat for juvenile loggerheads (Polovina et al., 2006). These turtles, whose oceanic phase lasts a decade or more, have been tracked swimming against the prevailing current, apparently to remain in the areas of highest productivity. Juvenile loggerheads originating from nesting beaches in Japan migrate through the North Pacific Transition Zone en route to important foraging habitats in Baja California, and are likely to be found in the Transit Corridor of the Study Area (Bowen et al., 1995).

Seminoff et al. (2014) report that waters off of the southern Baja Peninsula support a high abundance of loggerheads that originate from the Japanese nesting grounds. Capistrán et al. (2023) reported the first record of loggerhead presence in Monterey Bay, California, near the edge of the Monterey Submarine Canyon, sighted by a local kayaker in 2017.

The loggerhead sea turtle is known to occur at sea in the California Study Area but does not nest on Southern California beaches. Loggerhead sea turtles primarily occupy areas where the sea surface temperature is between 59 °F and 77°F (15 °C and 25°C). In waters off of the U.S. West Coast, most records of loggerhead sightings, stranding events, and incidental bycatch have been of juveniles documented from the nearshore waters of southern California. In general, sea turtle sightings increase during the summer, peaking from July to September off southern California and southwestern Baja California.

During El Niño events, foraging loggerheads from Mexican waters may expand their range north into Southern California waters. For this reason, U.S. Pacific Ocean waters east of 120° W longitude are closed to the large mesh drift gillnet fishery targeting swordfish and thresher shark during June, July, and August during a forecast or occurring El Niño event. These waters are considered an area of occurrence during the warm-water period. Allen et al. (2013) conducted stable isotope analysis on loggerheads in both the Southern California Bight and North Pacific Ocean loggerheads and noted strong genetic kinship among these population segments. Loggerheads are generally not found in waters colder than 60.8°F (16°C), so the area north of the 60.8°F (16°C) isotherm is depicted as an area of rare occurrence.

The loggerhead embarks on transoceanic migrations and has been reported as far north as Alaska and as far south as Chile. Loggerheads foraging in and around Baja California originate from breeding areas in Japan (Conant et al., 2009), while Australian stocks appear to migrate to foraging grounds off the coasts of Peru and Chile (Alfaro-Shigueto et al., 2004).

C.7.2.4.3 Population Trends

No loggerhead nesting occurs within the Study Area. The largest nesting aggregation in the Pacific Ocean occurs in southern Japan, where fewer than 1,000 females breed annually (Briscoe et al., 2021; Kamezaki et al., 2003; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020a). Despite historic long-term declines from Japan nesting beaches (50 to 90 percent), nesting populations in Japan have gradually increased since 2000 (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020a), based in part on modeling conducted by . (Morales-Zárate et al., 2021); Seminoff et al. (2014) carried out aerial surveys for loggerhead sea turtles along the Pacific Coast of the Baja California Southern California Bight resulted in 215 loggerhead sea turtle sightings over the course of one month in the fall of 2015 (Eguchi, 2015).

C.7.2.4.4 Population Threats

Loggerheads that occur within the Study Area primarily originate from nesting grounds in Japan and use the North Pacific as migration and foraging grounds. Therefore, species-specific threats are limited to this geographic area. A primary threat to North Pacific loggerheads is the high degree of juvenile and adult mortality off the Baja California Peninsula. As discussed previously, this location is considered a biological hotspot for loggerheads in a location where bycatch and human consumption present significant threats (Fisheries and Oceans Canada, 2011, 2016b; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020a). Mortality associated with shrimp trawls has been a substantial threat to juvenile loggerheads because these trawls operate in the nearshore habitats commonly used by this species. Although shrimping nets have been modified with turtle excluder devices to allow sea turtles to escape, the overall effectiveness of these devices has been difficult to assess (Bugoni et al., 2008; Ellis, 2016). Shrimp trawl fisheries account for the highest number of loggerhead sea turtle fishery mortalities; however, loggerheads are also captured and killed in other trawls, traps and pots, longlines, and dredges (Morales-Zárate et al., 2021; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020a).

C.7.2.5 Leatherback Sea Turtle (*Dermochelys coriacea*)

C.7.2.5.1 Status and Management

The leatherback sea turtle is listed as a single population and is classified as endangered under the ESA (35 FR 8491, January 26, 2012). Although USFWS and NMFS believe the current listing is valid, preliminary information indicates an analysis and review of the species should be conducted under the DPS policy (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2013, 2020b). Recent information on population structure (through genetic studies) and distribution (through telemetry, tagging, and genetic studies) have led to an increased understanding and refinement of the global stock structure (Clark et al., 2010; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020b).

In 2012, NMFS designated critical habitat for the leatherback sea turtle off the western coast of the U.S. (77 FR 4170, January 26, 2012) This designation includes approximately 16,910 mi.^{2} (43,798 km^2) stretching along the California coast from Point Arena to Point Arguello east of the 3,000 m depth contour; and 25,004 mi.^{2} (64,760 km^2) stretching from Cape Flattery, Washington to Cape Blanco, Oregon east of the 2,000 m depth contour. The designated areas comprise approximately 41,914 mi.^{2} (108,558 km^2) of marine habitat and include waters from the ocean surface down to a maximum depth of 262 ft. (80 m) (National Marine Fisheries Service, 2012a). This critical habitat designation overlaps with the California portion of the Study Area. NMFS focused on the known and consistent coastal foraging areas that leatherbacks rely on after long migrations across the Pacific Ocean, which include neritic waters off the central California coast to encompass a prominent oceanographic front that occurs between cool, nearshore upwelling-modified waters and warmer offshore waters of the California Current. The front is located within 60 mi of the coast, providing a mechanism for aggregating leatherback prey, primarily brown sea nettles that have been advected from neritic central California waters and moon jellies (National Marine Fisheries Service, 2012a).

Recovery Goals

See the U.S. Pacific (NMFS and USFWS 1998a) and U.S. Caribbean, Gulf of Mexico and Atlantic Recovery Plans (NMFS and USFWS 1992) for leatherback sea turtles for complete down listing/delisting criteria for each of their respective recovery goals. The top five recovery actions identified in the Leatherback Five Year Action Plan were 1) Reduce fisheries interactions; 2) Improve nesting beach protection and increase reproductive output; 3) International cooperation; 4) Monitoring and research and 5) Public engagement.

C.7.2.5.2 Habitat and Distribution

The leatherback sea turtle is distributed worldwide in tropical and temperate waters of the Atlantic, Pacific, and Indian Oceans. Pacific leatherbacks are split into western and eastern Pacific subpopulations based on their distribution and biological and genetic characteristics. Eastern Pacific leatherbacks nest along the Pacific coast of the Americas, primarily in Mexico and Costa Rica, and forage throughout coastal and pelagic habitats of the eastern tropical Pacific, between the months of October and February (Burns et al., 2016; Eckert et al., 2015; Kuschke et al., 2023; Stewart et al., 2016b). Western Pacific leatherbacks nest in the Indo-Pacific, primarily in Indonesia, Papua New Guinea and the Solomon Islands. A proportion of this population migrates north through the waters of Indonesia, Malaysia, Philippines, and Japan, and across the Pacific past Hawaii to feeding areas off the Pacific coast of North America. Another segment of the western subpopulation migrates into the southern hemisphere through the Coral Sea, into waters of the western South Pacific Ocean (National Marine Fisheries Service, 2016u). The Western Pacific leatherback group is the primary stock that occurs within the Study Area.

Leatherback sea turtles are regularly sighted by fishermen in offshore waters surrounding the Hawaiian Islands, generally beyond the 3,800 ft. depth contour, and especially at the southeastern end of the island chain and off the northern coast of Oahu. Leatherbacks encountered in these waters, including those caught accidentally in fishing operations, may be migrating through waters surrounding Hawaii (National Marine Fisheries Service & U. S. Fish and Wildlife Service, 1998). Sightings and reported interactions with the Hawaii longline fishery commonly occur around seamount habitats above the Northwestern Hawaiian Islands (from 35°N to 45°N and 175°W to 180°W) (Skillman & Balazs, 1992; Skillman & Kleiber, 1998).

The leatherback sea turtle occurs in offshore areas surrounding the Hawaiian Islands beyond the 100 m isobath. Leatherbacks rarely occur inshore of this isobath. Incidental captures of leatherbacks have also occurred at several offshore locations around the main Hawaiian Islands (McCracken, 2000). Although leatherback bycatches are common off the island chain, leatherback-stranding events on Hawaiian beaches are uncommon. Since 1982, only five leatherbacks strandings have been reported in the Hawaiian Islands. Aerial and shipboard surveys in nearshore Hawaiian waters also suggest that nearshore occurrences are extremely rare (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2013). Leatherbacks were not sighted during any of the NMFS shipboard surveys; their deep diving capabilities and long submergence times reduce the probability that observers could spot them during marine surveys. One leatherback sea turtle was observed along the Hawaiian shoreline during monitoring surveys in 2006 (Martin et al., 2020; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2013, 2020b).

Leatherback sea turtles are regularly seen off the western coast of the U.S., with the greatest densities found in waters off of central California. Off central California, sea surface temperatures are highest during the summer and fall. These warmer temperatures and other oceanographic conditions create favorable habitat for leatherback sea turtle prey (jellyfish). There is some evidence that they follow the 61°F (16°C) isotherm into Monterey Bay, and the length of their stay apparently depends on prey availability. Satellite telemetry studies link leatherback sea turtles off the U.S. West Coast to one of the two largest remaining Pacific Ocean breeding populations in Jamursba Medi, Indonesia. Thus, nearshore waters off central California represent an important foraging region for the critically endangered Pacific Ocean leatherback sea turtle. There were 96 sightings of leatherbacks within 50 km of Monterey Bay from 1986 to 1991, mostly by recreational boaters (Benson, 2022; Benson et al., 2020; Eguchi et al., 2017; Starbird et al., 1993).

Leatherbacks lack the crushing chewing plates characteristic of hard-shelled sea turtles that feed on hard-bodied prey. Instead, they have pointed tooth-like cusps and sharp-edged jaws that are adapted for a diet of soft-bodied open-ocean prey such as jellyfish and salps. Leatherback sea turtles feed throughout the water column (Davenport, 1988; Eckert et al., 1989; Eisenberg & Frazier, 1983; Grant & Ferrell, 1993; James et al., 2005; Salmon et al., 2004). Leatherback prey is predominantly jellyfish (Aki et al., 1994; Bjorndal, 1997; James & Herman, 2001; Salmon et al., 2004). Engelhaupt et al. (2016)

conducted gastrointestinal analysis on two leatherbacks southeast of Hawaii and found 94 percent of stomach contents to be comprised of salps, the remaining portion were unidentifiable invertebrates.

C.7.2.5.3 Population Trends

Most stocks in the Pacific Ocean are faring poorly, where nesting populations have declined more than 80 percent since the 1980s, and because the threats to these subpopulations have not ceased, the International Union for Conservation of Nature has predicted a decline of 96 percent for the western Pacific subpopulation and a decline of nearly 100 percent for the eastern Pacific subpopulation by 2040 (Clark et al., 2010; National Marine Fisheries Service, 2016u; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020b; Sarti-Martinez et al., 1996; Stewart et al., 2016b). In contrast, western Atlantic and South African populations are generally stable or increasing. Causes for this decline include the intensive egg harvest in Pacific leatherback rookeries and high levels of mortality through the 1980s associated with bycatch in Pacific gill net fisheries (Fisheries and Oceans Canada, 2016a; Florida Fish and Wildlife Conservation Commission, 2015).

C.7.2.5.4 Population Threats

In addition to the general threats to sea turtles described previously, bycatch in commercial fisheries is a particular threat to leatherback sea turtles. Incidental capture in longline and coastal gillnet fisheries has caused a substantial number of leatherback sea turtle deaths, likely because leatherback sea turtles dive to depths targeted by longline fishermen and are less maneuverable than other sea turtle species. Natural factors, including the 2004 tsunami in the Indian Ocean and the tsunami that affected Japan in 2011, may have impacted leatherback nesting beach habitat through encroachment, erosion, or increased inundation with debris in leatherback foraging habitats and migratory routes (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2013). Eckert and Sarti-Martinez (1997) attributed the decline in the Mexican population of leatherbacks to the growth of the longline and coastal gillnet fisheries in the Pacific. Leatherbacks from this population migrate to the north Pacific and southeastern Pacific where these fisheries operate. Robinson et al. (2013) suggest that climate change impacts are contributing to the Pacific leatherback population declines through a shifting of nesting dates, which increases stressor exposure. Lastly, climate change may impact leatherback distribution because leatherback distributions are closely associated with jellyfish aggregations (which are affected by changing ocean temperatures and dynamics) (Pike, 2014).

C.7.3 Species Not Listed under the Endangered Species Act

The only marine reptile species in the Study Area not listed under the ESA is the yellow-bellied sea snake. This species is described in more detail in the following subsections.

C.7.3.1 Yellow-bellied Sea Snake (*Pelamis platura*)

C.7.3.1.1 Status and Management

This species is not managed under any international or U.S. regulatory framework.

C.7.3.1.2 Habitat and Distribution

The species is the most pelagic of all sea snakes, occurring in the open ocean well away from coasts and reefs. However, a small number of sea snakes wash ashore, are observed in coastal waters, or occur in inter-tidal habitats (Murphy, 2012). In the open ocean, yellow-bellied sea snakes often occur in large numbers associated with long lines of debris. These aggregations are associated with sea caves, nesting sites, or near drift lines in the open ocean. In some areas, such as the Gulf of Panama in the eastern

Pacific Ocean, the aggregations can vary in width from 1 to 300 m and include up to 1,000 individuals (Brischoux et al., 2016; Cook et al., 2015).

The yellow-bellied sea snake is the most widely distributed species of marine sea snake, ranging from the Cape of Good Hope westward across the Indo-Pacific to the western coastline of Central America (Brischoux et al., 2016; Cook et al., 2015; Lillywhite et al., 2015). Because this sea snake species exhibits a passive drifting ecology, the yellow-bellied sea snake may be carried into regions where it does not maintain a resident breeding population (e.g., California, Hawaii, New Zealand, Tasmania, the Sea of Japan, and the Galapagos) (Lillywhite et al., 2015; Udyawer et al., 2013).

The strong El Niño conditions that developed throughout the Pacific in 2015 and 2016 likely caused changes in sea levels and living marine resources distributions (Milstein, 2015). Coupled with oceanic temperature warming trends, these factors are thought to facilitate sea snake occurrence in coastal waters of California.

Yellow-bellied sea snakes are believed to prey exclusively on fish, primarily in pelagic environments (Cook et al., 2015; Lillywhite et al., 2014). Yellow-bellied sea snakes likely make shallow dives (with average depths of approximately 11 m). Cook et al. (2015) implanted temperature-depth loggers on three other sea snake species in New Caledonia. Logging 1,850 dives, nearly all dives were less than 30 m deep. A maximum dive duration was approximately 124 minutes.

C.7.3.1.3 Population Trends

Lillywhite et al. (2015) suspected that the pan-oceanic population of yellow-bellied sea snakes is exceptionally large compared to other snakes because of this species' wide range and given that aggregations number in the thousands at various locations. Estimating population size for this species is difficult, as the range is very broad over several oceans. This species, however, is fairly common throughout its known range. In addition, the distribution pattern of the yellow-bellied sea snake is very clumped. Visual surveys from boats are probably the most suitable technique for estimating population size when they occur in large aggregations associated with marine debris or from opportunistic sightings on boats or when they wash ashore (Brischoux et al., 2016; Lillywhite et al., 2014).

C.7.3.1.4 Population Threats

Squid trawlers may be a source of bycatch, but is this is thought to be a minor threat because of this species' preference for open pelagic habitats (Brischoux et al., 2016). Marine debris may also be a minor threat to this species. Udyawer et al. (2013) reported the entrapment of a sea snake (*Hydrophis elegans*) with a ceramic washer encircling its body. The authors of this study report that a post-mortem examination determined that the snake was malnourished because of the constriction.

C.8 Birds

C.8.1 General Background

Seabirds – birds that forage primarily on the open ocean - are of particular interest as the group of birds with the broadest distribution and exposure to Navy activities in the Study Area. Seabirds are a diverse group that are adapted to living in aquatic environments (Enticott & Tipling, 1997; North American Bird Conservation Initiative, 2022) and, in the Study Area use coastal (nearshore) waters, offshore waters (continental shelf), or open ocean areas (Harrison, 1983). There are many biological, physical, and behavioral adaptations that are different for seabirds than for terrestrial birds. Seabirds typically live longer, breed later in life, and produce fewer young than other bird species (Onley & Scofield, 2007). The feeding habits of seabirds are related to their individual physical characteristics, such as body mass, bill shape, and wing area (Hertel & Ballance, 1999). Some seabirds look for food (forage) on the sea surface, whereas others dive to variable depths to obtain prey (Burger, 2001). Many seabirds spend most of their lives at sea and come to land only to breed, nest, and occasionally rest (Schreiber & Chovan, 1986). Most species nest in groups (colonies) on the ground of coastal areas or oceanic islands, where breeding colonies number from a few individuals to thousands. However, many species are distributed nesters, and some seabirds are cavity nesters. Typical bird behavior to be encountered within the Study Area would include breeding, foraging, roosting, and migration. Beaches and wetlands within or bordering the Study Area may also be used as molting grounds by some species.

Additional information on the biology, life history, and conservation of bird species, including species-specific descriptions, is available from the websites of these sources:

- USFWS Migratory Bird Program and Endangered Species Program
- Birdlife International
- International Union for Conservation of Nature and Natural Resources Red List of Threatened Species
- National Audubon Society
- The Waterbird Society
- Department of Defense’s Partners in Flight

The following sections contain additional information on group size, habitat use, dive behavior, hearing and vocalization, and general threats.

C.8.1.1 Group Size

Avian radar studies at sea shows some bird groups moving across open oceans in large numbers (Desholm et al., 2006; Gauthreaux & Belser, 2003). During the winter months, large groups of ducks (rafts) could be encountered. During the nesting and breeding season, pelagic seabirds could be encountered in large groups following the currents and upwellings in pursuit of prey (Sibley, 2014). In the nearshore environments, terns, gulls, shorebirds, and plovers may occur in large groups while in their breeding and feeding areas.

Most seabird species nest in groups (colonies) on the ground of coastal areas or oceanic islands, where breeding colonies number from a few individuals to thousands. This breeding strategy is believed to have evolved in response to the limited availability of relatively predator-free nesting habitats and distance to foraging sites from breeding grounds (Siegel-Causey & Kharitonov, 1990). Outside of the breeding season, most seabirds within the Order Procellariiformes, such as the black-vented shearwater (*Puffinus opisthomelas*), are solitary, though they may join mixed-species flocks while foraging and can be associated with whales and dolphins (Onley & Scofield, 2007) or areas where prey density is high (U.S. Fish and Wildlife Service, 2005b). During the breeding season, these seabirds usually form large nesting colonies. Pelecaniform (birds within the Order Pelecaniformes) breeding, whether on the ground or in trees, is typically colonial and the most common species in the Study Area. Foraging seabirds of the order Charadriiformes (e.g., Heermann’s gull [*Larus heermanni*]) can range from singles or pairs (International Union for Conservation of Nature and Natural Resources, 2010a; U.S. Fish and Wildlife Service, 2005a) and can extend upward into larger groups where juveniles accompany adults to post-breeding foraging areas, where the water is calm and the food supply is good. There are post-season dispersal sites, where adults and fledglings congregate (U.S. Fish and Wildlife Service, 2006). Large groups are occasionally observed foraging at great distances from colonies, including at inland water sources (Atwood & Minsky, 1983).

C.8.1.2 Habitat Use

The entire world populations of Hawaiian petrels (*Pterodroma sandwichensis*) and Newell's shearwaters (*Puffinus auricularis newelli*) and more than 95 percent of the world's Laysan and black-footed albatrosses (*Phoebastria immutabilis* and *Phoebastria nigripes*, respectively) nest in the northwest Hawaiian Islands. Most of the world's ashy storm-petrels (*Oceanodroma homochroa*), western gulls (*Larus occidentalis*), and Brandt's cormorants (*Phalacrocorax penicillatus*) nest along the west coast of the U.S. (U.S. Fish and Wildlife Service, 2005a). In addition to breeding seabirds, millions of seabirds from more than 100 different species migrate to or through the Study Area. For example, an estimated 5.5 to 6 million seabirds representing more than 100 species are thought to occur off California based on at-sea surveys (U.S. Fish and Wildlife Service, 2005a). Surveys around the Hawaiian Islands found 40 different species of seabirds, half of which were local breeders and the remainder were migrant species (U.S. Fish and Wildlife Service, 2005a).

The Southern California Bight, within the California Current Large Marine Ecosystem, is important for both breeding and migratory bird species. More than 195 species of birds use coastal or offshore aquatic habitats in the Southern California Bight—the area of the Pacific Ocean lying between Point Conception on the Santa Barbara County coast to a point south of the U.S.-Mexico border (Anderson et al., 2007; Baird, 1993; Bearzi et al., 2009a; Hunt & Butler, 1980).

More than 300 bird species have been documented in and around San Diego Bay. The majority of these bay birds, representing 30 families, are migratory and may only stop to rest and feed, while others spend the winter or breed. Surveys in 2016 and 2017 identified 161 bird species and a total of 564,752 individual observations (Tierra Data Inc., 2018). Several are terrestrial birds of special concern or influence that are found about the Bay but may not directly depend upon it. Resident birds live and breed in the area year-round. Migrants that would not usually be in the area, disoriented in their travel, on the edges of their range, or simply looking for suitable habitat are regarded as vagrants. Although vagrants are not considered ordinarily dependent on the bay, a considerable number of them pass through and visit each year (U.S. Department of the Navy, 2013d).

C.8.1.3 Dive Behavior

Many of the seabird species found in the Study Area will dive, skim, or grasp prey at the water's surface or within the upper portion (1 to 2 m) of the water column (Cook et al., 2011; Jiménez et al., 2012; Sibley, 2014). However, numerous seabirds, including various species of diving ducks, cormorants, and alcids (the family that includes murres, murrelets, auks, auklets, shearwaters, and puffins), including the threatened Newell's shearwater are known to feed at depths greater than 100 ft. (Ehrlich et al., 1988). Some seabirds are aerial plunge divers in which they dive from above the surface and make generally shallow dives into the water column after prey (e.g., terns, gannets). Others are considered surface divers where they plunge directly from the surface underwater after prey (e.g., puffins, loons). Most diving species tend to catch the majority of their prey near the surface of the water column or on the bottom in shallow water (e.g., clams, mussels, and other invertebrates) (Cook et al., 2011), although some pursue prey to considerable depths as noted previously. Dive durations are correlated with depth and range from a few seconds in shallow divers to several minutes in alcids (Ponganis, 2015). Petrels forage both night and day; they capture prey by resting on the water surface and dipping their bill and by aerial pursuit of flying fish (International Union for Conservation of Nature and Natural Resources, 2010b). Hawaiian petrels eat mostly squid (50 to 75 percent of their diet), fish, and crustaceans (International Union for Conservation of Nature and Natural Resources, 2010b).

C.8.1.4 Hearing and Vocalization

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

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

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

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

C.8.1.5 General Threats

Beach-nesting birds are vulnerable to disturbance from people, pets, and off road vehicles that may inadvertently destroy or disturb nests (North American Bird Conservation Initiative U.S. Committee, 2009). Feral species (primarily cats [*Felis catus*] and rats [*Rattus* spp.], occasionally pigs [*Sus scrofa*], and cattle [*Bos taurus*]) may destroy nesting colonies. Seabirds are especially vulnerable to feral species on islands where nests and populations have been devastated through predation or habitat destruction. Invasive plants can also eliminate nesting habitat on beaches (Clavero et al., 2009; North American Bird Conservation Initiative U.S. Committee, 2009).

Lighting on boats and on offshore oil and gas platforms 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). Recent studies have looked at different lighting systems and how they may impact migrating songbirds (Poot et al., 2008). Land-based lighting has been linked to episodes of "fallout" (grounding) involving seabirds, especially petrels, and ship-based lighting could have similar effects (Rodríguez et al., 2017).

Large-scale wind energy development offshore Southern California and Hawaii is currently under consideration (Bureau of Ocean Energy Management, 2018) and has the potential to affect bird populations through (1) displacement from favored foraging habitats, especially to species that forage in deeper, offshore waters; and (2) mortality to species that tend to fly within the rotor-swept zones of large wind turbines (approximately 20 m and 200 m from the surface) (Biodiversity Research Institute, 2015).

Natural causes of seabird and shorebird population declines include disease, storms, and harmful algal blooms, although human activities are also associated with harmful algal blooms (Jessup et al., 2009; North American Bird Conservation Initiative U.S. Committee, 2009; Onley & Scofield, 2007). In addition, seabird distribution, abundance, breeding, and other behaviors are affected by cyclical environmental events, such as the El Niño Southern Oscillation and Pacific Decadal Oscillation in the Pacific Ocean (Congdon et al., 2007; Vandenbosch, 2000).

C.8.1.5.1 Water Quality

Spills of oil and other petroleum products pose a risk to seabirds and shorebirds through direct contamination and destruction of nesting, roosting, and foraging habitats. Estimates of bird mortality from the BP *Deepwater Horizon* oil spill in the Gulf of Mexico during 2010 are that approximately 200,000 birds were killed in the offshore area and approximately 700,000 killed along the coastline during the 103-day duration of the spill (Haney et al., 2014a, 2014b). Additional mortality occurred subsequently but has not been estimated.

Within the Study Area, the main risk of oil or other petroleum product spills is from ships, whether carrying petroleum to and from ports, or in fuel tanks, and from pipelines and onshore facilities that transport and store oil and gas. One of the largest spills in the Study Area occurred in 1989 when the tanker *Exxon Houston* broke away from its moorings and ran aground at Barber's Point on Oahu, spilling approximately 117,000 gallons of fuel and crude oil (U.S. Coast Guard, 2015).

C.8.1.5.2 Commercial Industries

Commercial fisheries are considered the most serious threat to the world's seabirds, while invasive species are the most pervasive – affecting the largest number of species; other threats include pollution, hunting, trapping, energy production, and mining (BirdLife International, 2012). A recent review of reported bycatch estimates suggests that at least 400,000 birds die in gillnets each year (Zydelis et al., 2013). Seabird bycatch in longline fisheries worldwide has been estimated as 160,000 to more than 320,000 annually (Anderson et al., 2011).

Large-scale offshore wind development may occur in highly productive areas offshore Southern California and Oahu, and impact bird populations by (1) displacing some species from their preferred foraging habitats and migration routes, and (2) increasing the mortality of species that fly within the rotor-swept zones of large turbines (Biodiversity Research Institute, 2015).

C.8.1.5.3 Disease and Parasites

Avian diseases can cause chronic population declines, dramatic die-offs or reductions in the reproductive success and survival of individual birds. They can even cause extinctions. Certain avian diseases appear to be spreading to populations previously unaffected, including to species already threatened by other factors. Examples include avian botulism, cholera, *Erysipelothrix rhusiopathiae*, West Nile virus, Mycoplasmal conjunctivitis, and highly pathogenic avian influenza (H5N1). A brief description of each follows from (BirdLife International, 2008).

Avian botulism is a bacterial disease that is arguably the most important disease of migratory birds worldwide, affecting millions of birds. Avian cholera and *Erysipelothrix rhusiopathiae* are two bacterial diseases that caused considerable declines of Indian yellow-nosed albatross (*Thalassarche carteri*) on Amsterdam Island (French Southern Territories). These two diseases may have spread to nearby colonies of sooty albatross (*Phoebetria fusca*) and Amsterdam albatross (*Diomedea amsterdamensis*) with a world population of approximately 130 birds. Avian cholera has also devastated the population of Cape cormorant (*Phalacrocorax capensis*) in Western Cape Province, South Africa, killing approximately 13,000 individuals between May and October 2002. The West Nile Virus, a largely mosquito-borne viral disease (causing both bird and human mortalities), has established itself over much of eastern U.S. since 1999, spreading to Latin America and the Caribbean. American crow (*Corvus brachyrhynchos*) and other corvid species have shown very high levels of mortality from this disease but remains relatively stable across its range. Mycoplasmal conjunctivitis, as the disease is commonly called, is caused by a unique strain of *Mycoplasmal gallisepticum*, a parasitic bacterium previously known to infect only poultry. This infectious disease has recently caused a significant decline in the introduced population of house finch (*Carpodacus mexicanus*) in eastern North America, and has started to spread to the native population of this species in western North America (BirdLife International, 2008).

Before humans arrived on the Hawaiian Islands, there were no mosquitoes or any other biting or blood sucking insects. By the mid-1800's however, mosquitoes were firmly established throughout the archipelago, introduced inadvertently through human activities. Of these, the southern house mosquito, (*Culex quinquefasciatus*) is the greatest challenge to Hawaiian bird conservation. Introduced in 1926, it is the primary vector of avian malaria and avian pox, and is implicated in the devastation and extinction of many of the native forest bird populations. The first was avian pox virus, first documented in forest birds in 1902. The second, first detected in the 1940s, was avian malaria. It is unclear when or how these diseases first appeared on the islands, it is thought they were first introduced with the importation of non-native bird species but it may also be that they have always been present on the islands at a low level with migratory seabirds, shorebirds and waterfowl. That all changed with the relatively recent introduction of mosquitoes (Maui Forest Bird Recovery Project, 2017).

Recent research suggests that avian malaria has not only contributed to population declines and extinctions, but is affecting the elevational distributions of many native birds on the Island of Hawai'i. Native bird species experience lower rates of transmission/infection in high-altitude forests where conditions are relatively inhospitable for the mosquito vector. Bird species endemic to low- and mid-altitude forests are at greater risk from this disease (Samuel et al., 2015).

Highly Pathogenic Avian Influenza viruses (H5N1) have infected domestic poultry and migratory birds since August 2020. The viruses have been linked to illness and deaths in birds throughout North America (U.S. Fish & Wildlife Service, 2024b). As of September 2024, approximately 10,036 wild U.S. birds have been detected with H5NI (Center for Disease Control and Prevention, 2024)

C.8.1.5.4 Invasive Animals

Significant threats to seabirds occur on islands, which is where seabirds breed, including predation and habitat disturbance from invasive alien species such as rats, cats and pigs. Ground nesting seabirds are particularly vulnerable to these threats, and invasive predators on islands have been the primary cause of global seabird declines, extirpations, and local extinctions (Spatz et al., 2014). Although effective island conservation can mitigate these threats, successful eradication or control of invasive species is extremely difficult in vast and sometime remote seabird habitat.

C.8.1.5.5 Climate Change

In the long term, global climate change could be the greatest threat to seabirds (North American Bird Conservation Initiative U.S. Committee, 2009). Species of seabirds that breed on low-lying Pacific islands are particularly vulnerable. For example, great frigatebird (*Fregata minor*) breeding colonies in the Pacific are largely restricted to islands with a maximum elevation of 8 m, making them vulnerable to sea level rise. Additionally, a sea level rise of 2 m would flood 39 to 91 percent of black-footed albatross nests on Midway Atoll, Hawaii (BirdLife International & National Audubon Society, 2015). Climate change impacts include changes in air and sea temperatures, precipitation, the frequency and intensity of storms, pH level of sea water, and sea level. These changes could impact the timing of migration and overall marine productivity, which could in turn have an impact on the food resources, distribution, and reproductive success of seabirds at critical times in their life cycles (Aebischer et al., 1990; Congdon et al., 2007; Davoren et al., 2012; Melillo et al., 2014).

Open-ocean seabird species are particularly vulnerable to climate change due to their low reproductive rates, their use of islands for nesting, and their reliance on a highly variable marine system (North American Bird Conservation Initiative & U.S. Committee, 2010). Coastal birds are vulnerable to climate change due to rising sea levels, which are expected to impact foraging and nesting habitat quality and quantity by flooding or fragmenting habitats such as barrier islands, beaches, and mudflats (North American Bird Conservation Initiative U.S. Committee, 2009).

C.8.1.5.6 Marine Debris

Marine debris is any persistent solid material that is manufactured or processed and directly or indirectly, intentionally or unintentionally, disposed of or abandoned into the marine environment or the Great Lakes (National Oceanic and Atmospheric Administration, 2016b). Marine debris is a growing environmental concern. With the rapid increase in global plastics production and the resulting large volume of litter that enters the marine environment, determining the consequences of this debris on marine fauna, including seabirds and ocean health has now become a critical environmental priority, particularly for threatened and endangered species (Wilcox et al., 2016).

Plastic debris is abundant and pervasive in the world oceans and, because of its durability, is continuing to increase. The ingestion of plastics by seabirds such as albatrosses and shearwaters occurs with high frequency and is of particular concern because of impacts on body condition and the transmission of toxic chemicals, both of which affect mortality and reproduction. The rates of plastic ingestion by seabirds are closely related to the concentrations of plastics in different areas of the ocean due to waste discharges and ocean currents, and are increasing (Kain et al., 2016; Wilcox et al., 2015).

The impacts from entanglement of marine species in marine debris are clearly profound, and in many cases, entanglements appear to be increasing despite efforts over four decades to reduce the threat. Many coastal states have undertaken certain efforts to reduce entanglement rates through marine debris clean-up measures and installed fishing line recycle centers at boat landings in part due to entanglement of seabirds and other marine species. One such program is the California Lost Fishing Gear Recycling Project administered by the University of California–Davis Wildlife Health Center that began in 2005 (National Oceanic and Atmospheric Administration Marine Debris Program, 2014).

Fishing related gear, balloons and plastic bags were estimated to pose the greatest entanglement risk to marine fauna. In contrast, experts identified a broader suite of items of concern for ingestion, with plastic bags and plastic utensils ranked as the greatest threats. Entanglement and ingestion affected a similar range of taxa, although entanglement was rated as slightly worse because it is more likely to be lethal. Contamination was scored the lowest in terms of impact, affecting a smaller portion of the taxa and being rated as having solely non-lethal impacts (Wilcox et al., 2016).

There are likely other species from other regions of the U.S. that suffer injury or death from being entangled in marine debris, but are not widely recognized or reported. Most of the literature describes entanglement of marine species from Alaska, California, Puget Sound, and Florida. However, the Mid-Atlantic and Gulf of Mexico regions of the U.S. are lacking in reports of marine debris entanglement. Similarly, reports of marine debris entanglement on seabirds are limited to a few papers (National Oceanic and Atmospheric Administration Marine Debris Program, 2014). This review reported entanglement in marine debris in the U.S. of 44 species of seabirds. The majority of cases revolve around entanglement in fishing gear and abandoned, lost or otherwise discarded fishing gear and to a lesser degree other plastic debris.

Variable impacts of common debris items on the health of marine wildlife were identified, with entanglement by fishing related gear, balloons and plastic bags emerging as the greatest threat to seabirds. However, a wide variety of other items posed at least some threat to these organisms through either ingestion, contamination or both, suggesting that a comprehensive approach to preventing plastics from entering the ocean is vitally needed (Wilcox et al., 2016).

The amount, composition, and trends of marine debris on the U.S. Pacific Coast and Hawaii were assessed using data from 1998 to 2007 collected by the National Debris Monitoring Program (Ribic et al., 2012). Data from five sites on the southern California coast indicated the majority of marine debris was land-based (60 percent), followed by general source debris (31 percent) and ocean-based (8 percent) recreational and commercial sources. In contrast, data from five sites on Hawaii (Oahu) indicated most debris was from ocean-based sources (38 percent), followed by general source debris (33 percent) and land-based sources (29 percent). No items of military origin were differentiated. Land-based debris was associated with the concentration of people; general source debris consisted primarily of plastic bottles, which may have been discarded either on land or at sea, and ocean-based debris from fishing and other maritime activities. The deposition of marine debris on beaches of the Study Area was found to be correlated with weather and to have declined in more recent years (Ribic et al., 2012).

For updated and detailed information on marine debris refer to Sections C.1.3.1.3 (Marine Debris in Nearshore and Offshore Areas off the Hawaiian Islands) and C.1.3.1.4 (Marine Debris in the Nearshore and Offshore Areas of the California Study Area).

C.8.2 Endangered Species Act-Listed Species

The following sections provide detailed species descriptions, including status and management, habitat and geographic range, population trends, predator and prey interactions, and species-specific threats for birds listed under the ESA.

C.8.2.1 California Least Tern (*Sternula antillarum browni)*

The California least tern is a federal and state endangered species that has been listed since 1970. California least terns are coastal and nearshore foragers and surface-feeding fish eaters. They are opportunistic in their search for prey, eating fish that are small enough to catch including anchovies (*Engraulidae* sp.) and smelt (*Atherinops* sp.). California least terns have black-capped heads and long, black-tipped narrow pale gray wings that contrast with their white bodies, and a broad, forked tail. When full grown, this species is less than 25 cm long and has a 75 cm wingspan (U.S. Fish and Wildlife Service, 2010b).

C.8.2.1.1 Status and Management

On October 13, 1970, the California least tern was listed as endangered under the ESA (35 FR 16047). No critical habitat has been designated for the California least tern. Conservation for the California least tern is addressed in multiple memoranda of understanding and INRMP for military lands and adjacent waters in the Study Area, including Naval Base Coronado (U.S. Department of the Navy, 2013c), San Diego Bay (U.S. Department of the Navy, 2013d), and Marine Corps Base Camp Pendleton (Marine Corps Base Camp Pendleton, 2012).

C.8.2.1.2 Habitat and Geographic Range

Preferred nesting habitat for the California least tern consists of beaches, dunes, and sand bars on the ocean shore (U.S. Fish and Wildlife Service, 1985). The California least tern nests in areas generally free of vegetation above the high tide mark. Colony sites are often near estuaries, lagoons, rivers, or the seacoast (U.S. Fish and Wildlife Service, 1985). Atwood and Minsky (1983) noted that before the decline of the species, at least 82 percent of known nesting sites in California were within 1 mi. of a river mouth or estuarine habitat.

California least terns spend the breeding season (April through August) in coastal waters along the central and southern California coast, as well as along the west and southwestern coast of Mexico. Their distribution is from San Francisco to Baja California on the Pacific Coast of North America (U.S. Fish and Wildlife Service, 2010b). The California least tern historically nested on coastal beaches of Monterey, California, to Cabo San Lucas, Baja California (Atwood & Minsky, 1983). The two largest nesting colonies in the state are on Marine Corps Base Camp Pendleton and Naval Base Coronado (Frost, 2016).

Foraging habitats include nearshore ocean waters, bays, river mouths, salt marshes, marinas, river channels, lakes, and ponds (Thompson et al., 1997). California least terns feed within 2 mi. of the shoreline in ocean waters less than 60 ft. deep, with most foraging within 1 mi. of shore (Atwood &

Minsky, 1983). Atwood and Minsky (1983) also observed a tendency for foraging birds to be concentrated in coastal waters near major river mouths. Foraging habitat use varies within and between years, depending on the stage of breeding and prey availability (Atwood & Minsky, 1983; BirdLife International, 2009). Atwood and Minsky (1983) noted in their coastal colony study that, before terns disperse after breeding, they typically forage within 2 mi. of nesting sites, although large groups were occasionally observed foraging at greater distances from colonies, including inland water sources. The presence of eelgrass is important because it is habitat for several prey species of the least tern such as topsmelt (U.S. Department of the Navy, 2013d).

California least terns occur in coastal waters throughout the California portion of the HCTT Study Area during the breeding, nonbreeding, and migration seasons. The current nesting range is from San Francisco Bay and south along the California coast to San Diego County which includes the California Study Area in the California Current Large Marine Ecosystem and parts north of the Study Area (Massey & Fancher, 1989). During migration, California least terns remain near the coast, although they have been observed foraging in multispecies feeding flocks 1 to 20 mi. off the western coast of Baja California in late April and early May (U.S. Fish and Wildlife Service, 2005a). The California least tern can be found in more offshore waters during the breeding season (courtship and incubation stages) when they forage farther from the nest site over open and deep water. Adults tend to travel farther when food availability is low, foraging in open ocean waters (BirdLife International, 2009).

California least terns forage by plunge-diving to catch prey in upper surface waters, usually within the first 1–2 m of water depth (U.S. Department of the Navy, 2013d). No information exists on specific dive depths for California least terns. Prey species include anchovies, topsmelt, opaleye, and gobies (BirdLife International, 2009). Prey species composition varies throughout the year, depending on availability. Length of foraging and peak foraging behavior typically occur from the end of May through mid-July after chicks hatch.

C.8.2.1.3 Population Trends

The California least tern population in California averaged about 4,300 pairs between 2000 and 2002, making up about 10 percent of the North American population (U.S. Fish and Wildlife Service, 2005a). The California population increased almost 12-fold from a low of 600 pairs in the early 1970s to roughly 7,100 pairs in 2005 (U.S. Fish and Wildlife Service, 2001, 2005a), but declined in recent years to an estimated 4,202–5,295 pairs as of 2015 (Frost, 2016). Monitoring in the 2018 and 2019 California least tern breeding season estimated 60 nesting sites and 3,741 to 4,502 breeding pairs (Sin et al., 2024).

C.8.2.1.4 Population Threats

Threats to breeding least terns include changing food resources, habitat modification due to encroaching vegetation, the alteration of river habitat, flooding and development of coastal areas, disruptive recreation, an increase in aggressive gulls that compete for nesting sites, predation by native and feral species, such as rats, great horned owls (*Bubo virginianus*), American peregrine falcons (*Falco peregrinus anatum*), burrowing owls (*Athene cunicularia*), gull-billed terns (*Gelochelidon nilotica*), black-crowned night herons (*Nycticorax nycticorax*), dogs, and cats, and disease (e.g., West Nile Virus and avian influenza) (Sidle et al., 1992; U.S. Fish and Wildlife Service, 1990, 2020b).

Anthropogenic related threats to California least terns include environmental contamination (e.g., DDT, selenium oil, and mercury), the degradation of habitat and rising sea levels associated with climate change, recreational beach use, disturbance during nesting season, intentional release of feral cats, and ultralight aircraft and drone use (U.S. Fish and Wildlife Service, 2010a, 2020b).

C.8.2.2 Hawaiian Petrel (*Pterodroma Sandwichensis*)

The Hawaiian petrel was recently split from the Galapagos petrel (*Pterodroma phaeopygia*) based on genetic and morphological evidence; before the split they were collectively known as the dark-rumped petrel (U.S. Fish and Wildlife Service, 2005a). The Hawaiian Petrel has a dark gray head, wings, and tail, and a white forehead and belly. It has a stout grayish-black bill that is hooked at the tip, and pink and black feet. This bird measures 16 in. in length and has a wing span of three ft. It has a distinctive call during breeding season that sounds like "oo ah oo." They also have calls that sound like the yapping of a small dog (U.S. Fish and Wildlife Service, 2012b).

C.8.2.2.1 Status and Management

The Hawaiian petrel is found only in Hawaii and is listed as endangered throughout its range under the ESA (U.S. Fish and Wildlife Service, 2005a); there is no designated critical habitat. It is known from five locations in the main Hawaiian Islands, at least two of which (Mauna Loa on Hawaii Island and West Maui) are threatened by development. Its limited distribution and declines primarily result from predation by introduced mammals and urbanization (International Union for Conservation of Nature, 2017). In some cases, predation has caused more than 70 percent nesting failure (U.S. Fish and Wildlife Service, 2005a). Numerous knowledge gaps remain for the Hawaiian Petrel such as foraging and other at-sea behavior; annual and age-specific survival, especially for non-breeders; and the scope and severity of threats at sea (U.S. Fish and Wildlife Service, 2011a). On Maui, there is a long-term monitoring program in the Haleakala National Park and efforts are made to control introduced mammals. In 1976, a perimeter fence was put up around the main colony to exclude feral goats and pigs from the habitat. The predator enclosures placed around the national park may have facilitated an increase in the number of birds in eastern Maui (International Union for Conservation of Nature, 2017). On Kauai, auditory surveys to detect colonies are ongoing, and have included documenting the first known nesting sites on Kauai in the Upper Limahuli Preserve. Also on Kauai, street lighting is shielded in critical areas and lighting on some buildings has been modified to reduce collisions. A ruling by the USFWS under the ESA has resulted in a campaign running since 2005, in which nonessential lights on Kauai are turned off or shielded between 15 September and 15 December when young birds leave their nests. The island's electricity company is helping by darkening all of its 3,000 street lights, and shielding or turning some of them off. The company has also fitted large balls to power lines in an effort to reduce the number of birds that collide with the cables. Significant improvements have been made in reducing light attraction and collision, although there is still a considerable amount of new and existing infrastructure that requires modification (International Union for Conservation of Nature, 2017).

C.8.2.2.2 Habitat and Geographic Range

Hawaiian Petrel ranges in the central Pacific and breeds only in the main Hawaiian Islands, though there are specimen records from Japan, Philippines, and Moluccas at the western edge of the distribution. On Maui, Mauna Kea and Mauna Loa, nesting takes place mainly between 2,000 and 3,000 m, in lava cavities with little vegetation nearby. Elsewhere, for example West Maui and Kauai, it nests at lower elevations, amongst dense shrubs and ferns, or in native grasslands with bracken. On Haleakala, Maui, birds nest in rock crevices and tunnels that are over 0.5 m deep, often exceeding 2 m. Generally, the nest chamber can be from 1 to 9 m deep. Pairs nest in cavities in the volcanic terrain, in burrows beneath rocks or at the base of clay cliffs. At lower elevations, they excavate burrows or nest in cavities often at the base of trees, although many burrows on Lanai are not at the base of trees. On Lanai, birds breed in dense uluhe fern habitat (*Dicranopteris linearis* and *Diplopterygium pinnatum*) and start breeding in March, whereas on Haleakala, Maui, birds begin breeding in early February (International Union for Conservation of Nature, 2017).

The Hawaiian petrel typically feeds well offshore but tends to feed closer to shore (0 to 45 mi.) during spring than in the fall (most abundant at 170 to 230 mi.) (Spear et al., 1995). The Hawaiian petrel favors open ocean water conditions, with an average sea surface temperature of 80°F, sea surface salinity of 34 parts per thousand, wind speed of 19 mi. per hour (mph), and a wave height of 5 ft. It also prefers an average depth from the warmer surface water to the point where cold water begins (the thermocline) of 35 ft. (Spear et al., 1995).

The Hawaiian petrel is an open ocean species of the central tropical Pacific (U.S. Fish and Wildlife Service, 2005a). They occur in open ocean waters throughout most of the Hawaii portion of the Study Area and the western portion of the Transit Corridor in the Insular Pacific-Hawaiian Large Marine Ecosystem. The Hawaiian petrel occurs largely in equatorial waters of the eastern tropical Pacific, generally from 10 °S to 20 °N. Because of the difficulty in identification, the precise southeastern extent of the Hawaiian petrel and the northwestern extent of the similar Galapagos petrel remains uncertain (Spear et al., 1995).

Hawaiian petrels have important resting sites in coastal waters throughout the Hawaii Study Area in portions of the Insular Pacific-Hawaiian Large Marine Ecosystem. An area of the north shore of Kauai is widely known as a resting location for Hawaiian petrels (Birding Hawaii, 2004). Based on known or suspected colony sites, gathering areas likely occur near shore on Lehua Rock, Kauai, Molokai, Lanai, Maui, and Hawaii (Day & Cooper, 1995; Day et al., 2003; U.S. Fish and Wildlife Service, 2005a) and perhaps around Kahoolawe (U.S. Fish and Wildlife Service, 2005a). These areas provide resting habitat before the birds fly to inland nesting colonies. Hawaiian petrels move to and from nesting colonies during dusk and dawn (International Union for Conservation of Nature, 2017).

It has been found that at least some Hawaiian petrels nesting on Lanai feed in waters around the Aleutian Islands, as shown through the use of tracking devices on several breeding individuals. The species usually forages in mixed species flocks, typically over schools of predatory fish species. Hawaiian petrels eat mostly squid (50 to 75 percent of their diet), fish, and crustaceans. They forage both night and day; they capture prey by resting on the water surface and dipping their bill and by aerial pursuit of flying fish (International Union for Conservation of Nature, 2017). Although the range of Hawaiian petrels is unknown, birds have been recorded as far as 600 mi. north and 1,000 mi. south of Hawaii (U.S. Fish and Wildlife Service, 2022).

C.8.2.2.3 Population Trends

A population estimate supported by pelagic surveys put the total population at 19,000 (range 10,600–34,400), including a best estimate of 4,500–5,000 breeding pairs. However, the discovery of previously unknown colonies in 2006–2007 may bring the total population closer to the upper estimate of 6,500–8,300 pairs (International Union for Conservation of Nature, 2017). The current status of the Hawaiian petrel is uncertain due to the difficulty surveying this species. Recent at-sea surveys are currently being analyzed for Hawaiian petrel and Newell's shearwater (U.S. Fish and Wildlife Service, 2011a). The total population of Hawaiian petrels was estimated at 20,000, with a breeding population of 4,500–5,000 pairs (Spear et al., 1995; U.S. Fish and Wildlife Service, 2005a) overall population trends on the Hawaiian Islands are not known (U.S. Fish and Wildlife Service, 2005a, 2022). Numbers of breeding Hawaiian petrels on Maui appear stable and have increased in areas of the Haleakala National Park, where predators are being managed (U.S. Fish and Wildlife Service, 2005a). On Hawaii, numbers may be declining because of predation by introduced species (U.S. Fish and Wildlife Service, 2005a).

C.8.2.2.4 Population Threats

Threats to the Hawaiian petrel include predation by introduced mammals, development, light attraction and collision, ocean pollution, and disturbance of its breeding grounds. The petrel does not have any natural defenses against predators such as rats, feral cats, and mongooses, and its burrows are very vulnerable (U.S. Fish and Wildlife Service, 2012b). Although predator control now occurs at several Hawaiian petrel breeding sites, the threat posed by introduced predators remains significant throughout the species' range (U.S. Fish and Wildlife Service, 2011a). Recently, Young et al. (2023) successfully translocated 110 Hawaiian petrels (along with 86 Newell's shearwaters) to a predator free enclosure on Kauai, establishing the first predator-free colony for Hawaiian petrels.

Collisions with artificial lights, utility poles, and fences kill Hawaiian petrels on some islands (International Union for Conservation of Nature, 2017). Little progress has been made toward addressing the chief threats to or meeting the recovery criteria for Hawaiian petrels. Remnant breeding colonies thought to occur on west Maui, Hawaii Island, Kauai, Lanai, and possibly Molokai are not mapped or managed. These colonies are certainly subject to predation by alien mammals, possibly are subject to the threat of light attraction and collision, and most are thought to be dwindling as well (U.S. Fish and Wildlife Service, 2011a).

The species may be adversely affected by declines in the populations of large predatory fish that drive prey species to the surface. In addition, a significantly lower percentage of birds come ashore to nest during El Niño years (ca.40 percent compared to ca.65 percent normally), suggesting that the species is sensitive to such disturbances in environmental conditions (International Union for Conservation of Nature, 2017). In addition, the species is currently threatened with habitat disturbance by goats, pigs and cattle. On Lanai, habitat degradation caused by the invasive tree, strawberry guava *Psidium cattleianum*, may be the biggest threat to the long term survival of the colony. On occasion, fledglings become grounded after colliding with lights, and mortality sometimes results from collisions with fences and powerlines. Once on the ground, fledglings are unable to fly and are killed by cars or cats and dogs, or die from starvation or dehydration (International Union for Conservation of Nature, 2017).

C.8.2.3 Short-tailed Albatross (*Phoebastria albatrus*)

The short-tailed albatross was formerly in the genus *Diomedea* and known as Steller's albatross. The largest of the north Pacific albatrosses, the adult short-tailed albatross has a prominent pink bill, white body, and a yellow wash on the head. Immature birds are dark but can be distinguished from black-footed albatross by their pink bill and flesh-colored feet. Adults can reach wingspans of 7 ft. (U.S. Fish and Wildlife Service, 2012d).

C.8.2.3.1 Status and Management

On July 31, 2000, the short-tailed albatross was listed as endangered throughout its range under the ESA throughout its range (65 FR 46643). Their decline at the beginning of the 20th century was attributed, in part, to albatross being killed for their feathers and various other body parts, including their down feathers being used for quilts and pillows, and wing and tail feathers being used for writing quills; their bodies were processed into fertilizer, and their eggs were collected for food (65 FR 46643). In 2020, USFWS indicated that short-tailed albatross are making good progress toward meeting some delisting recovery criteria, however a change in status was not recommended in their review. Critical habitat has not been designated or proposed for the short-tailed albatross. In the 2000 final rule (65 FR 46643), the

USFWS determined that designation of critical habitat was not prudent due to the lack of habitat-related threats to the species, the lack of specific areas in U.S. jurisdiction that could be identified as meeting the definition of critical habitat (U.S. Fish and Wildlife Service, 2008c).

C.8.2.3.2 Habitat and Geographic Range

The range of the short-tailed albatross extends from Siberia south to the China coast, into the Bering Sea and Gulf of Alaska south to Baja California, Mexico, and throughout the North Pacific. Breeding is primarily restricted to islands in Japan and Taiwan, principally Torishima Island and the Senkaku Islands of Japan (U.S. Fish and Wildlife Service, 2008c, 2020a). Short-tailed albatrosses are typically found in the open ocean and tend to concentrate along the edge of the continental shelf (NatureServe, 2004). Upwelling zones are not only nutrient rich, but they also bring prey (for example, squid and fish) typically found only in deeper water to the surface, where they become available to albatrosses. Upwelling occurs when the wind moves warm, nutrient poor water away from the area, which allows colder, nutrient rich water to rise to the surface of the ocean. Short-tailed albatross nest on isolated, windswept, offshore islands with restricted human access (U.S. Fish and Wildlife Service, 2000). Current and historical nesting habitat can be described as flat to steep slopes that are sparsely or fully vegetated. Short-tailed albatrosses disperse throughout the temperate and subarctic North Pacific approximately from May to October when they are not breeding, from Japan through California (U.S. Fish and Wildlife Service, 2005a, 2008a). Non-breeders and failed breeders disperse from the colony months sooner. While many non-breeders return to the colonies each year, the presence of immature birds far from the colony (such as the U.S. Pacific coast) during the breeding season suggests that some immature birds may spend years at sea before they return to the colony (U.S. Fish and Wildlife Service, 2005b).

The diet includes squid (especially the Japanese common squid [*Todarodes pacificus*]), shrimp, fish (including bonitos [*Sarda* sp.], flying fishes [*Exocoetidae*] and sardines [*Clupeidae*]), flying fish eggs, and other crustaceans. Short-tailed albatross forages diurnally and possibly nocturnally, either singly or in groups (occasionally in the 100s). The species is a surface feeder and scavenger, and predominately takes prey by surface-seizing, not diving (U.S. Fish and Wildlife Service, 2008b).

Open Ocean

The short-tailed albatross is an open ocean species that occurs throughout the Hawaii Study Area, Transit Corridor, and California Study Area. The range of the short-tailed albatross extends from Siberia south to the China coast, into the Bering Sea and Gulf of Alaska south to Baja California, Mexico, and throughout the North Pacific, including the Northwestern Hawaiian Islands (Committee on the Status of Endangered Wildlife in Canada, 2003; Harrison, 1983; Roberson, 2000). Their at-sea distribution includes the entire North Pacific Ocean north of about 20 latitude °N l. Short-tailed albatrosses move seasonally around the North Pacific Ocean, with high densities observed during the breeding season (December through May) in Japan and throughout Alaska and along the west coast of North America during the post-breeding season (April through September) (International Union for Conservation of Nature, 2017). Nonbreeding subadults can be found in all areas throughout the year. They are seen regularly in the North Pacific Subtropical Gyre (U.S. Fish and Wildlife Service, 2005b).

California Current Large Marine Ecosystem

Short-tailed albatrosses occasionally occur in California Study Area portion of the California Current Large Marine Ecosystem, which is part of the Study Area. As the population began a gradual recovery after 1950, sporadic sightings have been recorded off California (International Union for Conservation of

Nature and Natural Resources, 2010c). Based on the number of sightings in the California Study Area, the short-tailed albatross is considered rare in that portion of the Study Area, as well as off the entire California coast. Breeding does not occur in the Southern California Bight, but because of the unique circulation and upwelling characteristics of this area, potential foraging habitat exists. Two documented sightings of the short-tailed albatross have occurred in southern California. Roberson (2000) reported a sighting in 1977 of an all-dark immature bird approximately 90 mi. west of the San Diego area. McCaskie and Garrett (2002) reported a sighting in the vicinity of Santa Barbara Island in late February of 2002.

Insular Pacific-Hawaiian Large Marine Ecosystem

Short-tailed albatrosses occur in coastal waters throughout the Hawaii portion of the Study Area in the Insular Pacific-Hawaiian Large Marine Ecosystem. The short-tailed albatross regularly occurs on Midway Atoll and has been observed at other Northwestern Hawaiian Islands. Since the 1930s, short-tailed albatrosses have been occasionally reported during the breeding season at Midway Atoll. Some of these short-tailed albatrosses were recorded for several successive years. The first confirmed nest site that produced an egg did not occur until 1993 (International Union for Conservation of Nature and Natural Resources, 2010c). A chick was fledged by a breeding pair on Midway Atoll in 2011, 2012, and 2014 (U.S. Fish and Wildlife Service, 2014). Nesting elsewhere on the Northwestern Hawaiian Islands has been attempted, but successful nesting has not been confirmed (U.S. Fish and Wildlife Service, 2005b). In the Hawaiian Islands, there was an unconfirmed sighting at Barking Sands on Kauai during March 2000 (Birding Hawaii, 2004). Other known occurrences in Hawaii are of single birds (in 1976 and 1981) at French Frigate Shoals in the Northwestern Hawaiian Islands (U.S. Fish and Wildlife Service, 2008b).

Short-tailed albatrosses are surface feeders and scavengers, feeding more inshore than other North Pacific albatrosses. In Japan, their diet consists of shrimp, squid, and fish (including bonita, flying fish, and sardines); diet information is not available for birds in the Study Area (U.S. Fish and Wildlife Service, 2005b). Unlike other North Pacific albatrosses, short-tailed albatrosses frequently feed in sight of land.

C.8.2.3.3 Population Trends

The total population estimate for breeding age short-tailed albatrosses as of the 2013–2014 nesting season is 1,928 individuals, including approximately 1,624 at Torishima (Japan), 293 on the Senkaku (or Diaoyutai) Islands (in disputed ownership among China, Taiwan, and Japan), 4 in the Northwestern Hawaiian Islands, and a few birds on other Japanese islands (U.S. Fish and Wildlife Service, 2014). Per the 2018–2019 breeding season, the number of breeding pairs was 1,011 and the total population of short-tailed albatross was estimated at 7,365 (U.S. Fish and Wildlife Service, 2020a).

The Torishima population is growing rapidly, averaging 8.9 percent annually (U.S. Fish and Wildlife Service, 2020a). Overall, the high population growth rates continue, and some substantial threats are being addressed in much of the species' range. However, to ensure the continuing recovery of the species, it is important to consider sources of uncertainty and work toward reducing those threats that adversely affect short-tailed albatrosses. The population does not yet meet the recovery goals for downlisting or delisting, and therefore, the short-tailed albatross remains endangered throughout its range (U.S. Fish and Wildlife Service, 2020a).

C.8.2.3.4 Population Threats

Short-tailed albatrosses have survived multiple threats to their existence. During the late 1800s and early 1900s, feather hunters clubbed to death an estimated five million of them, stopping only when the species was nearly extinct. In the 1930s, nesting habitat on the only active nesting island in Japan was damaged by volcanic eruptions, leaving fewer than 50 birds by the 1940s. Loss of nesting habitat to volcanic eruptions, severe storms, and competition with black-footed albatrosses for nesting habitat continue to be natural threats to short-tailed albatrosses today. In addition, predation of short-tailed albatross by white-tailed eagles (*Haliaeetus albicilla*) and Steller's sea eagles (*H. pelagicus*) are known to occur.

Current threats to this species include ingestion of plastics mistaken for food items, volcanic eruption (at Torishima, Japan), typhoons, sunken longline fishing in Alaska and Russia, jig/troll fishery in Japan, invasive species at colonies (cats, rats, and plants), and researcher disturbance (U.S. Fish and Wildlife Service, 2005b). The prevalence and extent of plastic impacts needs further investigation to determine its acute and long-term effects on the short-tailed albatross (U.S. Fish and Wildlife Service, 2014). Additional human-induced threats include contamination from oil spills, and potential predation by introduced mammals on breeding islands. Furthermore, incidental interactions from commercial longline gear, though McCracken (2014) asserts in an internal report, there has not been an observed incidental interaction with a short-tailed albatross during the history of the NMFS observer program with respect to the Hawaii deep-set longline fishery. Shipping conflicts and potential oil spills appear to be growing threats but the likelihood and magnitude is uncertain. Global climate change may be causing changes in the distribution of the short-tailed albatross in the North Pacific, but the overall impact of that change is also unknown (U.S. Fish & Wildlife Service, 2024a). The lack of information about the magnitude or impact of these threats on the short-tailed albatross results in uncertainty about the future recovery of the population (U.S. Fish and Wildlife Service, 2014).

C.8.2.4 Marbled Murrelet (*Brachyramphus marmoratus*)

The marbled murrelet is a small, chubby seabird that has a very short neck. During the breeding season it has dark brown to blackish upperparts and a white belly and throat that are greatly mottled. During the winter the upperparts become grey, dark marks form on the sides of the breast and a white ring develops around the eye. Males and females are similar in appearance and size (U.S. Fish & Wildlife Service, 2024a).

C.8.2.4.1 Status and Management

On October 1, 1992, the marbled murrelet (*Brachyramphus marmoratus*) was listed as a threatened species in California, Oregon, and Washington under the ESA (57 FR 45328). This species is also considered endangered by the state of California (California Department of Fish and Wildlife, 2016). Marbled murrelet populations have suffered significant declines in the Pacific Northwest, caused primarily by the removal of essential nesting habitat by logging and coastal development . The most recent 5-year review was completed in 2024 (U.S. Fish & Wildlife Service, 2024a). Based on the evaluation of murrelet populations, habitats, and threats, USFWS (2024a) determined that the murrelet should remain listed as threatened. To stem these declines, critical habitat was designated in 1996 (61 FR 26256) and revised in 2011 (81 FR 51348) to protect mature and old-growth forest nesting habitat determined to have been occupied by the species at the time of listing. The entire area of designated critical habitat occurs outside of the Study Area.

C.8.2.4.2 Habitat and Geographic Range

Murrelets spend most of their lives in the marine environment where they forage in nearshore areas and consume a diversity of prey species, including small fish and invertebrates. In their terrestrial environment, the presence of platforms (large branches or deformities) used for nesting is the most important characteristic of their nesting habitat. Murrelet habitat use during the breeding season is positively associated with the presence and abundance of mature and old-growth forests, large core areas of old-growth, low amounts of edge habitat, reduced habitat fragmentation, proximity to the marine environment, and forests that are increasing in stand age and height (U.S. Fish and Wildlife Service, 2009). Marbled murrelets do not build a nest but use natural features, such as moss, clumps of mistletoe, or piles of needles as a nest site on tree limbs (International Union for Conservation of Nature, 2017; Wilk et al., 2016). Nests are in large conifers, such as coast redwood and western hemlock, in old-growth stands typically within 35 mi. of marine waters. Important features in nesting habitat are stands of 500 acres or larger, multistoried canopy layers, and less than average canopy closures (Grenier & Nelson, 1995; Hamer & Nelson, 1995; Miller & Ralph, 1995). In addition, habitat along major drainages (e.g., rivers and streams) is a key component, as murrelets tend to use these drainages as flight corridors to and from inland nest sites (McIver et al., 2021).

Marbled murrelets generally remain near breeding sites year-round in most areas (U.S. Fish and Wildlife Service, 2005a). Birds occur closer to shore in exposed coastal areas and farther offshore in protected coastal areas (McIver et al., 2021). The highest concentrations are found in protected inshore waters (U.S. Fish and Wildlife Service, 2005a). Physical and biological oceanographic processes that concentrate prey (such as upwelling and rip currents) have an important influence on the foraging distribution of marbled murrelets (Day & Nigro, 2000; Jodice & Collopy, 1999; Strachan et al., 1995; Whitworth et al., 2000). They are more commonly found inland during the summer breeding season but make daily trips to the ocean to gather food and have been detected in forests throughout the year. When not nesting, the birds live at sea, spending their days feeding close to shore and then moving several miles offshore at night.

Marbled murrelets only occur in coastal waters of the California Current Large Marine Ecosystem within the northeast corner of the California Study Area. Eight reported sightings of marbled murrelets have been documented within the Study Area off the California coast. Sightings have been reported at Marina del Rey, off Santa Barbara Island, at Mugu Lagoon in Ventura County, along the coast in San Diego County, and at the northern end of the Study Area near San Simeon Point (McCaskie & Garrett, 2001). All of these documented sightings were recorded between November and March.

Foraging habitat in the Southern California Bight occurs usually within 3 mi. of the coast in waters less than 195 ft. deep (Day & Nigro, 2000; Felis et al., 2022; Henkel et al., 2004; Strong, 2020); however, because upwelling areas represent important foraging habitat for the marbled murrelet, the potential exists for individuals to be observed farther offshore in the Southern California Bight.

Winter distributions of marbled murrelets are poorly documented. In California, most birds appear to be year-round residents near breeding areas (Naslund, 1993), although dispersal in the winter as far south as Southern California and northern Mexico has been documented (Erickson et al., 1995). A single sighting has occurred at Ensenada Harbor (Erickson et al., 1995). The species is a rare fall/winter vagrant (occurring outside of its normal range) to Southern California, and is “accidental” from the U.S.-Mexico border south along the Mexico coastline (Strong, 2020).

Marbled murrelets feed opportunistically on small fish, including sand lance, anchovy, herring, capelin, and smelt, and also on invertebrates (U.S. Fish and Wildlife Service, 1996, 2005b). Feeding takes place in the nearshore marine environment, primarily in protected waters where both Pacific sand lance and surf smelt occur (Burger, 2002; Whitworth et al., 2005). Individuals forage by diving, using their wings for underwater propulsion. The murrelet forages by pursuit diving in relatively shallow waters, usually between 20 and 80 m in depth. Most birds are found as pairs or as singles in a band about 300–2,000 m from shore. Foraging dive times averaged about 16 seconds. Murrelets generally forage during the day and are most active in the morning and late afternoon hours. Some foraging occurs at night (Ralph & Miller, 1995).

C.8.2.4.3 Population Trends

The total estimated marbled murrelet population is 358,200–417,500 individuals, rounded here to 350,000–420,000 individuals, based on 271,000 individuals in Alaska, 72,600–125,600 in British Columbia, and 14,631–20,952 individuals in Washington, Oregon and California (U.S. Fish & Wildlife Service, 2024a). The population was estimated to have declined by approximately 15 percent in 2000 to 2007 in Washington, Oregon, and California, with a 50 percent decrease in Alaska in 1972 to 1992 and a 29 percent decrease in 2001 to 2010, and a 40 percent decrease in some parts of British Columbia in 1982–1992. At-sea surveys over the past 25 years in British Columbia suggest declines of approximately 1 percent per year although radar surveys suggest the population may have been relatively stable since 1999. Availability of nesting habitat in British Columbia, which is strongly correlated with local breeding populations, has declined by 22 percent between 1978 and 2008 and is continuing. Declines are suspected to be very rapid and ongoing due to very low measured productivity rates (U.S. Fish & Wildlife Service, 2024a). Marbled murrelet 2020 population monitoring in southern Oregon and northern California estimated 6,821 individuals in the region. This estimate is consistent with past population estimates (Strong, 2020).

Within Conservation Zones 1 and 2, recent trends in abundance showed declines of -4.6 percent and -3.5 percent, respectively (U.S. Fish & Wildlife Service, 2024a). Within Zones 3 and 4, recent trends in abundance indicated slight increases of 1.6 percent and 2.8 percent, respectively. Within Zones 5 and 6, trends were slightly positive, but statistically flat at 1.5 percent and 0.01 percent, respectively. Productivity was low in all zones. Population resiliency ranges from moderate to very low. It is highest in the central portion of the range, in Conservation Zones 3 and 4, and lower in in the northern and southern portions of the range, in Conservation Zones 1, 2, 5, and 6. USFWS (2024a) described the 2018–2023 Zone-specific abundance range for marbled murrelets as Zone 1: 3,143–3,843 individuals, Zone 2: 1,018–1,657 individuals, Zone 3: 8,249–8,414 individuals, Zone 4: 5,132–6,822 individuals, Zone 5: 42 individuals, and Zone 6: 5,132–6,822 individuals.

C.8.2.4.4 Population Threats

The decline of marbled murrelets has been largely caused by extensive removal of late-successional and old growth coastal forest which serve as nesting habitat for murrelets. Additional factors in its decline include high nest-site predation rates and human-induced mortality in the marine environment from disturbance, gillnets, and oil spills. In addition, murrelet reproductive success is strongly correlated with the abundance of mid-trophic level prey. Effects to the marine environment that impact the availability of prey can occur through overfishing or oceanographic variation from weather or climate events. Affects to adults in the marine environment from disturbance events like underwater detonations or pile driving can also impact their ability to forage and successfully provide for their young (U.S. Fish and Wildlife Service, 2009). An estimated 3,500 murrelets are killed annually in Alaska by gill-net fisheries (Carter et al., 2005; Piatt & Naslund, 1995). In addition, more than 1,000 oiled marbled murrelet carcasses were collected after the Exxon Valdez oil spill in Alaska (Carter & Kuletz, 1995). Nest failure is caused by predation by raptors, ravens, and jays (Nelson, 1997; Pastran & Lank, 2024).

C.8.2.5 Newell's Shearwater (*Puffinus auricularis newelli*)

The Newell's shearwater or ʻaʻo as native peoples refer to it, is a medium-sized shearwater measuring 12 to 14 in. with a wing span of 30–35 in. It has a glossy black top, a white bottom, and a black bill that is sharply hooked at the tip. Its claws are well adapted for burrow excavation and climbing (U.S. Fish and Wildlife Service, 2012c).

As of May 2016, Newell's shearwater is considered a subspecies of Townsend's shearwater (*P. auricularis*) by the USFWS (*FR* 81 (91), 29165–29166, May 11, 2016). It is regarded by some authorities as a distinct species, *P. newelli* (American Ornithological Society, 2018; International Union for Conservation of Nature, 2017), and at least one author (Harrison, 1983) considered Newell's shearwater a subspecies of Manx shearwater (*P. puffinus newelli*), but since 1982, most authorities have considered it a subspecies of Townsend's shearwater (American Ornithologists' Union, 1998). Newell's shearwater is also known as Newell's dark-rumped shearwater. The Newell's shearwater is a bird of the open tropical seas and offshore waters near breeding grounds. During their nine-month breeding season from April through November, they nest in burrows under ferns on forested mountain slopes. These burrows are used year after year and usually by the same pair of birds. Although Newell's shearwater is capable of climbing shrubs and trees before taking flight, it needs an open downhill flight path through which it can become airborne (U.S. Fish and Wildlife Service, 2012c).

C.8.2.5.1 Status and Management

The Newell's shearwater was listed as a threatened species by the USFWS in 1975 (U.S. Fish and Wildlife Service, 2012c), and under evaluation to be upgraded to endangered (U.S. Fish and Wildlife Service, 2011b). This species is also listed as threatened by the state of Hawaii (U.S. Fish and Wildlife Service, 2005a). Newell's shearwater appears to have declined very rapidly on its main breeding island, possibly associated with the impacts of Hurricane Iniki in 1992, and continues to decline, with two known colonies in the early 1980s, and possibly a third, now abandoned. Combined with longer term declines owing to a number of other threats, it qualifies as Endangered (U.S. Fish and Wildlife Service, 2017a). A federal recovery plan was finalized in 1983 (U.S. Fish and Wildlife Service, 1983). The Newell's shearwater was once abundant on all main Hawaiian Islands. Today, the majority of these birds nest primarily in mountainous terrain between 500 and 2,300 ft. on Kauai. This seabird was reported to be in danger of extinction by the 1930s. The introduction of the mongoose, cat, black rat, and Norway rat may have played a primary role in the reduction of ground nesting seabirds (U.S. Fish and Wildlife Service, 2012c).

Newell's shearwater faces a high level of threat, the population is declining sharply, the threats are difficult and costly to mitigate, the threats are largely unmitigated, and the listed entity is currently recognized as a subspecies. This species' sharp decline and the level of threat warrant reclassification of Newell's shearwater from threatened to endangered as it is no longer "likely to become an endangered species" but is now "in danger of extinction throughout all of its range." (U.S. Fish and Wildlife Service, 2011b). Within the Hawaiian Islands Bird Conservation Region, Newell's shearwater is evaluated as highly imperiled, the most serious category, because of restricted breeding distribution and threats to breeding populations (U.S. Fish and Wildlife Service, 2003). There is no critical habitat designation for the Newell's shearwater.

Newell's shearwater was thought to be extinct by 1908 as a consequence of subsistence hunting by Polynesians and predation by introduced rats, pigs, and dogs. However, they were rediscovered offshore in 1947. One was collected on Oahu in 1954 (Day et al., 2003) and Newell's shearwaters were confirmed as still breeding on Kauai in 1967 (U.S. Fish and Wildlife Service, 2005a). The two most important factors limiting population growth are low breeding probability (birds do not mature until 6 years of age and a high proportion are nonbreeding adults), and high rates of predation on adults and subadults (U.S. Fish and Wildlife Service, 2011b). Predator control in key habitat areas, the establishment of Bird Salvage-Aid Stations, translocation, and light attraction studies have been initiated to help save the Newell's shearwater. Outreach to Kauai's local community has resulted in people picking up injured birds and bringing them to aid stations for care and release, giving the seabirds a chance to live (U.S. Fish and Wildlife Service, 2012c).

C.8.2.5.2 Habitat and Geographic Range

Newell's shearwater occurs in open ocean waters in the southern portion of the Hawaii portion of the Study Area and into the western portion of the Transit Corridor Study Area. They spend most of their time in the open ocean year-round (U.S. Fish and Wildlife Service, 2005a) and come ashore only to nest. They avoid inshore waters except when gathering before they fly inland to breeding colonies at night (International Union for Conservation of Nature, 2017).

Newell's shearwaters forage only over open ocean waters of depths reportedly much greater than 6,560 ft. (Spear et al., 1995). Even when nesting, they feed over deep waters and are typically not within 15 mi. of island shores (International Union for Conservation of Nature, 2017). In particular, they find abundant food along oceanic fronts, such as the Equatorial Countercurrent (Spear et al., 1995). Preferred average ocean conditions are 80°F sea surface temperature, 34.5 parts per thousand sea surface salinity, and 250 ft. depth to cold water (Spear et al., 1995). The meteorological conditions favored by Newell's shearwaters are frequent clouds and rain squalls typical of intertropical convergence zones (Spear et al., 1995).

Although the diet of the Newell's shearwater is not well known, evidence suggests that squid are a major dietary item. It often forages hundreds of km offshore, often in large, mixed species flocks associated with schools of large, predatory fish that drive prey species to the ocean surface. Newell's shearwaters capture food by pursuit-plunging (diving into water and swimming after prey, typically 10 to 30 m deep), usually in company with multispecies feeding flocks associated with tuna (International Union for Conservation of Nature, 2017). This species is not attracted to discarded fish byproducts and does not follow ships (Onley & Scofield, 2007).

Insular Pacific-Hawaiian Large Marine Ecosystem

Newell's shearwater occurs in coastal waters throughout the Hawaii portion of the Study Area during the breeding season. Newell's shearwater nesting is entirely confined to the main Hawaiian Islands, from Lehua Rock east to Hawaii. Nesting is known on Lehua Rock, Kauai, Molokai, and Hawaii. No population estimates exist for the small nesting colonies that exist on Lehua Rock and Molokai (Day & Cooper, 1995; International Union for Conservation of Nature, 2017; U.S. Fish and Wildlife Service, 2005a). About 20 breeding colonies of Newell's shearwaters are known in the main Hawaiian Islands, but others probably exist (International Union for Conservation of Nature, 2017). In 1992, 11 colonies were known on Kauai. There is evidence but no confirmation of nesting on Oahu, Maui, and Lanai (U.S. Fish and Wildlife Service, 2005a).

Newell's shearwaters nest on Kauai at high elevations (525–3,935 ft.) on steep, densely vegetated mountain slopes and in burrows or deep rock crevices, although a substantial number also nest on dry sparsely vegetated cliffs on the Na Pali coast of Kauai and on Lehua Island (Reynolds & Ritchotte, 1997; U.S. Fish and Wildlife Service, 2005a). The use of steep slopes (mostly greater than 65 degrees) for nesting is probably a consequence of predation pressure from introduced pigs, mongooses, and cats; they select sites where there is either an open canopy of trees and ground cover of uluhe ferns or a dense ground cover of tussock grasses (International Union for Conservation of Nature, 2017).

On the Island of Hawaii, Newell's shearwaters fly over the entire island except the southwestern coast. Shearwaters are most numerous flying to and from the Kohala Mountains on the north coast (Day et al., 2003). During adult presence in the breeding season (April to September), Newell's shearwaters gather on the water close to shore before they fly inland around sunset (International Union for Conservation of Nature, 2017). Based on known or suspected colony locations, Newell's shearwaters are expected to be found gathering in early evening at Niihau (north end around Lehua Rock), Kauai, Oahu, Maui, Molokai, Lanai, and Hawaii from April to September.

Open Ocean

During the breeding season, some Newell's shearwaters forage west and north of the Hawaiian Islands so that the central part of their marine range moves northward in the Transit Corridor portion of the Study Area (International Union for Conservation of Nature, 2017; U.S. Fish and Wildlife Service, 2005a).

C.8.2.5.3 Population Trends

Results from 1980 and 1994 surveys estimated Newell's shearwater pelagic populations at 84,000 individuals (with a 95 percent confidence interval of 57,000 to 115,000) and from this the breeding population is estimated at 16,700 to 19,300 pairs. Radar data from 1993 and 1999–2001 across 13 sites indicates a 60 to 62 percent decline in numbers visiting Kauai, while recoveries of stranded young birds showed a 72 percent decline over the same time period (Day & Cooper, 1995; Day et al., 2003). From 1998 to 2011 surveys, the at-sea populations are estimated at 27,011 (with a 95 percent confidence interval of 18,254 to 37,125) (U.S. Fish & Wildlife Service, 2016).

Population models incorporating best estimates of breeding effort and success yielded a population decline of 3.2 percent annually. When variables estimating the anthropogenic mortality suffered by the species (predation, light attraction, and collision) were included, these models predicted a population decline of 30 to 60 percent over 10 years. Combining this with longer term declines owing to habitat loss, introduced predators, disorientation owing to urban lighting and collision with powerlines, the species is estimated to be declining at rates exceeding 50 percent over 47 years, three generations (International Union for Conservation of Nature, 2017). Population in the 1980s and early 1990s was estimated at about 84,000, but numbers in 2000 may have been only 21 percent of what they were in 1987 (U.S. Fish and Wildlife Service, 2005a). The largest known population, found on Kauai, was devastated by two hurricanes in 1982 and 1992. Since that last storm, the species has been in steady decline on Kauai. The remaining adults and fledglings are suffering significant deaths from utility pole and line strikes (International Union for Conservation of Nature, 2017). Between 1978 and 2007, more than 30,000 Newell's shearwaters were picked up by island residents from Kauai's highways, athletic fields, and hotel grounds (U.S. Fish and Wildlife Service, 2012c). Continuing forest habitat destruction and predation from introduced mammals are also taking a toll on this species (International Union for Conservation of Nature, 2017).

C.8.2.5.4 Population Threats

Historical threats to Newell's shearwater included subsistence hunting by Polynesians and predation by introduced species (a continuing threat) including rats, dogs, pigs, barn owls, feral cats, and the small Indian mongoose (*Herpestes auropunctatus*) (Hawaii Department of Land and Natural Resources, 2005).

Recently, Young et al. (2023) successfully translocated 86 Newell's shearwaters (along with 110 Hawaiian petrels) to a predator free enclosure on Kauai, establishing the first predator-free colony for Newell's shearwaters.

Current threats include hooking and drowning on commercial longline gear (McCracken, 2014), artificial lights (e.g., street and resort lights) along the coast that blind and disorient fledglings. Once on the ground, these fledglings are unable to fly and thousands are killed each year by cars, cats, and dogs. In addition, adults can collide with power facilities and associated utility wires and associated lines are in the direct path of known Newell's flight corridors. Additional threats are the loss and degradation of forested habitat caused by introduced plants and herbivores.

On Kauai, hurricanes Iwa and Iniki devastated the forests in 1982 and 1992 (Ainley et al., 1997) and, since the latter, the species' population has been declining. Given that a large proportion of the population breeds on Kauai, catastrophic events, like hurricanes, are a serious threat (International Union for Conservation of Nature, 2017).

C.8.2.6 Band-rumped Storm-petrel (Oceanodroma [Hydrobates] castro), Hawaii Distinct Population Segment

The Hawaii DPS of band-rumped storm-petrel is also known as the Hawaiian storm-petrel (American Ornithologists' Union, 1998; Harrison, 1983). It is intermediate in many respects between the Wilson's and Leach's storm-petrels. Plumage is blackish-brown overall with pale wing bars and a clear, curved white band across rump; white on rump is more extensive than on Leach's but less than on Wilson's (where white extends fully onto undertail coverts). The band-rumped storm-petrel is difficult to identify reliably at sea (U.S. Fish and Wildlife Service, 2017b).

C.8.2.6.1 Status and Management

Storm-petrels are the smallest of all the oceanic seabirds (Onley & Scofield, 2007). The Hawaiian population had been a candidate for listing under the ESA since 1989 (U.S. Fish and Wildlife Service, 2004), and was listed as an endangered DPS in 2016 (81 FR 67786). The global population is not a conservation concern due to large populations in Japan and the Galapagos Islands (International Union for the Conservation of Nature and Natural Resources, 2010; U.S. Fish and Wildlife Service, 2005a). In the Hawaiian Islands, band-rumped storm-petrels are the rarest breeding seabirds (International Union for the Conservation of Nature and Natural Resources, 2010; U.S. Fish and Wildlife Service, 2005a). The State of Hawaii categorizes the local population as endangered (U.S. Fish and Wildlife Service, 2005a) and regards it as highly imperiled within the Hawaiian Islands Bird Conservation Region, based on population size, breeding distribution, and threats to breeding distribution (U.S. Fish and Wildlife Service, 2003).

C.8.2.6.2 Habitat and Geographic Range

Hawaiian Band-rumped storm-petrels prefer warm deep water of 1,000 m to more than 2,000 m deep. This species occurs close to land where deep water is near an island; otherwise, they occur offshore or in upwelling regions (International Union for the Conservation of Nature and Natural Resources, 2010). Preferred waters range from 80 to 84 °F (International Union for the Conservation of Nature and Natural Resources, 2010). Nesting habitat in the main Hawaiian Islands consists of steep cliffs and barren lava flows at high elevations. Nests are in burrows or crevices in rock or lava (International Union for the Conservation of Nature and Natural Resources, 2010; U.S. Fish and Wildlife Service, 2004, 2005a). Once widespread along the Hawaiian island chain, as evidenced by midden sites across the main Hawaiian Islands, its range is now isolated to pockets of high elevation nesting habitat (Price & Antaky, 2020). Band-rumped storm-petrels have been documented using artificial nest boxes (Beard et al., 2022; Price

& Antaky, 2020; Raine et al., 2017). These sites may well be the last resort of predator avoidance for a species that formerly most likely nested closer to the coast (International Union for the Conservation of Nature and Natural Resources, 2010).

Band-rumped storm-petrels most likely feed on small fish, squid, and crustaceans, based on records from the Galapagos Islands; diet information is not available for Hawaiian birds (U.S. Fish and Wildlife Service, 2005a). Foraging is confirmed diurnally and suspected nocturnally. Food is captured while sitting on the water or off the surface by bill snatching as the bird gently flaps just above the surface of the water (International Union for the Conservation of Nature and Natural Resources, 2010). Foraging occurs mostly in deep water in all seasons. They are attracted to discarded fish by-product from fishing boats (Onley & Scofield, 2007). Band-rumped storm-petrels are vulnerable to predation by introduced rats, mice, cats, mongooses, pigs, and barn owls (U.S. Fish and Wildlife Service, 2005a).

Insular Pacific-Hawaiian Large Marine Ecosystem

Band-rumped storm-petrels occur in coastal waters of the Hawaii portion of the Study Area and into the western portion of the Transit Corridor portion of the Study Area. Colonies in the main Hawaiian Islands are known or suspected on Lehua Island, Kauai, Maui, Kahoolawe, and Hawaii. Other colonies are likely in Waimea Canyon and Hanapepe Valley on the western side of Kauai. On Hawaii, one small population is known to nest on the upper west slope of Mauna Loa. Nesting on remote cliffs on Lehua Island is also confirmed (81 FR 67786). Vocalizations have been heard, indicating occurrence on Kahoolawe, Lanai, Lehua Rock, and Maui (Haleakala Crater) (International Union for the Conservation of Nature and Natural Resources, 2010; Raine et al., 2017; U.S. Fish and Wildlife Service, 2004). There is no known nesting in the Northwestern Hawaiian Islands (U.S. Fish and Wildlife Service, 2004).

During the nesting season, deep water (more than 1,000 m) close to shore can be used for foraging. Fishermen report them mostly at about 3 mi. off the Na Pali coast of Kauai (International Union for the Conservation of Nature and Natural Resources, 2010; Price & Antaky, 2020). Band-rumped storm-petrels are known to gather in nearshore waters before they fly inland to nesting colonies in the early evening.

Open Ocean

Band-rumped storm-petrels occur in the Hawaii portion of the Study Area and the western portion of the Transit Corridor Study Area. They are distributed in the Pacific from Japan east to Central America and northern South America (Harrison, 1983). Pacific populations are divided into distinct Japanese, Hawaiian, and Galapagos breeding populations (U.S. Fish and Wildlife Service, 2004). The Hawaiian population at sea is thought to remain in the central Pacific, ranging south to the Equatorial Countercurrent. Some individuals spend most of their time in open ocean, occurring far offshore from nesting islands; others seem to remain close to nesting colonies year-round (U.S. Fish and Wildlife Service, 2005a).

C.8.2.6.3 Population Trends

The global population of band-rumped storm-petrels is estimated to be 20,000 to 200,000 individuals. Due to the difficulty in studying this species, the number of birds breeding in Hawaii is currently unknown but is thought to be in the low hundreds (Kauai Endangered Seabird Recovery Project, 2017; U.S. Fish and Wildlife Service, 2021c). The Hawaiian population appears to be significantly reduced in numbers following human occupation of the Hawaiian Islands (81 FR 67786) and is likely a tiny remnant of historical numbers (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2005). In 2004, the population of band-rumped storm-petrels at sea was estimated at about 5,500 (U.S. Fish and

Wildlife Service, 2004). In 2002, the population on Kauai was estimated at 171 to 221 breeding pairs, mostly occurring along the Na Pali coast (Pohakuao Valley, Kalalau Valley, Awaawapuhi Valley, Nuololo Aina, and Nuololo Kay) on the west side of the island.

C.8.2.6.4 Population Threats

The small population size and limited distribution of the band-rumped storm-petrel in Hawaii threaten this endangered population by increasing the potential population consequences of other threats, including natural catastrophes such as hurricanes and landslides (81 FR 67786). The band-rumped storm-petrel is highly vulnerable to predation by introduced rats, mice, cats, mongooses, pigs, and barn owls, as well as being vulnerable to striking power lines and street lights at night (U.S. Fish and Wildlife Service, 2005a). Street and resort lights disorient fledglings, causing them to collide with structures or fall to the ground, where they are at risk from predators and cars. Additional threats are commercial fishing and ocean pollution, and the loss and degradation of forested habitat caused by introduced plants and herbivores.

C.8.3 Species Not Listed under the Endangered Species Act

At least 195 species of birds are found within the Study Area that are not listed under the ESA. The major groups of birds are described in Section C.8.2 (Endangered Species Act-Listed Species) and Section C.8.3 (Species Not Listed under the Endangered Species Act), and Section C.8.3.9 (Migratory Birds) describes species that are protected and of conservation concern under the Migratory Bird Treaty Act (MBTA) and Bald and Golden Eagle Protection Act.

C.8.3.1 Geese, Swans, Dabbling and Diving Ducks (Order Anseriformes)

There are 50 species of swans, geese, and dabbling and diving ducks in the family Anatidae in North America. No birds from this group are considered Birds of Conservation Concern (BCC) (U.S. Fish and Wildlife Service, 2008a). Birds from this group range from dabbling ducks found in coastal bays, estuaries, and lagoons to more open water ducks found in deeper water environments. Several of these species are diving ducks that inhabit nearshore or offshore waters of the Study Area, and San Diego Bay in particular (Sibley, 2014). Scaups (*Aythya spp.*) and surf scoters (*Melanitta perspicillata*) are abundant during winter throughout San Diego Bay, diving and foraging on the bottom from shallow (intertidal) to relatively deep (> 20 ft.) waters (U.S. Department of the Navy, 2013d).

Representative species that can be found in the Study Area include geese (e.g., Canada goose [*Branta canadensis*], brant [*Branta bernicla*]); dabbling ducks (e.g., mallard [*Anas platyrhynchos*], American wigeon [*Anas americana*], northern shoveler [*Anas clypeata*]); diving ducks (e.g., bufflehead [*Bucephala albeola*], greater scaup [*Aythya marila*], lesser scaup [*Aythya affinis*], and red-breasted merganser [*Mergus serrator*]); and scoters (e.g., surf scoter [*Melanitta perspicillata*], black scoter [*Melanitta americana*]) (American Ornithologists' Union, 1998).

C.8.3.2 Loons (Order Gaviiformes)

There are five species of loons in the family Gaviidae in North America (American Ornithologists' Union, 1998), three of which occur in the Study Area. The common loon (*Gavia immer*) and the red-throated loon (*G. stellata*) are BCC (U.S. Fish and Wildlife Service, 2008a). Loons are medium to large fish-eating birds that capture prey by diving underwater (Sibley, 2014). Loons can dive down to 250 ft. with an average dive time of 40 seconds (Sibley, 2014). Loons move ashore only to breed, and all loon species nest on banks of inland ponds or lakes, requiring specific habitat features such as undeveloped shoreline and nest sites that have steep drop offs so they can approach their nest from underwater (Cornell Lab of

Ornithology, 2009). For example, common loons spend their time in both freshwater and saltwater environments but prefer to nest on islands where the shoreline is not developed. Most loons need about 100 ft. of room to take off, so size is another habitat feature that is important for nesting areas. During migration, loons fly high above land or water in loose groups or singly. They winter in coastal, nearshore, or open water marine habitats (Sibley, 2014). For example, the Pacific loon (*G. pacifica*) prefers deep water and is found on the open ocean and in bays. The red-throated loon, a representative species within the Study Area, has a circumpolar distribution, breeds in high latitudes on remote ponds, and winters along the Atlantic and Pacific coasts (American Ornithologists' Union, 1998).

C.8.3.3 Albatrosses, Fulmars, Petrels, Shearwaters, and Storm-Petrels (Order Procellariiformes)

The Procellariiformes is a large order of open ocean seabirds that are divided into four families: Diomedeidae (albatrosses), Procellariidae (petrels and shearwaters), Hydrobatidae (storm-petrels), and Pelecanoididae (diving-petrels) (Enticott & Tipling, 1997; Onley & Scofield, 2007). These seabirds are generally long-lived, breed once a year, and lay only one egg, thus, they have a low reproductive output. They have extremely broad distributions and include all marine birds that spend most of their lives at sea and exclusively feed in the open ocean, primarily on fish, crustaceans, and crabs. They can be found in high numbers resting on the water in flocks where prey is concentrated (Enticott & Tipling, 1997). Some species feed around fishing boats or become injured from longline gear (Enticott & Tipling, 1997) (Onley & Scofield, 2007). They nest in colonies on remote islands uninhabited by people. Some are ground nesters; others nest in cavities or burrows (Ramos et al., 1997). They return to their birth colonies. Most species of this order are monogamous and mate for life. Both parents participate in egg incubation and chick rearing (Elphick et al., 2001). Representative species that occur in the Study Area include Laysan albatross (*Phoebastria immutabilis*), Northern fulmar (*Fulmaris glacialis*), mottled petrel (*Pterodroma inexpectata*), pink-footed shearwater (*Puffinus creatopus*), and Wilson's storm-petrel (*Oceanites oceanicus*).

C.8.3.4 Boobies, Gannets, Cormorants, and Frigatebirds (Order Suliformes)

The Suliformes order is a diverse group of large seabirds including anhingas, gannets, boobies, cormorants, and frigatebirds. This order is composed of 16 species in 4 families – 12 species representing 2 families that occur within the Study Area. Three of these species are considered BCC (U.S. Fish and Wildlife Service, 2008a). Species of concern within the Study Area include the brown booby (*Sula leucogaster*), great cormorant (*Phalacrocorax carbo*), and magnificent frigatebird (*Fregata magnificens*) (American Ornithologists' Union, 1998).

Suliformes are less pelagic than the Procellariiformes, although some of these species such as frigatebirds are pelagic. Most species are colonial, feed on fish, and use a variety of breeding habitats including trees and bushes (but not burrows). Breeding strategies vary among species, with some being long-lived and having low breeding success, while others have higher annual breeding success, but higher annual adult death (Enticott & Tipling, 1997; Onley & Scofield, 2007).

Cormorants are voracious predators on inshore fishes and have been implicated as a major threat to the recovery efforts of Atlantic salmon in the Gulf of Maine where they feed on juvenile salmon (smolts) leaving the estuaries (Fay et al., 2006; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2005). Their offshore foraging range is limited by their need for undisturbed, dry nocturnal roosting sites (Shields et al., 2002).

Boobies and gannets are large seabirds that plunge from the air to capture their prey. Filling similar niches, boobies inhabit warmer areas and gannets colder regions. Boobies and gannets often nest on islands in colonies, with gannets nesting on cliffs (BirdLife International, 2012) and boobies generally on the ground if predators allow (Pratt et al., 1987). They forage offshore, often in large flocks, feeding on fish but also foraging at night, often feeding on squid (Pratt et al., 1987).

Like tropicbirds and pelicans, members of this group all have webbed feet and eight toes, and all have a throat sac, called a gular sac (Brown & Harshman, 2008). This sac is highly developed and visible in pelicans and frigatebirds but is also readily apparent in boobies and cormorants. Pelicans use the sac to trap fish, frigatebirds use it as a mating display and to feed on fish, squid, and similar marine life (Dearborn et al., 2001), and cormorants and boobies use the sac for heat regulation. These birds nest in colonies, but individual birds are monogamous (Brown & Harshman, 2008).

C.8.3.5 Tropicbirds (Order Phaethontiformes)

Tropicbirds are medium-sized seabirds, predominately white with black patterning on the back, wings, and face. They have thick, pointed bills that are red or orange in color that are slightly decurved. Their most notable feature is the extremely long and narrow central tail feathers, which can be 11 to 22 in. long. Their wingspans average around 3 ft. Superficially, tropicbirds resemble terns. Tropicbirds are plunge divers that feed on fish and are highly pelagic foragers in tropical and subtropical oceans, coming to land mainly to breed (Sibley, 2014). Red-billed tropicbirds (*Phaethon aethereus*) are rare visitors to the Study Area in both southern California and Hawaiian waters, whereas white-tailed and red-tailed tropicbirds (*P. lepturus* and *P. rubricauda*, respectively) nest in the Hawaiian Islands occur in the surrounding waters (Sibley, 2014).

C.8.3.6 Pelicans, Herons, Egrets, Ibis, and Spoonbills (Order Pelecaniformes)

Pelecaniformes is a large group composed of long-legged, large billed species that includes pelicans, herons, egrets, ibis, and spoonbills. However, with the exception of two species of pelicans (described below), they are inhabitants of freshwater marshes and are unlikely to occur in the Study Area.

The brown pelican (*Pelecanus occidentalis*) primarily occurs in shallow (less than 150 ft.) warm coastal marine and estuarine environments, as well as offshore where they forage primarily on fish by head first plunge-diving. Most plunge-diving is limited to 1–2 m within the water column. Foraging occurs within 20 km of nesting islands during the breeding season, and up to 75 km offshore during the nonbreeding season (Shields et al., 2002). American white pelicans (*Pelecanus erythrorhynchos*) are found in shallow coastal bays, inlets, and estuaries that support forage fish (Knopf & Evans, 2004). Flocks forage cooperatively, swimming and encircling fish as a coordinated group or driving them into shallows, where they are caught with synchronized bill dipping (Enticott & Tipling, 1997; Onley & Scofield, 2007).

C.8.3.7 Osprey, Bald Eagles, Kites and Falcons (Orders Accipitriformes and Falconiformes)

Accipitriformes is a large group consisting of 60 species in three families (American Ornithologists' Union, 1998). This order generally has broad wings well suited for soaring. Falconiformes include 9 North American species that, with the exception of the caracara (*Caracara cheriway*), are fast-flying predators with pointed wings and a streamline body shape (Sibley, 2014). Members of both orders hunt by day and feed on a variety of prey, including fish, small mammals, reptiles, and carrion. Species that are likely to occur within the Study Area include the osprey (*Pandion haliaetus)* bald eagle (*Haliaeetus leucocephalus*), peregrine falcon (*Falco peregrinus*), and swallow-tailed kite (*Elanoides forficatus*). The bald eagle, peregrine falcon, and swallow-tailed kite are BCC.

Ospreys live near slow-moving waters of coastal, nearshore, and freshwater environments in many parts of the Study Area. They are plunge feeders but also have the ability to capture prey with their feet while keeping their head above water. Fish make up a large portion of their diet, and therefore, their vision is well adapted to detecting underwater objects from 10 to 40 m above water (Poole et al., 2002). Ospreys nest around the shores of San Diego Bay and forage in the waters of the bay (U.S. Department of the Navy, 2013d).

C.8.3.8 Shorebirds, Phalaropes, Gulls, Noddies, Terns, Skimmers, Skuas, Jaegers, and Alcids (Order Charadriiformes)

Shorebirds are small, generally long-legged coastal birds, many of which forage below the high tide in the surf zone by picking and probing for small aquatic prey (Sibley, 2014). Shorebirds undergo some of the longest distance migrations known for birds, for example, the red knot annually migrates more than 15,000 km. Though most of their life cycle is spent in coastal areas, shorebird migration over open ocean does occur (Elphick, 2007). Although taxonomically grouped among some shorebirds, two species of phalaropes in the family Scolopacidae that occur within the Study Area are functionally seabirds, spending the nonbreeding months out on the open ocean. These include the red-necked phalarope (*Phalaropus lobatus*) and red phalarope (*Phalaropus fulicarius*), both of which breed in high arctic habitats but spend migrate and winter at sea, gathering in small flocks at upwellings and convergence zones, foraging on zooplankton and other small aquatic animals that rise to the surface (Rubega et al., 2000).

Gulls, noddies, and terns in the family Laridae are a diverse group of small to medium sized seabirds that inhabit coastal, nearshore, and open sea waters. Skuas and jaegers in the family Stercorariidae are stocky powerful birds with long pointed wings, long tails, strong hooked bills, and sharp talons known for robbing the food of smaller seabirds, teasing and harassing them until they drop their prey. Murres, murrelets, and auklets in the family Alcidae are good swimmers and divers and have short wings, which require them to flap their wings rapidly to fly.

Species in the order Charadriiformes occupy diverse habitats. Some species in this order spend most of their time at sea (e.g., jaegers, skuas, alcids), whereas others are more coastal or near shore (e.g., gulls). Many charadriiforms inhabit marine and freshwater wetlands; others spend most of their lives in or near the ocean. Many species breed in colonies, and some species lay more than one egg (Ericson et al., 2003; Fain & Houde, 2007; Harrison, 1983; Onley & Scofield, 2007). Representative species within the Study Area include semipalmated plover (*Charadrius semipalmatus*), Bonaparte's gull (*Larus philadelphia*), black-legged kittiwake (*Rissa tridactyla*), Forster's tern (*Sterna forsteri*), parasitic jaeger (*Stercorarius parasiticus*), common murre (*Uria aalge*), sooty shearwater (*Puffinus tenuirostris*), and rhinoceros auklet (*Cerorhinca monocerata*).

C.8.3.9 Migratory Birds

Most of the bird species that would be encountered in the Study Area are listed under the MBTA (U.S. Fish and Wildlife Service, 2015). The MBTA established federal responsibilities for protecting nearly all migratory species of birds as defined in the Act, their eggs, and nests. Under the MBTA, migratory bird means any bird, whatever its origin and whether or not raised in captivity, which belongs to a species listed in Section 10.13 of the MBTA, or which is a mutation or a hybrid of any such species, including any part, nest, or egg of any such bird, or any product, whether or not manufactured, which consists, or is composed in whole or part, of any such bird or any part, nest, or egg thereof. Under the MBTA regulations applicable to military readiness activities (50 CFR part 21), the USFWS has promulgated a rule that authorizes the incidental take of migratory birds provided they do not result in a significant impact on the population of a migratory bird species. Of the 1,027 species protected under the MBTA

(U.S. Fish and Wildlife Service, 2013), over 100 species occur in the Study Area. These species are not analyzed individually, but rather are grouped based on taxonomic or behavioral similarities based on the stressor that is being analyzed.

BCC are species, subspecies, and populations of migratory and non-migratory birds that the USFWS has determined to be the highest priority for conservation actions (U.S. Fish and Wildlife Service, 2021a).

The purpose of the BCC list is to prevent or remove the need for additional ESA bird listings by implementing proactive management and conservation actions needed to conserve these species. Of the species that occur within the Study Area, 15 are considered BCC (Table C-7). These species are not analyzed individually, but rather are grouped by taxonomic or behavioral similarities based on the stressor that is being analyzed.

Table C-7: Birds of Conservation Concern that Occur within the Study Area

Order/Family	***Common Name***	***Scientific Name***
Order Procellariiformes		
Family Diomedeidae		
	Laysan albatross	*Phoebastria immutabilis*
	Black-footed albatross	*Phoebastria nigripes*
Family Procellariidae		
	Pink-footed shearwater	*Puffinus creatopus*
	Christmas shearwater	*Puffinus nativitatis*
	Black-vented shearwater	*Puffinus opisthomelas*
Family Hydrobatidae		
	Ashy storm-petrel	*Oceanodroma homochroa*
	Band-rumped storm-petrel	*Oceanodroma [Hydrobates] castro*
	Tristram’s storm-petrel	*Oceanodroma tristrami*
Order Falconiformes		
Family Falconidae		
	Peregrine falcon	*Falco peregrinus*
Order Charadriiformes		
Family Lardiae		
Subfamily Sterninae	Blue noddy	*Procelsterna cerulean*
	Gull-billed tern	*Sterna nilotica*
Subfamily Rynchopinae	Black skimmer	*Rynchops niger*
Family Ardeidae		
	Guadalupe murrelet	*Synthliboramphus hypoleucus*
	Scripps’s murrelet	*Synthliboramphus scrippsi*
	Cassin’s auklet	*Ptychoramphus aleuticus*

C.8.3.9.1 Grebes (Order Podicipediformes)

There are seven species of grebes in the family Podicipedidae in North America, all of which occur in the Study Area (American Ornithologists' Union, 1998). Two of these species, the pied-billed grebe (*Podilymbus podiceps*) and horned grebe (*Podiceps auritus*) are BCC (U.S. Fish and Wildlife Service, 2008a). Grebes can be found in a variety of aquatic habitats ranging from seasonally flooded scrubland and roadside ditches to deep lakes and coastal bays. Most grebe species winter in open waters while preferring marshy, vegetated habitats during the summer months (Sibley, 2014). Grebes forage by diving for small aquatic animals such as insects, fish, and crustaceans in the water column. For example, horned grebes can dive for up to 3 minutes and travel 500 ft. underwater, where they are sometimes preyed upon by sharks and orcas (Ehrlich et al., 1988). Grebes tend to escape predators by diving or sinking, leaving only the head exposed, rather than taking flight. All grebe species build floating nests in marshes and winter on the ocean and nearshore coastal areas (Sibley, 2014).

References

Abadía-Cardoso, A., D. E. Pearse, S. Jacoson, J. Marshall, D. Dalrymple, F. Kawasaki, G. Ruiz-Campos, and J. C. Garza. (2016). Population genetic struture and ancestry of steelhead/rainbow trout (*Oncorhynchus mykiss*) at the extreme southern edge of their range in North America. 675–689.

Abecassis, M., J. Polovina, R. W. Baird, A. Copeland, J. C. Drazen, R. Domokos, E. Oleson, Y. Jia, G. S. Schorr, D. L. Webster, and R. D. Andrews. (2015). Characterizing a Foraging Hotspot for Short-Finned Pilot Whales and Blainville's Beaked Whales Located off the West Side of Hawaiʻi Island by Using Tagging and Oceanographic Data. *PLoS ONE 10* (11): 1–22. DOI:10.1371/journal.pone.0142628

Abrahms, B., H. Welch, S. Brodie, M. G. Jacox, E. Becker, S. J. Bograd, L. Irvine, D. Palacios, B. Mate, and E. Hazen. (2019). Dynamic ensemble models to predict distributions and anthropogenic risk exposure for highly mobile species. *Diversity and Distributions 00* 1–12. DOI:10.1111/ddi.12940

Aebischer, N. J., J. C. Coulson, and J. M. Colebrook. (1990). Parallel long-term trends across four marine trophic levels and weather. *Nature 347* (6295): 753–755.

AECOM Technical Services. (2019). *Basis of Design Field Investigation Report: Pearl Harbor Sediment*. Honolulu, HI: Naval Facilities Engineering Command Pacific.

Afsal, V. V., P. P. Manojkumar, K. S. S. M. Yousuf, B. Anoop, and E. Vivekanandan. (2009). The first sighting of Longman's beaked whale, *Indopacetus pacificus* in the southern Bay of Bengal. *Marine Biodiversity Records 2*. DOI:10.1017/s1755267209990510

Agency for Toxic Substances and Disease Registry. (1999). *Toxicological Profile for Mercury*. Atlanta, GA: U.S. Department of Health and Human Services, Public Health Service.

Agency for Toxic Substances and Disease Registry. (2005). *Public Health Assessment for Pearl Harbor Naval Complex*. Pearl Harbor, HI: Department of Health and Human Services.

Aguayo, L. A. and T. R. Sanchez. (1987). Sighting records of Fraser's dolphin in the Mexican Pacific waters. *The Scientific Reports of the Whales Research Institute 38* 187–188.

Aguilar de Soto, N., M. P. Johnson, P. T. Madsen, F. Diaz, I. Dominguez, A. Brito, and P. Tyack. (2008). Cheetahs of the deep sea: Deep foraging sprints in short-finned pilot whales off Tenerife (Canary Islands). *The Journal of Animal Ecology 77* (5): 936–947. DOI:10.1111/j.1365-2656.2008.01393

Ainley, D. G., C. T. Thomas, and M. H. Reynolds. (1997). *Townsend's and Newell's Shearwater (Puffinus auricularis)* (The Birds of North America Online). Ithaca, NY: Cornell Lab of Ornithology.

Aki, K., R. Brock, J. Miller, J. R. Mobley, Jr., P. J. Rappa, D. Tarnas, and M. Yuen. (1994). *A Site Characterization Study for the Hawaiian Islands Humpback Whale National Marine Sanctuary*. Honolulu, HI: University of Hawaii Sea Grant College Program School of Ocean and Earth Science and Technology.

Alaska Department of Fish and Game. (2024). *Coho Salmon*. Retrieved January 4, 2024, from https://www.adfg.alaska.gov/index.cfm?adfg=cohosalmon.main.

Alfaro-Shigueto, J., P. H. Dutton, J. Mangel, and D. Vega. (2004). First confirmed occurrence of loggerhead turtles in Peru. *Marine Turtle Newsletter 103* 7–11.

Allen, C. D., G. E. Lemons, T. Eguchi, R. A. LeRoux, C. C. Fahy, P. H. Dutton, S. H. Peckham, and J. A. Seminoff. (2013). Stable isotope analysis reveals migratory origin of loggerhead turtles in the Southern California Bight. *Marine Ecology Progress Series 472* 275–285.

Allen, J. A. (1998). Mangroves as alien species: The case of Hawaii. *Global Ecology and Biogeography Letters 7* (1): 61–71.

Allen, L. G. and J. N. Cross. (2006). Surface waters. In. In L. G. Allen, D. J. Pondella, II, & M. H. Horn (Eds.), *The Ecology of Marine Fishes: California and Adjacent Waters* (pp. 320–341). Berkeley, CA: University of California Press.

Allen, L. G. and D. J. Pondella, II. (2006). Surf zone, coastal pelagic zone, and harbors. In L. G. Allen, D. J. Pondella, II, & M. H. Horn (Eds.), *The Ecology of Marine Fishes: California and Adjacent Waters*. Berkeley, CA: University of California Press.

Allen, L. G., D. J. Pondella, II, and M. H. Horn. (2006). The Ecology of Marine Fishes: California and Adjacent Waters (pp. 660). Berkeley, CA: University of California Press.

Allen, M. J. (2006). Pollution. In L. G. Allen, D. J. Pondella, II, & M. H. Horn (Eds.), *The Ecology of Marine Fishes: California and Adjacent Waters* (pp. 595–610). Berkeley, CA: University of California Press.

Alter, S. E., M. P. Simmonds, and J. R. Brandon. (2010). Forecasting the consequences of climate-driven shifts in human behavior on cetaceans. *Marine Policy 34* (5): 943–954. DOI:doi:10.1016/j.marpol.2010.01.026

American Ornithological Society. (2018). *Checklist of North and Middle American Birds*. Retrieved from http://checklist.aou.org/taxa.

American Ornithologists' Union. (1998). *The AOU Check-List of North American Birds* (7th ed.). Washington, DC: American Ornithologists' Union.

Anderson, D. M., P. M. Glibert, and J. M. Burkholder. (2002). Harmful algal blooms and eutrophication: Nutrient sources, composition, and consequences. *Estuaries 25* (4, Part B): 704–726.

Anderson, D. W., C. J. Henny, C. Godinez-Reyes, F. Gress, E. L. Palacios, K. Santos del Prado, and J. Bredy. (2007). *Size of the California Brown Pelican Metapopulation during a non-El Niño year*. Reston, VA: U.S. Geological Survey.

Anderson, O., C. Small, J. Croxall, E. Dunn, B. Sullivan, O. Yates, and A. Black. (2011). Global seabird bycatch in longline fisheries. *Endangered Species Research 14* 91–106. DOI:10.3354/esr00347

Anderson, R. C., R. Clark, P. T. Madsen, C. Johnson, J. Kiszka, and O. Breysse. (2006). Observations of Longman's Beaked Whale (*Indopacetus pacificus*) in the Western Indian Ocean. *Aquatic Mammals 32* (2): 223–231. DOI:10.1578/am.32.2.2006.223

Andrady, A. (2015). Persistence of plastic litter in the oceans. In M. Bergmann, L. Gutow, & M. Klages (Eds.), *Marine Anthropogenic Litter*. New York, NY: Springer International Publishing.

Antonelis, G. A., J. D. Baker, T. C. Johanos, R. C. Braun, and A. L. Harting. (2006). Hawaiian monk seal (*Monachus schauinslandi*): Status and conservation issues. *Atoll Research Bulletin 543* 75–101.

Anttila, C. K., C. C. Daehler, N. E. Rank, and D. R. Strong. (1998). Greater male fitness of a rare invader (*Spartina alterniflora*, Poaceae) threatens a common native (*Spartina foliosa*) with hybridization. *American Journal of Botany 85* 1597–1601.

Appeltans, W., P. Bouchet, G. A. Boxshall, K. Fauchald, D. P. Gordon, B. W. Hoeksema, G. C. B. Poore, R. W. M. van Soest, S. Stöhr, T. C. Walter, and M. J. Costello. (2010). *World Register of Marine Species*. Retrieved from http://www.marinespecies.org/index.php.

Aquatic Mammals. (2015a). Supplemental tables: Biologically important areas for cetaceans within U.S. Waters – Hawaii region. *Aquatic Mammals 41* (1): 33–44. DOI:10.1578/AM.41.1.2015.3

Aquatic Mammals. (2015b). Supplemental tables: Biologically important areas for selected cetaceans within U.S. Waters – West Coast region. *Aquatic Mammals 41* (1): 30–32. DOI:10.1578/AM.41.1.2015.3

Aragones, L., M. Roque, M. Flores, R. Encomienda, G. Laule, B. Espinos, F. Maniago, G. Diaz, E. Alesna, and R. Braun. (2010). The Philippine Marine Mammal Strandings from 1998 to 2009: Animials in the Philippines in Peril? *Aquatic Mammals 36* (3): 219–233. DOI:10.1578/AM.36.3.2010.219

Archer, F. I., II. (2009). Striped dolphin, *Stenella coeruleoalba*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 1127–1129). Academic Press.

Ariano-Sánchez, D., C. Muccio, F. Rosell, and S. Reinhardt. (2020). Are trends in Olive Ridley sea turtle (*Lepidochelys olivacea*) nesting abundance affected by El Niño Southern Oscillation (ENSO) variability? Sixteen years of monitoring on the Pacific coast of northern Central America. *Global Ecology and Conservation 24*.

Arnold, T., C. Mealey, H. Leahey, A. W. Miller, J. M. Hall-Spencer, M. Milazzo, and K. Maers. (2012). Ocean acidification and the loss of phenolic substances in marine plants. *PLoS ONE 7* (4): e35107–e35107. DOI:10.1371/journal.pone.0035107

Asada, A., S. A. Eckert, F. P. Gelwick, W. H. Hagey, and R. W. Davis. (2022). Diving behavior and energetic strategy of leatherback sea turtles during internesting intervals on St. Croix, US Virgin Islands. *Journal of Experimental Marine Biology and Ecology 550* 151722.

Asch, R. G. (2015). Climate change and decadal shifts in the phenology of larval fishes in the California Current ecosystem. *PNAS* 4065–4074.

Aschettino, J., D. Engelhaupt, A. Engelhaupt, J. Bell, and J. B. Thornton. (2015, 13-18 December 2015). *Humpback Whale Presence and Habitat-Use in High-Traffic Areas off Virginia*. Presented at the 21st Biennial Conference on the Biology of Marine Mammals. San Francisco, CA.

Aschettino, J. M., R. W. Baird, D. J. McSweeney, D. L. Webster, G. S. Schorr, J. L. Huggins, K. K. Martien, S. D. Mahaffy, and K. L. West. (2012). Population structure of melon-headed whales (*Peponocephala electra*) in the Hawaiian Archipelago: Evidence of multiple populations based on photo identification. *Marine Mammal Science 28* (4): 666–689. DOI:10.1111/j.1748-7692.2011.00517

Ateweberhan, M., D. A. Feary, S. Keshavmurthy, A. Chen, M. H. Schleyer, and C. R. Sheppard. (2013). Climate change impacts on coral reefs: Synergies with local effects, possibilities for acclimation, and management implications. *Marine Pollution Bulletin 74* (2): 526–539. DOI:10.1016/j.marpolbul.2013.06.011

Ateweberhan, M. and T. R. McClanahan. (2010). Relationship between historical sea-surface temperature variability and climate change-induced coral mortality in the western Indian Ocean. *Marine Pollution Bulletin 60* (7): 964–970. DOI:doi:10.1016/j.marpolbul.2010.03.033

Atwood, J. L. and D. E. Minsky. (1983). Least tern foraging ecology at three major California breeding colonies. *Western Birds 14* (2): 57–71.

Au, D. W. K. and W. L. Perryman. (1985). Dolphin habitats in the eastern tropical Pacific. *Fishery Bulletin 83* 623–643.

Au, W. W. L. and K. Banks. (1998). The acoustics of the snapping shrimp *Synalpheus parneomeris* in Kaneohe Bay. *The Journal of the Acoustical Society of America 103* (1): 41–47.

Au, W. W. L., J. K. B. Ford, J. K. Horne, and K. A. N. Allman. (2004). Echolocation signals of free-ranging killer whales (*Orcinus orca*) and modeling of foraging for chinook salmon (*Oncorhynchus tshawytscha*). *The Journal of the Acoustical Society of America 115* (2): 901–909. DOI:10.1121/1.1642628

Aurioles-Gamboa, D. and F. J. Camacho-Rios. (2007). Diet and feeding overlap of two otariids, *Zalophus californianus* and *Arctocephalus townsendi*: Implications to survive environmental uncertaintly. *Aquatic Mammals 33* (3): 315–326. DOI:doi: 10.1578/AM.33.3.2007.315

Aurioles-Gamboa, D., F. Elorriaga-Verplancken, and C. J. Hernandez-Camacho. (2010). The current population status of Guadalupe fur seal (*Arctocephalus townsendi*) on the San Benito Islands, Mexico. *Marine Mammal Science 26* (2): 402–408.

Aurioles-Gamboa, D., C. J. Hernandez-Camacho, and E. Rodriguez-Krebs. (1999). Notes on the southernmost records of the Guadalupe fur seal, *Arctocephalus townsendi*, in Mexico. *Marine Mammal Science 15* (2): 581–583.

Aurioles-Gamboa, D. and J. Urban-Ramirez. (1993). Sexual dimorphism in the skull of the pygmy beaked whale (*Mesoplodon peruvianus*). *Revista de Investigacion Cientifica 1* 39–52.

Ayres, K. L., R. K. Booth, J. A. Hempelmann, K. L. Koski, C. K. Emmons, R. W. Baird, K. Balcomb-Bartok, M. B. Hanson, M. J. Ford, and S. K. Wasser. (2012). Distinguishing the impacts of inadequate prey and vessel traffic on an endangered killer whale (*Orcinus orca*) population. *PLoS ONE 7* (6): e36842. DOI:10.1371/journal.pone.0036842

Azzellino, A., S. Gaspari, S. Airoldi, and B. Nani. (2008). Habitat use and preferences of cetaceans along the continental slope and the adjacent pelagic waters in the western Ligurian Sea. *Deep Sea Research Part I: Oceanographic Research Papers 55* (3): 296–323. DOI:10.1016/j.dsr.2007.11.006

Baco, A. R. (2007). Exploration for deep-sea corals on North Pacific seamounts and islands. *Oceanography 20* (4): 108–117.

Bahr, K. D., P. L. Jokiel, and K. S. Rodgers. (2015a). The 2014 coral bleaching and freshwater flood events in Kaneohe Bay, Hawaii. *PeerJ 3* e1136. DOI:10.7717/peerj.1136

Bahr, K. D., P. L. Jokiel, and K. S. Rodgers. (2015b). The 2014 coral bleaching and freshwater flood events in Kaneohe Bay, Hawaii. *Peer-reviewed Journal 3* e1136. DOI:10.7717/peerj.1136

Bahr, K. D., P. L. Jokiel, and K. u. S. Rodgers. (2016). Relative sensitivity of five Hawaiian coral species to high temperature under high-pCO_2 conditions. *Coral Reefs 35* (2): 729–738. DOI:10.1007/s00338-016-1405-4

Bahr, K. D., K. D. Rodgers, and P. L. Jokiel. (2017). Impact of Three Bleaching Events on the Reef Resiliency of Kaneohe Bay, Hawaii. *Frontiers in Marine Science 4* (398): 1–13. DOI:10.3389/fmars.2017.00398

Baird, P. (1993). *Ecology of the Southern California Bight: Chapter 10, Birds*. Berkeley, CA: University of California Press.

Baird, R. W. (2001). Status of harbour seals*, Phoca vitulina*, in Canada. *Canadian Field-Naturalist 115* (4): 663–675.

Baird, R. W. (2005). Sightings of dwarf (*Kogia sima*) and pygmy (*K. breviceps*) sperm whales from the main Hawaiian Islands. *Pacific Science 59* 461–466.

Baird, R. W. (2006). Hawaii's other cetaceans. *Whale and Dolphin Magazine 11* 28–31.

Baird, R. W. (2009a). False killer whale *Pseudorca crassidens*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 405–406). Cambridge, MA: Academic Press.

Baird, R. W. (2009b). *A Review of False Killer Whales in Hawaiian Waters: Biology, Status, and Risk Factors*. Olympia, WA: Marine Mammal Commission.

Baird, R. W. (2013). False Killer Whales Around Kauai and Niihau. *Offshore Neighbors*, 24–25. Retrieved from.

Baird, R. W., J. M. Aschettino, D. J. McSweeney, D. L. Webster, G. S. Schorr, S. Baumann-Pickering, and S. D. Mahaffy. (2010). *Melon-headed Whales in the Hawaiian Archipelago: An Assessment of Population Structure and Long-term Site Fidelity based on Photo-Identification*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Baird, R. W., D. Cholewiak, D. L. Webster, G. S. Schorr, S. D. Mahaffy, C. Curtice, J. Harrison, and S. M. Van Parijs. (2015a). 5. Biologically Important Areas for cetaceans within U.S. waters—Hawaii region. In S. M. Van Parijs, C. Curtice, & M. C. Ferguson (Eds.), *Biologically Important Areas for cetaceans within U.S. waters* (Vol. Aquatic Mammals (Special Issue) 41, pp. 54–64).

Baird, R. W., D. Cholewiak, D. L. Webster, G. S. Schorr, S. D. Mahaffy, C. Curtice, J. Harrison, and S. M. Van Parijs. (2015b). Biologically Important Areas for Cetaceans within U.S. Waters—Hawaii region. In S. M. Van Parijs, C. Curtice, & M. C. Ferguson (Eds.), *Biologically Important Areas for Cetaceans Within U.S. Waters* (Vol. 41, pp. 54–64). Olympia, WA: Cascadia Research Collective.

Baird, R. W., D. Cholewiak, D. L. Webster, G. S. Schorr, S. D. Mahaffy, C. Curtice, J. Harrison, and S. M. van Parijs. (2015c). Biologically important areas for cetaceans within U.S. waters - Hawai'i region. *Aquatic Mammals 41* (1): 54–64.

Baird, R. W., D. Cholewiak, D. L. Webster, G. S. Schorr, S. D. Mahaffy, C. Curtice, J. Harrison, and S. M. Van Parijs. (2015d). Biologically Important Areas for cetaceans within U.S. waters – Hawaii Region. *Aquatic Mammals 41* (1): 54–64. DOI:10.1578/am.41.1.2015.54

Baird, R. W., A. N. Dilley, D. L. Webster, R. Morrissey, B. K. Rone, S. M. Jarvis, S. D. Mahaffy, A. M. Gorgone, and D. J. Moretti. (2015e). *Odontocete Studies on the Pacific Missile Range Facility in February 2014: Satellite-Tagging, Photo Identification, and Passive Acoustic Monitoring*. Pearl Harbor, HI: U.S. Navy Pacific Fleet.

Baird, R. W. and B. Hanson. (1997). Status of the northern fur seal, *Callorhinus ursinus*, in Canada. *Canadian Field-Naturalist 111* 263–269.

Baird, R. W., M. B. Hanson, E. E. Ashe, M. R. Heithaus, and G. J. Marshall. (2003a). *Studies of Foraging in "Southern Resident" Killer Whales during July 2002: Dive Depths, Bursts in Speed, and the Use of a "Crittercam" System for Examining Sub-surface Behavior*. Seattle, WA: U. S. Department of Commerce, National Marine Fisheries Service, National Marine Mammal Laboratory.

Baird, R. W., M. B. Hanson, G. S. Schorr, D. L. Webster, D. J. McSweeney, A. M. Gorgone, S. D. Mahaffy, D. M. Holzer, E. M. Oleson, and R. D. Andrews. (2012). Range and primary habitats of Hawaiian insular false killer whales: Informing determination of critical habitat. *Endangered Species Research 18* (1): 47–61. DOI:10.3354/esr00435

Baird, R. W., J. James, and C. Mata. (2022). Two Gray Whale (*Eschrichtius robustus*) Sightings off Hawaiʻi Island: the First Records for the Central Tropical Pacific. *Aquatic Mammals In Press*.

Baird, R. W., S. D. Mahaffy, A. M. Gorgone, T. Cullins, D. J. McSweeney, E. M. Oleson, A. L. Bradford, J. Barlow, and D. L. Webster. (2015f). False killer whales and fisheries interactions in Hawaiian waters: Evidence for sex bias and variation among populations and social groups. *Marine Mammal Science 31* (2): 579–590. DOI:10.1111/mms.12177

Baird, R. W., D. J. McSweeney, G. S. Schorr, S. D. Mahaffy, D. L. Webster, J. Barlow, M. B. Hanson, J. P. Turner, and R. D. Andrews. (2009). Studies of beaked whales in Hawaii: Population size, movements, trophic ecology, social organization, and behaviour. In S. J. Dolman, C. D. MacLeod, & P. G. H. Evans (Eds.), *Beaked Whale Research* (pp. 23–25). San Sebastián, Spain: European Cetacean Society.

Baird, R. W., D. J. McSweeney, D. L. Webster, A. M. Gorgone, and A. D. Ligon. (2003b). *Studies of Odontocete Population Structure in Hawaiian Waters: Results of a Survey Through the Main Hawaiian Islands in May and June 2003*. Seattle, WA: National Oceanic and Atmospheric Administration.

Baird, R. W., E. M. Oleson, J. Barlow, A. D. Ligon, A. M. Gorgone, and S. D. Mahaffy. (2013a). Evidence of an Island-Associated Population of False Killer Whales (*Pseudorca crassidens*) in the Northwestern Hawaiian Islands. *Pacific Science 67* (4): 513–521. DOI:10.2984/67.4.2

Baird, R. W., G. S. Schorr, D. L. Webster, S. D. Mahaffy, D. J. McSweeney, M. B. Hanson, and R. D. Andrews. (2011). Open-ocean movements of a satellite-tagged Blainville's beaked whale (*Mesoplodon densirostris*): Evidence for an offshore population in Hawaii? *Aquatic Mammals 37* (4): 506–511.

Baird, R. W., G. S. Schorr, D. L. Webster, D. J. McSweeney, and S. D. Mahaffy. (2006). *Studies of beaked whale diving behavior and odontocete stock structure in Hawaii in March/April 2006* (Report prepared under Contract No. AB133F-06-CN-0053). Olympia, WA: Cascadia Research Collective.

Baird, R. W., D. L. Webster , J. M. Aschettino, G. S. Schorr, and D. J. McSweeney. (2013b). Odontocete cetaceans around the main Hawaiian islands: Habitat use and relative abundance from small-boat sighting surveys. *Aquatic Mammals 39* (3): 253–269. DOI:10.1578/am.39.3.2013.253

Baird, R. W., D. L. Webster, J. M. Aschettino, G. S. Schorr, and D. J. McSweeney. (2013c). Odontocete Cetaceans Around the Main Hawaiian Islands: Habitat Use and Relative Abundance from Small-Boat Sighting Surveys. *Aquatic Mammals 39* (3): 253–269. DOI:10.1578/am.39.3.2013.253

Baird, R. W., D. L. Webster, S. D. Mahaffy, D. J. McSweeney, G. S. Schorr, and A. D. Ligon. (2008). Site fidelity and association patterns in a deep-water dolphin: Rough-toothed dolphins (*Steno bredanensis*) in the Hawaiian Archipelago. *Marine Mammal Science 24* (3): 535–553. DOI:10.1111/j.1748-7692.2008.00201

Baird, R. W., D. L. Webster, S. D. Mahaffy, G. S. Schorr, J. M. Aschettino, and A. M. Gorgone. (2013d). *Movements and Spatial Use of Odontocetes in the western Main Hawaiian Islands: Results of a Three-year Study off Oʻahu and Kauaʻi* (Final report under Grant No. N00244-10-1-0048 from the Naval Postgraduate School).

Baker, E. K., K. A. Puglise, and P. T. Harris. (2016a). *Mesophotic coral ecosystems — A lifeboat for coral reefs?* Arendal, Norway and Nairobi, Kenya: The United Nations Environment Programme and Global Resources Information Database.

Baker, J., M. Baumgartner, E. A. Becker, P. Boveng, D. Dick, J. Fiechter, J. Forcada, K. A. Forney, R. Griffis, J. Hare, A. Hobday, D. Howell, K. Laidre, N. Mantua, L. Quakenbush, J. Santora, P. Spencer, C. Stock, K. Stafford, W. Sydeman, K. Van Houtan, and R. Waples. (2016b). *Report of a Workshop on Best Approaches and Needs for Projecting Marine Mammal Distributions in a Changing Climate*. Santa Cruz, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Baker, J. D., A. L. Harting, and T. C. Johanos. (2006). Use of discovery curves to assess abundance of Hawiian monk seals. *Marine Mammal Science 22* (4): 847–861.

Baker, J. D., A. L. Harting, T. C. Johanos, and C. L. Littnan. (2016c). Estimating Hawaiian monk seal range-wide abundance and associated uncertainty. *Endangered Species Research 31* 317–324. DOI:10.3354/esr00782

Baker, J. D., A. L. Harting, T. A. Wurth, and T. C. Johanos. (2011). Dramatic shifts in Hawaiian monk seal distribution predicted from divergent regional trends. *Marine Mammal Science 27* (1): 78–93. DOI:10.1111/j.1748-7692.2010.00395

Baker, J. D. and T. C. Johanos. (2004). Abundance of the Hawaiian monk seal in the main Hawaiian Islands. *Biological Conservation 116* (1): 103–110. DOI:10.1016/s0006-3207(03)00181-2

Baker, P. F. and J. E. Morhardt. (2001). Survival of Chinook salmon smolts in the Sacramento-San Joaquin delta and Pacific Ocean. *Fish Bulletin 179 2* 163–182.

Balazs, G. H. (1980). *Synopsis of Biological Data on the Green Turtle in the Hawaiian Islands*. National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Balazs, G. H. (1986). Fibropapillomas in Hawaiian green turtles. *Marine Turtle Newsletter 39* 1–3.

Balazs, G. H. and M. Chaloupka. (2006). Recovery trend over 32 years at the Hawaiian green turtle rookery of French Frigate Shoals. *Atoll Research Bulletin 543* 147–158.

Balazs, G. H., K. S. Van Houtan, S. A. Hargrove, S. M. Brunson, and S. K. K. Murakawa. (2015). A review of the demographic features of Hawaiian green turtles (Chelonia mydas). *Chelonian Conservation and Biology 14* (2): 119–129.

Baldwin, R., M. Gallagher, and K. Van Waerebeek. (1999). A review of cetaceans from waters off the Arabian Peninsula. In M. Fisher, S. A. Ghazanfur, & J. A. Soalton (Eds.), *The Natural History of Oman: A Festschrift for Michael Gallagher* (pp. 161–189). SV Kerkwerve, The Netherlands: Backhuys Publishers.

Barlow, J. (1988). Harbor Porpoise, *Phocoena phocoena*, Abundances Estimation for California, Oregon, and Washington: I. Ship Surveys. *Fishery Bulletin 86* (3): 417–432.

Barlow, J. (1994a). Abundance of large whales in California coastal waters: A comparison of ship surveys in 1979–1980 and in 1991. *Report of the International Whaling Commision 44* 399–406.

Barlow, J. (1994b). Abundance of large whales in California coastal waters: A comparison of ship surveys in 1979–1980 and in 1991. *Report of the International Whaling Commission 44* 399–406.

Barlow, J. (1995). The abundance of cetaceans in California waters. Part I: Ship surveys in summer and fall of 1991. *Fishery Bulletin 93* 1–14.

Barlow, J. (1997). *Preliminary Estimates of Cetacean Abundance off California, Oregon and Washington based on a 1996 Ship Survey and Comparisons of Passing and Closing Modes*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Barlow, J. (2006). Cetacean abundance in Hawaiian waters estimated from a Summer–Fall survey in 2002. *Marine Mammal Science 22* (2): 446–464. DOI:10.1111/j.1748-7692.2006.00032.x

Barlow, J. (2010). *Cetacean Abundance in the California Current Estimated from a 2008 Ship-Based Line-Transect Survey* (NOAA Technical Memorandum NMFS-SWFSC-456). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Barlow, J. (2016). *Cetacean Abundance in the California Current Estimated from Ship-based Line-transect Surveys in 1991–2014.* (NOAA Administrative Report NMFS-SWFSC-LJ-1601). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Barlow, J., J. Calambokidis, E. A. Falcone, C. S. Baker, A. M. Burdin, P. J. Clapham, J. K. B. Ford, C. M. Gabriele, R. LeDuc, D. K. Mattila, T. J. Quinn, II, L. Rojas-Bracho, J. M. Straley, B. L. Taylor, J. Urbán R, P. Wade, D. Weller, B. H. Witteveen, and M. Yamaguchi. (2011). Humpback whale abundance in the North Pacific estimated by photographic capture-recapture with bias correction from simulation studies. *Marine Mammal Science 27* (4): 793–818. DOI:10.1111/j.1748-7692.2010.00444

Barlow, J. and G. A. Cameron. (2003). Field experiments show that acoustic pingers reduce marine mammal bycatch in the California drift gill net fishery. *Marine Mammal Science 19* (2): 265–283.

Barlow, J., M. Ferguson, E. Becker, J. Redfern, K. Forney, I. Vilchis, P. Fiedler, T. Gerrodette, and L. Ballance. (2009a). *Predictive Modeling of Cetacean Densities in the Eastern Pacific Ocean* (NOAA Technical Memorandum NMFS-SWFSC-444). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Barlow, J., M. C. Ferguson, E. A. Becker, J. V. Redfern, K. A. Forney, I. L. Vilchis, P. C. Fiedler, T. Gerrodette, and L. T. Ballance. (2009b). *Predictive Modeling of Cetacean Densities in the Eastern Pacific Ocean* (NOAA Technical Memorandum NMFS-SWFSC-444). La Jolla, CA: Southwest Fisheries Science Center.

Barlow, J., M. C. Ferguson, W. F. Perrin, L. Ballance, T. Gerrodette, G. Joyce, C. D. MacLeod, K. Mullin, D. L. Palka, and G. Waring. (2006). Abundance and densities of beaked and bottlenose whales (family Ziphiidae). *Journal of Cetacean Research and Management 7* (3): 263–270.

Barlow, J. and K. A. Forney. (2007). Abundance and population density of cetaceans in the California Current ecosystem. *Fishery Bulletin 105* 509–526.

Barlow, J., S. Fregosi, L. Thomas, D. Harris, and E. T. Griffiths. (2021). Acoustic detection range and population density of Cuvier's beaked whales estimated from near-surface hydrophones. *The Journal of the Acoustical Society of America 149* (1). DOI:10.1121/10.0002881

Barlow, J. and R. Gisiner. (2006). Mitigating, monitoring and assessing the effects of anthropogenic sound on beaked whales. *Journal of Cetacean Research and Management 7* (3): 239–249.

Barlow, J., S. Rankin, A. Jackson, and A. Henry. (2008). *Marine Mammal Data Collected During the Pacific Islands Cetacean and Ecosystem Assessment Survey (PICEAS) Conducted Aboard the NOAA Ship McArthur II, July–November 2005*. NOAA.

Barlow, J., S. Rankin, E. Zele, and J. Appler. (2004). *Marine Mammal Data Collected During the Hawaiian Islands Cetacean and Ecosystem Assessment Survey (HICEAS) Conducted Aboard the NOAA Ships McArthur and David Starr Jordan, July–December 2002*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Bassett, C., B. Polagye, M. Holt, and J. Thomson. (2012). A vessel noise budget for Admiralty Inlet, Puget Sound, Washington (USA). *The Journal of the Acoustical Society of America 132* (6): 3706–3719.

Bassett, C., J. Thomson, and B. Polagye. (2010). *Characteristics of Underwater Ambient Noise at a Proposed Tidal Energy Site in Puget Sound*. Seattle, WA: Northwest National Marine Renewable Energy Center.

Baum, J., E. Medina, J. A. Musick, and M. Smale. (2015). *Carcharhinus longimanus. The International Union for Conservation of Nature Red List of Threatened Species 2015: e.T39374A85699641*. Retrieved from http://www.iucnredlist.org/details/39374/0.

Baum, J. K., R. A. Myers, D. G. Kehler, B. Worm, S. J. Harley, and P. A. Doherty. (2003a). Collapse and conservation of shark populations in the northwest Atlantic. *Science 299* (5605): 389–392.

Baum, J. K., R. A. Myers, D. G. Kehler, B. Worm, S. J. Harley, and P. A. Doherty. (2003b). Collapse and conservation of shark populations in the northwest Atlantic. *Science 299* 389-392.

Baumann-Pickering, S., L. K. Baldwin, A. E. Simonis, M. A. Roche, M. L. Melcon, J. A. Hildebrand, E. M. Oleson, R. W. Baird, G. S. Schorr, D. L. Webster, and D. J. McSweeney. (2010). *Characterization of Marine Mammal Recordings from the Hawaii Range Complex*. Monterey, CA: Naval Postgraduate School.

Baumann-Pickering, S., M. A. Roch, R. L. Brownell, Jr., A. E. Simonis, M. A. McDonald, A. Solsona-Berga, E. M. Oleson, S. M. Wiggins, and J. A. Hildebrand. (2014). Spatio-temporal patterns of beaked whale echolocation signals in the north Pacific. *PLoS ONE 9* (1): e86072. DOI:10.1371/journal.pone.0086072

Baumann-Pickering, S., J. S. Trickey, M. A. Roch, and S. M. Wiggins. (2015). *Relative Densities and Spatial Distribution of Beaked Whales in Southern California*. La Jolla, CA: Scripps Institution of Oceanography – University of California San Diego and San Diego State University.

Baumgartner, M. F. (1997). The distribution of Risso's dolphin (*Grampus griseus*) with respect to the physiography of the northern Gulf of Mexico. *Marine Mammal Science 13* (4): 614–638.

Beamish, R. J., G. A. McFarlane, and J. R. King. (2005). Migratory patterns of pelagic fishes and possible linkages between open ocean and coastal ecosystems off the Pacific coast of north America. *Deep-Sea Research Part II-Topical Studies in Oceanography 52* (5–6): 739–755. DOI:10.1016/j.dsr2.2004.12.016

Beard, A., F. Hailer, R. Medeiros Mirra, and R. Thomas. (2022). Increasing use of artificial nest chambers by seasonally segregated populations of band-rumped storm petrels *hydrobates castro* at St Helena, South Atlantic. *Marine Ornithology* (51): 85–96.

Bearzi, M. (2005a). Aspects of the ecology and behavior of bottlenose dolphins (*Tursiops truncatus*) in Santa Monica Bay, California. *Journal of Cetacean Research and Management 7* (1): 75–83.

Bearzi, M. (2005b). Habitat partitioning by three species of dolphins in Santa Monica Bay, California. *Bulletin of the Southern California Academy of Sciences 104* (3): 113–124.

Bearzi, M., C. A. Saylan, and J. Feenstra. (2009a). Seabird observations during cetacean surveys in Santa Monica Bay, California. *Bulletin of Southern California Academy of Sciences 108* (2): 63–69.

Bearzi, M., C. A. Saylan, and A. Hwang. (2009b). Ecology and comparison of coastal and offshore bottlenose dolphins (*Tursiops truncatus*) in California. *Marine and Freshwater Research 60* 584–593.

Beason, R. (2004). *What Can Birds Hear?* Lincoln, NE: University of Nebraska.

Becker, E., K. Forney, D. Miller, P. Fiedler, J. Barlow, and J. Moore. (2020). *Habitat-based density estimates for cetaceans in the California Current Ecosystem based on 1991-2018 survey data* (Technical Memorandum NMFS-SWFSC-638). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Becker, E. A., K. A. Forney, M. C. Ferguson, J. Barlow, and J. V. Redfern. (2012a). *Predictive Modeling of Cetacean Densities in the California Current Ecosystem based on Summer/Fall Ship Surveys in 1991–2008* (NOAA Technical Memorandum NMFS-SWFSC-499). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Becker, E. A., K. A. Forney, M. C. Ferguson, D. G. Foley, R. C. Smith, J. Barlow, and J. V. Redfern. (2010). Comparing California Current cetacean–habitat models developed using in situ and remotely sensed sea surface temperature data. *Marine Ecology Progress Series 413* 163–183. DOI:10.3354/meps08696

Becker, E. A., K. A. Forney, P. C. Fiedler, J. Barlow, S. J. Chivers, C. A. Edwards, A. M. Moore, and J. V. Redfern. (2016). Moving Towards Dynamic Ocean Management: How Well Do Modeled Ocean Products Predict Species Distributions? *Remote Sensing 8* (2): 149. DOI:10.3390/rs8020149

Becker, E. A., K. A. Forney, D. G. Foley, and J. Barlow. (2012b). *Density and Spatial Distribution Patterns of Cetaceans in the Central North Pacific based on Habitat Models* (NOAA Technical Memorandum NMFS-SWFSC-490). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Becker, E. A., K. A. Forney, D. G. Foley, R. C. Smith, T. J. Moore, and J. Barlow. (2014). Predicting seasonal density patterns of California cetaceans based on habitat models. *Endangered Species Research 23* (1): 1–22. DOI:10.3354/esr00548

Becker, E. A., K. A. Forney, D. L. Miller, J. Barlow, L. Rojas-Bracho, J. Urbán R., and J. E. Moore. (2022a). Dynamic habitat models reflect interannual movement of cetaceans within the California Current ecosystem. *Frontiers in Marine Science 9*. DOI:10.3389/fmars.2022.829523

Becker, E. A., K. A. Forney, E. M. Oleson, A. L. Bradford, R. Hoopes, J. E. Moore, and J. Barlow. (2022b). *Abundance, distribution, and seasonality of cetaceans within the U.S. Exclusive Economic Zone around the Hawaiian Archipelago based on species distribution models*. Honolulu, HI: National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Becker, E. A., K. A. Forney, E. M. Oleson, A. L. Bradford, J. E. Moore, and J. Barlow. (2021). *Habitat-based density estimates for cetaceans within the waters of the U.S. Exclusive Economic Zone around the Hawaiian Archipelago* (NOAA Technical Memorandum NMFS-PIFSC-116). Honolulu, HI:

National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Becker, E. A., K. A. Forney, J. V. Redfern, J. Barlow, M. G. Jacox, J. J. Roberts, and D. M. Palacios. (2018). Predicting cetacean abundance and distribution in a changing climate. *Diversity and Distributions 2018* 1–18. DOI:10.1111/ddi.12867

Becker, E. A., K. A. Forney, B. J. Thayre, A. J. Debich, G. S. Campbell, K. Whitaker, A. B. Douglas, A. Gilles, R. Hoopes, and J. A. Hildebrand. (2017). Habitat-Based Density Models for Three Cetacean Species off Southern California Illustrate Pronounced Seasonal Differences. *Frontiers in Marine Science 4* (121): 1–14. DOI:10.3389/fmars.2017.00121

Becker, S. L., R. E. Brainard, and K. S. Van Houtan. (2019). Densities and drivers of sea turtle populations across Pacific coral reef ecosystems. *PLoS ONE 14* (4): e0214972.

Becking, L. E., T. C. J. M. van Bussel, A. O. Debrot, and M. J. A. Christianen. (2014). First record of a Caribbean green turtle (*Chelonia mydas*) grazing on invasive seagrass (*Halophila stipulacea*). *Caribbean Journal of Science 48* (2–3): 162–163.

Bednaršek, N., B. R. Carter, R. M. McCabe, R. A. Feely, E. Howard, F. P. Chavez, M. Elliott, J. L. Fisher, J. Jahncke, and Z. Siegrist. (2022). Pelagic calcifiers face increased mortality and habitat loss with warming and ocean acidification. *Ecological Applications 32* (7).

Beeson, M. (1998). and seabird mortality for two California gillnet fisheries: 1990–1995. *Fishery Bulletin 96* 271-284.

Behrens, M. D. and K. D. Lafferty. (2005). *Size frequency measures of white abalone with implications for conservation* (*In* Proceedings of the Sixth California Islands Symposium). Ventura, California: Institute for Wildlife Studies.

Belcher, R. I. and T. E. Lee, Jr. (2002). *Arctocephalus townsendi*. *Mammalian Species 700* 1–5.

Bellinger, M. R., M. A. Banks, S. J. Bates, E. D. Crandall, J. C. Garza, G. Sylvia, and P. W. Lawson. (2015). Geo-Referenced, Abundance Calibrated Ocean Distribution of Chinook Salmon (Oncorhynchus tshawytscha) Stocks across the West Coast of North America. *PLoS ONE 10* (7): e0131276. DOI:10.1371/journal.pone.0131276

Benson, S. (2022, 1/4/2022 and 3/30/2022). Personal communication between Scott Benson (NMFS) and Mike Zickel (ManTech International) on the occurrence of leatherback sea turtles in California waters and Baja California, Mexico.

Benson, S. R., K. A. Forney, J. E. Moore, E. L. LaCasella, J. T. Harvey, and J. V. Carretta. (2020). A long-term decline in the abundance of endangered leatherback turtles, *Dermochelys coriacea*, at a foraging ground in the California Current Ecosystem. *Global Ecology and Conservation, 24*. Retrieved from https://doi.org/10.1016/j.gecco.2020.e01371.

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

Bergmuller, R., R. A. Johnstone, A. F. Russell, and R. Bshary. (2007). Integrating cooperative breeding into theoretical concepts of cooperation. *Behavioural Processes 2* 67–72.

Bergstad, O. A., T. Falkenhaug, O. S. Astthorsson, I. Byrkjedal, A. V. Gebruk, U. Piatkowski, I. G. Priede, R. S. Santos, M. Vecchione, P. Lorance, and J. D. M. Gordon. (2008). Towards improved understanding of the diversity and abundance patterns of the mid-ocean ridge macro- and megafauna. *Deep-Sea Research II 55* (1–2): 1–5. DOI:10.1016/j.dsr2.2007.10.001

Bermudez, J. R., U. Riebesell, A. Larsen, and M. Winder. (2016). Ocean acidification reduces transfer of essential biomolecules in a natural plankton community. *Scientific Reports 6* 1–8. DOI:10.1038/srep27749

Bernard, H. J. and S. B. Reilly. (1999). Pilot whales, *Globicephala* Lesson, 1828. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 6, pp. 245–280). San Diego, CA: Academic Press.

Bernardino, A. F. and C. R. Smith. (2010). Community structure of infaunal macrobenthos around vestimentiferan thickets at the San Clemente cold seep, NE Pacific. *Marine Ecology* 608–621. DOI:10.1111/j.1439-0485.2010.00389.x

Bernhardt, P. (1979). *A Key to the Nemertea from the Intertidal Zone of the Coast of California*. Southern California Association of Marine Invertebrate Taxonomists. Long Beach, CA.

Bernstein, B., K. Merkel, B. Chensey, and M. Sutula. (2011). *Recommendations for a Southern California Regional Eelgrass Monitoring Program*. Ojai, San Diego, Long Beach, and Costa Mesa, CA: Southern California Coastal Water Research Project.

Berta, A., J. L. Sumich, and K. M. Kovacs. (2006). *Marine Mammals: Evolutionary Biology* (2nd ed.). Burlington, MA: Elsevier.

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

Bessell-Browne, P., A. P. Negri, R. Fisher, P. L. Clode, and R. Jones. (2017). Cumulative impacts: Thermally bleached corals have reduced capacity to clear deposited sediment. *Scientific Reports 7* (2716): 14. DOI:10.1038/s41598-017-02810-0

Best, P. B. (1996). Evidence of migration by Bryde's whales from the offshore population in the southeast Atlantic. *Reports of the International Whaling Commission 46* 315–322.

Bettridge, S., C. S. Baker, J. Barlow, P. J. Clapham, M. Ford, D. Gouveia, D. K. Mattila, I. Pace, R. M. , P. E. Rosel, G. K. Silber, and W. P. R. (2015a). *Status Review of the Humpback Whale (Megaptera novaeangliae) Under the Endangered Species Act*. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Bettridge, S., C. S. Baker, J. Barlow, P. J. Clapham, M. Ford, D. Gouveia, D. K. Mattila, R. M. Pace, III, P. E. Rosel, G. K. Silber, and P. R. Wade. (2015b). *Status Review of the Humpback Whale (Megaptera novaeangliae) under the Endangered Species Act* (NOAA Technical Memorandum NMFS-SWFSC-540). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Beuter, K. J., R. Weiss, and B. Frankfurt. (1986). *Properties of the auditory system in birds and the effectiveness of acoustic scaring signals*. Presented at the Bird Strike Committee Europe, 18th Meeting Part I, 26–30 May 1986. Copenhagen, Denmark.

Biggs, D. C., R. R. Leben, and J. G. Ortega-Ortiz. (2000). Ship and satellite studies of mesoscale circulation and sperm whale habitats in the northeast Gulf of Mexico during GulfCetII. *Gulf of Mexico Science 18* (1): 15–22.

Biggs, T. W. and H. D'Anna. (2012). Rapid increase in copper concentrations in a new marina, San Diego Bay. *Marine Pollution Bulletin 64* (3): 627–635.

Bignami, S., I. C. Enochs, D. P. Manzello, S. Sponaugle, and R. K. Cowen. (2013a). Ocean acidification alters the otoliths of a pantropical fish species with implications for sensory function. *Proceedings of the National Academy of Sciences of the United States of America 110* (18): 7366–7370. DOI:10.1073/pnas.1301365110

Bignami, S., S. Sponaugle, and R. K. Cowen. (2013b). Response to ocean acidification in larvae of a large tropical marine fish, *Rachycentron canadum*. *Global Change Biology 19* (4): 996–1006. DOI:10.1111/gcb.12133

Biodiversity Research Institute. (2015). *Mid-Atlantic Wildlife Studies: Distribution and Abundance of Wildlife along the Eastern Seaboard 2012–2014*. Portland, ME.

Birding Hawaii. (2004). *Annotated list of Hawaii's breeding birds*. Retrieved from http://www.birdinghawaii.co.uk/Annotatedlist2.htm.

BirdLife International. (2008). *Avian diseases are spreading to impact hitherto unaffected populations*. Retrieved May 11, 2017, from http://datazone.birdlife.org/sowb/casestudy/avian-diseases-are-spreading-to-impact-hitherto-unaffected-populations.

BirdLife International. (2009). *Sterna antillarum*. *IUCN 2010. IUCN Red List of Threatened Species. Version 2010.3.* Retrieved from www.iucnredlist.org.

BirdLife International. (2012). *Spotlight on seabirds*. Retrieved May 11, 2017, from http://datazone.birdlife.org/sowb/spotseabirds.

BirdLife International and National Audubon Society. (2015). *The Messengers: What Birds Tell Us About Threats from Climate Change and Solutions for Nature and People*. Cambridge, United Kingdom and New York, NY: BirdLife International and National Audubon Society.

Bjorge, A. (2002). How persistent are marine mammal habitats in an ocean of variability? In P. G. H. Evans & A. Raga (Eds.), *Marine Mammals: Biology and Conservation* (pp. 63–91). Norwell, MA: Kluwer Academic/Plenum Publishers.

Bjorndal, K. A. (1997). Foraging ecology and nutrition of sea turtles. In P. L. Lutz & J. A. Musick (Eds.), *The Biology of Sea Turtles* (pp. 199–231). Boca Raton, FL: CRC Press.

Bjorndal, K. A. (2003). Roles of loggerhead sea turtles in marine ecosystems. In A. B. Bolten & B. E. Witherington (Eds.), *Loggerhead Sea Turtles* (pp. 235–254). Washington, DC: Smithsonian Books.

Bjorndal, K. A. and A. B. Bolten. (1988). Growth rates of immature green turtles, *Chelonia mydas*, on feeding grounds in the southern Bahamas. *Copeia 1988* (3): 555–564.

Bjorndal, K. A., A. B. Bolten, and C. Lagueux. (1994). Ingestion of Marine Debris by Juvenile Sea Turtles in Coastal Florida Habitats. *Marine Pollution Bulletin 28* (3): 154–158. DOI:0025-326X/94

Blanchard, J. L. and C. Novaglio. (2024). *Climate change risks to marine ecosystems and fisheries – Projections to 2100 from the Fisheries and Marine Ecosystem Model Intercomparison Project* (FAO Fisheries and Aquaculture Technical Paper, No. 707). Rome, Italy: Food and Agriculture Organization of the United Nations.

Blechschmidt, J., M. J. Wittmann, and C. Blüml. (2020). Climate change and green sea turtle sex ratio—preventing possible extinction. *Genes 11* (5): 588.

Bloodworth, B. E. and D. K. Odell. (2008). *Kogia breviceps*. *Mammalian Species 819* 1–12. DOI:10.1644/819.1

Blott, S. J. and K. Pye. (2012). Particle size scales and classification of sediment types based on particle size distributions: Review and recommended procedures. *Sedimentology 59* (7): 2071–2096. DOI:10.1111/j.1365-3091.2012.01335.x

Bodkin, J. L. (2015). Chapter 3: Historic and Contemporary Status of Sea Otters in the North Pacific *Sea Otter Conservation*. Anchorage, AK: U.S. Geological Survey, Alaska Science Center.

Bodkin, J. L., G. G. Esslinger, and D. H. Monson. (2004). Foraging depths of sea otters and implications to coastal marine communities. *Marine Mammal Science 20* (2): 305–321.

Boland, G. S., P. J. Etnoyer, C. R. Fisher, and E. L. Hickerson. (2016). State of Deep-Sea Coral and Sponge Ecosystems in the Gulf of Mexico Region: Texas to the Florida Straits. In NOAA (Ed.), *The State of Deep-Sea Coral and Sponge Ecosystems in the United States* (Vol. NOAA Technical Memorandum NMFS-OHC-3, pp. 11–11–11–59). Silver Spring, MD: National Oceanic and Atmospheric Administration.

Bond, M. H. (2006). *Importance of estuarine rearing to central California steelhead (Oncorhynchus mykiss) growth and marine survival.* (M.A.). UC Santa Cruz.

Bond, M. H., S. A. Hayes, C. V. Hanson, and R. B. MacFarlane. (2008). Marine survival of steelhead (*Oncorhynchus mykiss*) enhanced by a seasonally closed estuary. *Canadian Journal of Fisheries and Aquatic Sciences 65* (10): 2242–2252.

Bond, N. A., M. F. Cronin, H. Freeland, and N. Mantua. (2015). Causes and impacts of the 2014 warm anomaly in the northeast Pacific. *Geophysical Research Letters 42* (9): 3414–3420. DOI:10.1002/2015gl063306

Boudreau, B. P. (1998). Mean mixed depth of sediments: The wherefore and the why. *Limnology and Oceanography 43* (3): 524–526.

Bouillon, S., V. H. Rivera-Monroy, R. R. Twilley, and J. G. Kairo. (2009). Mangroves. In D. D. A. Laffoley & G. Grimsditch (Eds.), *The Management of Natural Coastal Carbon Sinks*. Gland, Switzerland: International Union for the Conservation of Nature.

Bowen, B. W., F. A. Abreu-Grobois, G. H. Balazs, N. Kamezaki, C. J. Limpus, and R. J. Ferl. (1995). Trans-Pacific migrations of the loggerhead turtle (*Caretta caretta*) demonstrated with mitochondrial DNA markers. *Proceedings of the National Academy of Sciences of the United States of America 92* 3731–3734.

Boyd, C., J. Barlow, E. A. Becker, K. A. Forney, T. Gerrodette, J. E. Moore, and A. E. Punt. (2017). Estimation of population size and trends for highly mobile species with dynamic spatial distributions. *Diversity and Distributions 24* 1–12.

Boyer, J. N., J. W. Fourqurean, and R. D. Jones. (1999). Seasonal and long-term trends in the water quality of Florida Bay (1989–1997). *Estuaries 22* (2): 417–430.

Bradford, A., D. Weller, Y. Ivashchenko, A. Burdin, and R. Brownell, Jr. (2009). Anthropogenic Scarring of Western Gray Whales (*Eschrichtius robustus*). *Marine Mammal Science 25* (1): 161–175. DOI:10.1111/j.1748-7692.2008.00253

Bradford, A. L. (2018a). *Injury Determinations for Marine Mammals Observed Interacting with Hawaii and America Samoa Longline Fisheries During 2017*. Silver Spring, MD: National Marine Fisheries Service.

Bradford, A. L. (2018b). *Injury Determinations for Marine Mammals Observed Interacting with Hawaii and American Samoa Longline Fisheries During 2015–2016* (NOAA Technical Memorandum NMFS-PIFSC-70). Honolulu, HI: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Bradford, A. L. (2021). *Injury Determinations for Marine Mammals Observed Interacting with Hawaii and American Samoa Longline Fisheries During 2019*. Honolulu, HI: Pacific Islands Fisheries Science Center, National Marine Fisheries Service.

Bradford, A. L., R. W. Baird, S. D. Mahaffy, A. M. Gorgone, D. J. McSweeney, T. Cullins, D. L. Webster, and A. N. Zerbini. (2018). Abundance estimates for management of endangered false killer whales in the main Hawaiian Islands. *Endangered Species Research 36* 297–313.

Bradford, A. L., E. A. Becker, E. M. Oleson, K. A. Forney, J. E. Moore, and J. Barlow. (2020). *Abundance Estimates of False Killer Whales in Hawaiian Waters and the Broader Central Pacific* (NOAA Technical Memorandum NMFS-PIFSC-104). Honolulu, HI: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Bradford, A. L. and K. A. Forney. (2016). *Injury Determinations for Marine Mammals Observed Interacting with Hawaii and American Samoa Longline Fisheries During 2009–2013*: NOAA.

Bradford, A. L. and K. A. Forney. (2017). *Injury Determinations for Marine Mammals Observed Interacting with Hawaii and American Samoa Longline Fisheries During 2010–2014* (NOAA Technical Memorandum NMFS-PIFSC-62). Honolulu, HI: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Bradford, A. L., K. A. Forney, E. A. Oleson, and J. Barlow. (2013). *Line-Transect Abundance Estimates of Cetaceans in the Hawaiian EEZ*. Pacific Islands Fisheries Science Center.

Bradford, A. L., K. A. Forney, E. M. Oleson, and J. Barlow. (2012). *Line-transect Abundance Estimates of False Killer Whales (Pseudorca crassidens) in the Pelagic Region of the Hawaiian Exclusive Economic Zone and in the Insular Waters of the Northwestern Hawaiian Islands*. Honolulu, HI: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Bradford, A. L., K. A. Forney, E. M. Oleson, and J. Barlow. (2014). Accounting for subgroup structure in line-transect abundance estimates of false killer whales (*Pseudorca crassidens*) in Hawaiian waters. *PLoS ONE 9* (2): e90464. DOI:10.1371/journal.pone.0090464

Bradford, A. L., K. A. Forney, E. M. Oleson, and J. Barlow. (2017). Abundance estimates of cetaceans from a line-transect survey within the U.S. Hawaiian Islands Exclusive Economic Zone. *Fishery Bulletin 115* (2): 129–142. DOI:10.7755/fb.115.2.1

Bradford, A. L. and E. Lyman. (2015a). *Injury Determinations for Humpback Whales and Other Cetaceans Reported to NOAA Response Networks in the Hawaiian Islands During 2007–2012* (NOAA Technical Memorandum NMFS-PIFSC-45). Honolulu, HI: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Bradford, A. L. and E. Lyman. (2015b). *Injury Determinations for Humpback Whales and Other Cetaceans Reported to NOAA Response Networks in the Hawaiian Islands During 2007–2012*. Honolulu, HI: U.S. Department of Commerce.

Bradford, A. L. and E. Lyman. (2015c). *Injury Determinations for Humpback Whales and Other Cetaceans Reported to NOAA Response Networks in the Hawaiian Islands During 2007 – 2012*. National Oceanographic and Atmospheric Administration Technical Memorandum.

Bradford, A. L. and E. G. Lyman. (2018). *Injury determinations for humpback whales and other cetaceans reported to NOAA response networks in the Hawaiian Islands during 2013–2016* (NOAA Technical Memorandum NMFS-PIFSC-75). Honolulu, HI: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Bradford, A. L., E. A. Oleson, R. W. Baird, C. H. Boggs, K. A. Forney, and N. C. Young. (2015). *Revised Stock Boundaries for False Killer Whales (Psuedorca crassidens) in Hawaiian Waters* (NOAA Technical Memorandum NMFS-PIFSC-47). Honolulu, HI: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Bradford, A. L., E. M. Oleson, K. A. Forney, J. E. Moore, and J. Barlow. (2021). *Line-transect Abundance Estimates of Cetaceans in U.S. Waters around the Hawaiian Islands in 2002, 2010, and 2017* (NOAA Technical Memorandum NMFS-PIFSC-115). Honolulu, HI: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Bradshaw, C. J. A., K. Evans, and M. A. Hindell. (2006). Mass cetacean strandings—A plea for empiricism. *Conservation Biology 20* (2): 584–586. DOI:10.1111/j.1523-1739.2006.00329.x

Brainard, R. E., C. Birkeland, C. M. Eakin, P. McElhany, M. W. Miller, M. Patterson, and G. A. Piniak. (2011). *Status Review Report of 82 Candidate Coral Species Petitioned Under the U.S. Endangered Species Act* (NOAA Technical Memorandum NMFS-PIFSC-27). Honolulu, HI: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Branch, T. A. (2007). Abundance of Antarctic blue whales south of 60°S from three complete circumpolar sets of surveys. *Journal of Cetacean Research and Management 9* (3): 253–262.

Brander, K. (2010). Impacts of climate change on fisheries. *Journal of Marine Systems 79* (3–4): 389–402. DOI:10.1016/j.jmarsys.2008.12.015

Brander, K. M. (2007). Global fish production and climate change. *Proceedings of the National Academy of Sciences of the United States of America 104* (50): 19709–19714. DOI:10.1073/pnas.0702059104

Brandes, P. L. and J. S. McLain. (2000). Juvenile Chinook salmon abundance, distribution, and survival in the Sacramento-San Joaquin Estuary. *Fish Bulletin 179 2*.

Brehmer, P., F. Gerlotto, C. Laurent, P. Cotel, A. Achury, and B. Samb. (2007). Schooling behaviour of small pelagic fish: Phenotypic expression of independent stimuli. *Marine Ecology Progress Series 334* 263–272.

Breitbarth, E., E. P. Achterberg, M. V. Ardelan, A. R. Baker, E. Bucciarelli, F. Chever, P. L. Croot, S. Duggen, M. Gledhill, M. Hassellov, C. Hassler, L. J. Hoffmann, K. A. Hunter, D. A. Hutchins, J. Ingri, T. Jickells, M. C. Lohan, M. C. Nielsdottir, G. Sarthou, V. Schoemann, J. M. Trapp, D. R. Turner, and Y. Ye. (2010). Iron biogeochemistry across marine systems—progress from the past decade. *Biogeosciences 7* 1075–1097.

Bresette, M., D. Singewald, and E. De Maye. (2006). Recruitment of post-pelagic green turtles (*Chelonia mydas*) to nearshore reefs on Florida's east coast. In M. Frick, A. Panagopoulou, A. F. Rees, & K. Williams (Eds.), *Twenty-sixth Annual Symposium on Sea Turtle Biology and Conservation: Book of Abstracts* (pp. 288). Athens, Greece: National Marine Fisheries Service Southeast Fisheries Science Center, International Sea Turtle Society.

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.

Brischoux, F., C. Cotte, H. B. Lillywhite, F. Bailleul, M. Lalire, and P. Gaspar. (2016). Oceanic circulation models help to predict global biogeography of pelagic yellow-bellied sea snake. *Biology Letters 12* (8). DOI:10.1098/rsbl.2016.0436

Briscoe, D. K., C. N. Turner Tomaszewicz, J. A. Seminoff, D. M. Parker, G. H. Balazs, J. J. Polovina, M. Kurita, H. Okamoto, T. Saito, M. R. Rice, and L. B. Crowder. (2021). Dynamic Thermal Corridor May Connect Endangered Loggerhead Sea Turtles Across the Pacific Ocean. *Frontiers in Marine Science 8*. DOI:10.3389/fmars.2021.630590

Browlow, A., J. Onoufriou, A. Bishop, N. Davidson, and D. Thompson. (2016). Corkscrew Seals: Grey Seal (*Halichoerus grypus*) infanticide and cannibalisng may indicate the cause of spiral lacerations in seals. *PLoS ONE 11* (6): e0156464. DOI:10.1371/journal.pone.0156464

Brown, J. W. and J. Harshman. (2008). *Pelecaniformes*. Retrieved from http://tolweb.org/Pelecaniformes/57152/2008.06.27.

Browne, M. A., S. J. Niven, T. S. Galloway, S. J. Rowland, and R. C. Thompson. (2013). Microplastic moves pollutants and additives to worms, reducing functions linked to health and biodiversity. *Current Biology 23* (23): 2388–2392. DOI:10.1016/j.cub.2013.10.012

Brownell, R. L., Jr., K. Ralls, S. Baumann-Pickering, and M. M. Poole. (2009). Behavior of melon-headed whales, *Peponocephala electra*, near oceanic islands. *Marine Mammal Science 25* (3): 639–658. DOI:10.1111/j.1748-7692.2009.00281.x

Brownell, R. L., Jr., W. A. Walker, and K. A. Forney. (1999). Pacific white-sided dolphin, *Lagenorhynchus obliquidens* Gill, 1865. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 6: The second book of dolphins and the porpoises, pp. 57–84). Academic Press.

Bruland, K. W. (1992). Complexation of cadmium by natural organic ligands in the Central North Pacific. *ASLO Limnology and Oceanography 37* (5): 1008–1017.

Brunson, S., A. R. Gaos, I. K. Kelly, K. S. Van Houtan, Y. Swimmer, S. Hargrove, G. H. Balazs, T. M. Work, and T. T. Jones. (2022). Three decades of stranding data reveal insights into endangered hawksbill sea turtles in Hawaiʻi. *Endangered Species Research 47* 109–118. DOI:10.3354/esr01167

Brusca, R. C. and G. J. Brusca. (2003a). Invertebrates *In Invertebrates* (pp. 219–283). Sunderland: Sinauer Associates, Inc.

Brusca, R. C. and G. J. Brusca. (2003b). Phylum Cnidaria. In: *Invertebrates* (pp. 219–283). Sunderland, MA: Sinauer Associates, Inc.

Bryan, T. L. and A. Metaxas. (2007). Predicting suitable habitat for deep-water gorgonian corals on the Atlantic and Pacific Continental Margins of North America. *Marine Ecology Progress Series 330* 113–126.

Bryant, D., L. Burke, J. McManus, and M. Spalding. (1998). *Reefs at Risk: A Map-Based Indicator of Threats to the World's Coral Reefs*. Washington, DC: World Resources Institute.

Budelmann, B. U. (1992a). Hearing in Crustacea. In D. B. Webster, R. R. Fay, & A. N. Popper (Eds.), *Evolutionary Biology of Hearing* (pp. 131–139). New York, NY: Springer-Verlag.

Budelmann, B. U. (1992b). Hearing in nonarthropod invertebrates. In D. B. Webster, R. R. Fay, & A. N. Popper (Eds.), *Evolutionary Biology of Hearing* (pp. 141–155). New York, NY: Springer-Verlag.

Bugoni, L., T. S. Neves, N. O. Leite, Jr., D. Carvalho, G. Sales, R. W. Furness, C. E. Stein, F. V. Peppes, B. B. Giffoni, and D. S. Monteiro. (2008). Potential bycatch of seabirds and turtles in hook-and-line fisheries of the Itaipava Fleet, Brazil. *Fisheries Research 90* 217–224.

Buhl-Mortensen, L., A. Vanreusel, A. J. Gooday, L. A. Levin, I. G. Priede, P. Buhl-Mortensen, H. Gheerardyn, N. J. King, and M. Raes. (2010). Biological structures as a source of habitat heterogeneity and biodiversity on the deep ocean margins. *Marine Ecology 31* (1): 21–50. DOI:10.1111/j.1439-0485.2010.00359

Bureau of Ocean Energy Management. (2018). *State Activities*. Retrieved from https://www.boem.gov/Renewable-Energy-State-Activities/.

Burger, A. E. (2001). Diving depths of shearwaters. *The Auk 118* (3): 755–759.

Burger, A. E. (2002). *Conservation Assessment of Marbled Murrelets in British Columbia, a Review of the Biology, Populations, Habitat Associations and Conservation*. Pacific and Yukon Region, Canada: Canadian Wildlife Service, Environmental Conservation Branch.

Burgner, R. L., J. T. Light, L. Margolis, T. Okazaki, A. Tautz, and S. Ito. (1992). *Distribution and origins of steelhead trout (Oncorhynchus mykiss) in offshore waters of the North Pacific Ocean* (Bulletin Number 51). Vancouver, Canada: International North Pacific Fisheries Commission.

Burkholder, J., D. Eggleston, H. Glasgow, C. Brownie, R. Reed, G. Janowitz, M. Posey, G. Mella, C. Kinder, R. Corbett, D. Toms, T. Alphin, N. Deamer, and J. Springer. (2004). Comparative impacts of two major hurricane seasons on the Neuse River and western Pamlico Sound ecosystems. *Proceedings of the National Academy of Sciences 101* (25): 9291–9296. DOI:10.1073/pnas.0306842101

Burn, D. and A. Doroff. (2005). Decline in sea otter (*Enhydra lutris*) populations along the Alaska Peninsula, 1986–2001. *Fishery Bulletin 103* (2): 270–279.

Burns, J. J. (2008). Harbor seal and spotted seal *Phoca vitulina* and *P. largha*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals (Second Edition)* (pp. 533–542). Academic Press.

Burns, T. J., H. Davidson, and M. W. Kennedy. (2016). Large-scale investment in the excavation and "camouflaging" phases by nesting Leatherback Turtles (*Dermochelys coriacea*). *Canadian Journal of Zoology 94* (6): 443–448.

Busby, P. J., T. C. Wainwright, G. J. Bryant, L. J. Lienheimer, R. S. Waples, F. W. Waknitz, and I. V. Lagomarsino. (1996). *Status Review of West Coast Steelhead from Washington, Idaho, Oregon, and California* (NOAA Technical Memorandum NMFS-NWFSC-27). Seattle, WA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center, Coastal Zone and Estuarine Studies Division.

Buscaino, G., F. Filiciotto, M. Gristina, A. Bellante, G. Buffa, V. Di Stefano, V. Maccarrone, G. Tranchida, C. Buscaino, and S. Mazzola. (2011). Acoustic behavior of the European spiny lobster *Palinurus elephas*. *Marine Ecology Progress Series 441* 177–184. DOI:10.3354/meps09404

Butler, J., A. DeVogelaere, R. Gustafson, C. Mobley, M. Neuman, D. Richards, S. Rumsey, B. Taylor, and G. VanBlaricom. (2009). *Status Review Report for Black Abalone (Haliotis cracherodii Leach, 1814)*. Long Beach, CA: National Marine Fisheries Service, Southwest Region.

Butler, J., M. Neuman, D. Pinkard, R. Kvitek, and G. Cochrane. (2006a). The use of multibeam sonar mapping techniques to refine population estimates of the endangered white abalone (*Haliotis sorenseni*). *Fisheries Bulletin 104* (4): 521–532.

Butler, J., M. Neuman, D. Pinkard, R. Kvitek, and G. Cochrane. (2006b). The use of multibeam sonar mapping techniques to refine population estimates of the endangered white abalone (*Haliotis sorenseni*). *Fishery Bulletin 104* (4): 521–532.

Byrne, R. H. (1996). Specific problems in the measurement and intrepretation of complexation phenomena in seawater. *Pure and Applied Chemistry 68* (8): 1639–1656.

Byrnes, J. E., D. C. Reed, B. J. Cardinale, K. C. Cavanaugh, S. J. Holbrooks, and R. J. Schmitts. (2011). Climate-driven increases in storm frequency simplify kelp forest food webs. *Global Change Biology 17* 2513–2525. DOI:10.1111

Calambokidis, J. and J. Barlow. (2004). Abundance of blue and humpback whales in the eastern North Pacific estimated by capture-recapture and line-transect methods. *Marine Mammal Science 20* (1): 63–85.

Calambokidis, J. and J. Barlow. (2013). *Updated Abundance Estimates of Blue and Humpback Whales off the U.S. West Coast Incorporating Photo-Identifications from 2010 and 2011*.

Calambokidis, J. and J. Barlow. (2020). *Updated abundance estimates for blue and humpback whales along the U.S. West Coast using data through 2018* (NOAA Technical Memorandum NMFS-SWFSC-634). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Calambokidis, J., J. Barlow, J. K. B. Ford, T. E. Chandler, and A. B. Douglas. (2009a). Insights into the population structure of blue whales in the Eastern North Pacific from recent sightings and photographic identification. *Marine Mammal Science 25* (4): 816–832. DOI:10.1111/j.1748-7692.2009.00298

Calambokidis, J., J. A. Fahlbusch, A. R. Szesciorka, B. L. Southall, D. E. Cade, A. S. Friedlaender, and J. A. Goldbogen. (2019). Differential vulnerability to ship strikes between day and night for blue, fin, and humpback whales based on dive and movement data from medium duration archival tags. *Frontiers in Marine Science 6* 11. DOI:10.3389/fmars.2019.00543

Calambokidis, J., E. Falcone, A. Douglas, L. Schlender, and J. Huggins. (2009b). *Photographic Identification of Humpback and Blue Whales off the U.S. West Coast: Results and Updated Abundance Estimates from 2008 Field Season*. La Jolla, CA: Southwest Fisheries Science Center, and Olympia, WA: Cascadia Research Collective.

Calambokidis, J., E. A. Falcone, T. J. Quinn, A. M. Burdin, P. J. Clapham, J. K. B. Ford, C. M. Gabriele, R. LeDuc, D. Mattila, L. Rojas-Bracho, J. M. Straley, B. L. Taylor, J. Urbán R., D. Weller, B. H. Witteveen, M. Yamaguchi, A. Bendlin, D. Camacho, K. Flynn, A. Havron, J. Huggins, and N. Maloney. (2008). *SPLASH: Structure of Populations, Levels of Abundance and Status of*

Humpback Whales in the North Pacific (Final report for Contract AB133F-03-RP-00078). Olympia, WA: Cascadia Research.

Calambokidis, J., M. A. Kratofil, D. M. Palacios, B. A. Lagerquist, G. S. Schorr, M. B. Hanson, R. W. Baird, K. A. Forney, E. A. Becker, R. C. Rockwood, and E. L. Hazen. (2024). Biologically Important Areas II for cetaceans within U.S. and adjacent waters - West Coast Region. *Frontiers in Marine Science 11*. DOI:10.3389/fmars.2024.1283231

Calambokidis, J., J. Laake, and A. Perez. (2017). *Updated analysis of abundance and population structure of seasonal gray whales in the Pacific Northwest, 1996–2015*. Cambridge, United Kingdom: International Whaling Commission.

Calambokidis, J., G. H. Steiger, C. Curtice, J. Harrison, M. C. Ferguson, E. Becker, M. DeAngelis, and S. M. Van Parijs. (2015a). 4. Biologically Important Areas for selected cetaceans within U.S. waters – West coast region. *In S. M. Van Parijs, C. Curtice, & M. C. Ferguson (Eds.), Biologically Important Areas for cetaceans within U.S. waters Aquatic Mammals (Special Issue) 41* (1): 39–53. DOI:10.1578/AM.41.1.2015.39

Calambokidis, J., G. H. Steiger, C. Curtice, J. Harrison, M. C. Ferguson, E. Becker, M. DeAngelis, and S. M. Van Parijs. (2015b). Biologically Important Areas for Selected Cetaceans Within U.S. Waters – West Coast Region. *Aquatic Mammals 41* (1): 39–53. DOI:10.1578/am.41.1.2015.39

Calambokidis, J., G. H. Steiger, C. Curtice, J. Harrison, M. C. Ferguson, E. Becker, M. DeAngelis, and S. M. Van Parijs. (2015c). Biologically Important Areas for Selected Cetaceans Within U.S. Waters – West Coast Region. *Aquatic Mammals (Special Issue) 41* (1): 39–53. DOI:10.1578/am.41.1.2015.39

Calambokidis, J., G. H. Steiger, J. M. Straley, L. M. Herman, S. Cerchio, D. R. Salden, J. Urban R., J. K. Jacobsen, O. von Ziegesar, K. C. Balcomb, C. M. Gabriele, M. E. Dahlheim, S. Uchida, G. Ellis, Y. Miyamura, P. Ladron De Guevara, M. Yamaguchi, F. Sato, S. A. Mizroch, L. Schlender, K. Rasmussen, J. Barlow, and T. J. Quinn, II. (2001). Movements and population structure of humpback whales in the North Pacific. *Marine Mammal Science 17* (4): 769–794.

Calflora. (2016). *Spartina alterniflora*. Retrieved from http://www.calflora.org/cgi-bin/species_query.cgi?where-calrecnum=7699.

California Coastal Commission. (2018). *The Problem with Marine Debris*. Retrieved August 13, 2018, from https://www.coastal.ca.gov/publiced/marinedebris.html.

California Department of Fish & Game. (2010). *Status of Fisheries Report: An Update Through 2008*. Sacramento, CA: California Department of Fish and Game.

California Department of Fish and Game. (1998). *Report to the Fish and Game Commission: A Status Review of the Spring-Run Chinook Salmon (Oncorhynchus tshawytscha) in the Sacramento River Drainage*. Sacramento, CA: California Department of Fish and Game.

California Department of Fish and Wildlife. (2016). *Southern California Marine Protected Areas*. Retrieved 23 June, 2016, from https://www.wildlife.ca.gov/Conservation/Marine/MPAs/Network/SouthernCalifornia#26822437accessccrtitle14section632a4.

California Department of Fish and Wildlife. (2024a). *Coho Salmon*. Retrieved January 4, 2024, from https://wildlife.ca.gov/Conservation/Fishes/Coho-Salmon.

California Department of Fish and Wildlife. (2024b). *What are Marine Invasive Species?* Retrieved July 5, 2024, from https://wildlife.ca.gov/OSPR/Science/Marine-Invasive-Species-Program/Definition.

California Invasive Plant Council. (2016). *Invasive Plants of California's Wildland: Spartina alterniflora*. Retrieved 7/13/2016, from http://www.cal-ipc.org/ip/management/ipcw/pages/detailreport.cfm@usernumber=75&surveynumber=182.php.

California Wreck Divers. (2017). *Wreck Alley.* Retrieved May 30, 2017, from http://www.cawreckdivers.org/WreckAlley.htm.

Campbell, G. S., L. Thomas, K. Whitaker, A. B. Douglas, J. Calambokidis, and J. A. Hildebrand. (2014). Inter-annual and Seasonal Trends in Cetacean Distribution Density and Abundance off Southern California. *Deep-Sea Research II 112* 143–157. DOI:10.1016/j.dsr2.2014.10.008

Campbell, G. S., L. Thomas, K. Whitaker, A. B. Douglas, J. Calambokidis, and J. A. Hildebrand. (2015). Inter-annual and seasonal trends in cetacean distribution, density and abundance off southern California. *Deep Sea Research Part II: Topical Studies in Oceanography 112* 143–157. DOI:10.1016/j.dsr2.2014.10.008

Canadas, A., R. Sagarminaga, and S. Garcia-Tiscar. (2002). Cetacean distribution related with depth and slope in the Mediterranean waters off southern Spain. *Deep Sea Research I 49* 2053–2073.

Cao, L., S. Wang, M. Zheng, and H. Zhang. (2014). Sensitivity of ocean acidification and oxygen to the uncertainty in climate change. *Environmental Research Letters 9* (2014): 1–10.

Capistrán, M. M. E., N. Crane, L. Crowder, G. G. Melo, J. A. Seminoff, and D. Johnston. (2023). Local Ecological Knowledge enhances our capacity to document potential climate sentinels: a loggerhead sea turtle (*Caretta caretta*) case study. *bioRxiv* 2023.2011. 2001.565199. DOI:10.1101/2023.11.01.565199

Carlton, J. T. and L. G. Eldredge. (2015). Update and revisions of the marine bioinvasions of Hawaiʻi: The introduced and cryptogenic marine and estuarine animals and plants of the Hawaiian Archipelago. *Bishop Museum Bulletin in Zoology 9* 25–47.

Carlton, J. T. and E. Schwindt. (2024). The assessment of marine bioinvasion diversity and history. *Biological Invasions 26* (1): 237–298. DOI:10.1007/s10530-023-03172-7

Carmody, D. J., J. B. Pearce, and W. E. Yasso. (1973). Trace metals in sediments of New York Bight. *Marine Pollution Bulletin 4* 132–135.

Carpenter, R. C., C. A. Lantz, E. Shaw, and P. J. Edmunds. (2018). Responses of coral reef community metabolism in flumes to ocean acidification. *Marine Biology 165* (66): 1–13. DOI:https://doi.org/10.1007/s00227-018-3324-0

Carr, A. and A. B. Meylan. (1980). Evidence of passive migration of green turtle hatchlings in *Sargassum*. *Copeia 1980* (2): 366–368.

Carr, S. H., F. Tatman, and F. A. Chapman. (1996). Observations on the natural history of the Gulf of Mexico sturgeon (*Acipenser oxyrinchus de sotoi* Vladykov 1955) in the Suwannee River, southeastern United States. *Ecology of Freshwater Fish 5* 169–174.

Carretta, J. (2021). *Estimates of marine mammal, sea turtle, and seabird bycatch in the California large-mesh drift gillnet fishery: 1990-2019* (NOAA Technical Memorandum). Washington, DC: U.S. Department of Commerce.

Carretta, J., V. Helker, M. Muto, J. Greenman, K. Wilkinson, D. Lawson, J. Viezbicke, and J. Jannot. (2019a). *Sources of Human-Related Injury and Mortality for U.S. Pacific Coast Marine Mammal Stock Assessments, 2013–2017*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, Marine Mammal and Sea Turtle Division.

Carretta, J., J. Moore, and K. Forney. (2019b). *Estimates of Marine Mammal, Sea Turtle, and Seabird Bycatch from the California Large-Mesh Drift Gillnet Fishery: 1990-2017*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J., E. M. Oleson, K. A. Forney, A. L. Bradford, A. Yano, D. Weller, A. Lang, J. Baker, A. Orr, B. Hanson, J. E. Moore, M. Wallen, and R. L. Brownell Jr. (2024). *Draft U.S. Pacific Marine Mammal Stock Assessments: 2023*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V. (2022a). *Estimates of marine mammal, sea turtle, and seabird bycatch in the California large-mesh drift gillnet fishery: 1990-2021*. La Jolla, CA: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., Erin M. Oleson, Karin A. Forney, David W. Weller, Aimée R. Lang, Jason Baker, Anthony J. Orr, Brad Hanson, Jay Barlow, Jeffrey E. Moore, Megan Wallen, and Robert L. Brownell Jr. (2023). *U.S. Pacific Marine Mammal Stock Assessments: 2022* (NOAA Technical Memorandum NMFS-SWFSC-684.). Washington, DC: U.S. Department of Commerce.

Carretta, J. V., K. A. Forney, and J. L. Laake. (1998). Abundance of southern California coastal bottlenose dolphins estimated from tandem aerial surveys. *Marine Mammal Science 14* (4): 655–675.

Carretta, J. V., K. A. Forney, M. S. Lowry, J. Barlow, J. Baker, D. Johnston, B. Hanson, R. L. Brownell, Jr., J. Robbins, D. Mattila, K. Ralls, M. M. Muto, D. Lynch, and L. Carswell. (2010). *U.S. Pacific Marine Mammal Stock Assessments: 2009* (NOAA Technical Memorandum NMFS-SWFSC-453). La Jolla, CA: Southwest Fisheries Science Center.

Carretta, J. V., K. A. Forney, E. Oleson, K. Martien, M. M. Muto, M. S. Lowry, J. Barlow, J. Baker, B. Hanson, D. Lynch, L. Carswell, R. L. Brownell, J. Robbins, D. K. Mattila, K. Ralls, and M. C. Hill. (2011). *U.S. Pacific Marine Mammal Stock Assessments: 2010* (NOAA Technical Memorandum NMFS-SWFSC-476). La Jolla, CA: Southwest Fisheries Science Center.

Carretta, J. V., J. Greenman, K. Wilkinson, J. Freed, L. Saez, D. Lawson, J. Viezbicke, and J. Jannot. (2021a). *Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2015-2019*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., J. Greenman, K. Wilkinson, L. Saez, D. Lawson, and J. Viezbicke. (2023a). *Sources of human-related injury and mortality for U.S. Pacific West Coast marine mammal stock assessments, 2017-2021. U.S. Department of Commerce,* (NOAA Technical Memorandum NMFS-SWFSC-690). La Jolla, CA: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., V. Helker, M. M. Muto, J. Greenman, K. Wilkinson, D. Lawson, J. Viezbicke, and J. Jannot. (2018). *Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine*

Mammal Stock Assessments. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., Justin Greenman, Kristin Wilkinson, Lauren Saez, Dan Lawson, Justin Viezbicke. (2022b). *Sources of human-related injury and mortality for U.S. Pacific west coast marine mammal stock assessments, 2016-2020*. National Marine Fisheries Service.

Carretta, J. V., M. S. Lowry, C. E. Stinchcomb, M. S. Lynn, and R. E. Cosgrove. (2000). *Distribution and abundance of marine mammals at San Clemente Island and surrounding offshore waters: Results from aerial and ground surveys in 1998 and 1999*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., M. M. Muto, J. Greenman, K. Wilkinson, D. Lawson, J. Viezbicke, and J. Jannot. (2017a). *Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2011–2015* (NOAA Technical Memorandum NMFS-SWFSC-579). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., M. M. Muto, S. Wilkin, J. Greenman, K. Wilkinson, M. DeAngelis, J. Viezbicke, D. Lawson, and J. Jannot. (2016a). *Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2010–2014*. Southwest Fisheries Science Center.

Carretta, J. V., M. M. Muto, S. Wilkin, J. Greenman, K. Wilkinson, M. DeAngelis, J. Viezbicke, D. Lawson, and J. Jannot. (2016b). *Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2010–2014*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., E. Oleson, D. W. Weller, A. R. Lang, K. A. Forney, J. Baker, B. Hanson, K. Martien, M. M. Muto, M. S. Lowry, J. Barlow, D. Lynch, L. Carswell, R. L. J. Brownell, D. K. Mattila, and M. C. Hill. (2013). *U.S. Pacific Marine Mammal Stock Assessments: 2012*. U.S. DEPARTMENT OF COMMERCE, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., E. Oleson, D. W. Weller, A. R. Lang, K. A. Forney, J. Baker, M. M. Muto, B. Hanson, A. J. Orr, H. Huber, M. S. Lowry, J. Barlow, J. Moore, D. Lynch, L. Carswell, and R. L. Brownell. (2015). *U.S. Pacific Marine Mammal Stock Assessments: 2014* (NOAA Technical Memorandum NMFS-SWFSC-549). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Carretta, J. V., E. M. Oleson, J. Baker, D. W. Weller, A. R. Lang, K. A. Forney, M. M. Muto, B. Hanson, A. J. Orr, H. Huber, M. S. Lowry, J. Barlow, J. E. Moore, D. Lynch, L. Carswell, and R. L. Brownell, Jr. (2017b). *U.S. Pacific Marine Mammal Stock Assessments: 2016* (NOAA Technical Memorandum NMFS-SWFSC-561). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., E. M. Oleson, K. A. Forney, D. W. W. A. R. Lang, J. Baker, A. J. Orr, B. Hanson, J. Barlow, J. E. Moore, M. Wallen, and R. L. B. Jr. (2023b). *U.S. Pacific Marine Mammal Stock Assessments: 2022* (NOAA Technical Memorandum NMFS-SWFSC-684.). Washington, DC: U.S. Department of Commerce.

Carretta, J. V., E. M. Oleson, K. A. Forney, M. M. Muto, D. W. Weller, A. R. Lang, J. Baker, B. Hanson, A. J. Orr, J. Barlow, J. E. Moore, and R. L. J. Brownell. (2021b). *U.S. Pacific Marine Mammal Stock Assessments: 2020* (NMFS-SWFSC-646). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V., S. M. Wilkin, M. M. Muto, K. Wilkinson, and J. Rustin. (2014). *Sources of Human-Related Injury and Mortality for U.S. Pacific West Coast Marine Mammal Stock Assessments, 2008–2012*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carretta, J. V. E. M. O., K. A. Forney, M. M. Muto, D. W. Weller, A. R. Lang, J. Baker, B. Hanson, A. J. Orr, J. Barlow, J. E. Moore, and R. L. B. r. (2022). *U.S. Pacific Marine Mammal Stock Assessments: 2021* (NOAA Technical Memorandum NMFS-SWFSC-663). U.S. Department of Commerce.

Carter, H. R. and K. J. Kuletz. (1995). Mortality of Marbled Murrelets Due to Oil Pollution in North America. In C. J. Ralph, G. L. Hunt, Jr., M. G. Raphael, & J. F. Piatt (Eds.), *Ecology and Conservation of the Marbled Murrelet* (Vol. General Technical Report PSW-152, pp. 261–269). Washington, DC: U.S. Department of Agriculture Forest Service.

Carter, H. R., S. G. Sealy, E. E. Burkett, and J. F. Piatt. (2005). Biology and conservation of Xantus's Murrelet: Discovery, taxonomy, and distribution. *Marine Ornithology 33* 81–87.

Cascadia Research. (2012). *Beaked Whales in Hawaii*. Retrieved from http://www.cascadiaresearch.org/hawaii/beakedwhales.htm.

Cascadia Research Collective. (2010, 2 April 2010). *Hawaii's False Killer Whales*. Retrieved 13, 2010, from http://www.cascadiaresearch.org/hawaii/falsekillerwhale.htm.

Cascadia Research Collective. (2012, July 2). *An Update on our June–July 2012 Kauai Project*. Retrieved from http://www.cascadiaresearch.org/hawaii/june2012.htm.

Castro, P. and M. E. Huber. (2000a). Marine animals without a backbone. In *Marine Biology* (3rd ed., pp. 104–138). New York, NY: McGraw-Hill.

Castro, P. and M. E. Huber. (2000b). Marine prokaryotes, protists, fungi, and plants. In *Marine Biology* (3rd ed., pp. 83–103). New York, NY: McGraw-Hill.

Castro, P. and M. E. Huber. (2000c). Marine prokaryotes, protists, fungi, and plants. In: *Marine Biology* (3rd ed., pp. 83–103). New York, NY: McGraw-Hill.

Castro, P. and M. E. Huber. (2007). Chemical and physical features of seawater and the world ocean *Marine Biology* (6th ed., pp. 45–68). New York, NY: McGraw-Hill.

Catano, L. B., B. K. Gunn, M. C. Kelley, and D. E. Burkepile. (2015). Predation risk, resource quality, and reef structural complexity shape territoriality in a coral reef herbivore. *PLoS ONE 10* (2): e0118764. DOI:10.1371/journal.pone.0118764

Cato, D. H. and M. J. Bell. (1992). *Ultrasonic Ambient Noise in Australian Shallow Waters at Frequencies up to 200 kHz*. Ascot Vale, Australia: Materials Research Laboratory.

Catton, C. A., K. L. Stierhoff, and L. Rogers-Bennett. (2016). Population status assessment and restoration modeling of white abalone *Haliotis sorenseni* in California. *Journal of Shellfish Research 35* (3): 593–599. DOI:10.2983/035.035.0304

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

Center for Biological Diversity. (2009). *Petition to List 83 Coral Species under the Endangered Species Act*. San Francisco, CA: Center for Biological Diversity.

Center for Disease Control and Prevention. (2004). *Red Tide: Harmful Algal Blooms*. Retrieved from http://www.cdc.gov/hab/redtide/pdfs/about.pdf.

Center for Disease Control and Prevention. (2024). *USDA Reported H5N1 Bird Flu Detection in US Wild Birds*. Retrieved September 16, 2024, from https://www.cdc.gov/bird-flu/situation-summary/data-map-wild-birds.html.

Cetacean and Turtle Assessment Program. (1982). *A Characterization of Marine Mammals and Turtles in the Mid and North Atlantic Areas of the U.S. Outer Continental Shelf*. Kingston, RI: University of Rhode Island, Graduate School of Oceanography.

Chaloupka, M., G. H. Balazs, and T. M. Work. (2009). Rise and fall over 26 years of a marine epizootic in Hawaiian green sea turtles. *Journal of Wildlife Diseases 45* (4): 1138–1142.

Chaloupka, M., P. Dutton, and H. Nakano. (2004). *Status of sea turtle stocks in the Pacific* (Papers presented at the Expert Consultation on Interactions between Sea Turtles and Fisheries Within an Ecosystem Context). Rome, Italy: United Nations Environment Programme, Food and Agriculture Organization of the United Nations.

Chaloupka, M., T. M. Work, G. H. Balazs, S. K. K. Murakawa, and R. Morris. (2008). Cause-specific temporal and spatial trends in green sea turtle strandings in the Hawaiian Archipelago (1982–2003). *Marine Biology 154* 887–898. DOI:10.1007/s00227-008-0981-4.

Chambault, P., B. De Thoisy, K. Heerah, A. Conchon, S. Barrioz, V. Dos Reis, R. Berzins, L. Kelle, B. Picard, and F. Roquet. (2016). The influence of oceanographic features on the foraging behavior of the olive ridley sea turtle *Lepidochelys olivacea* along the Guiana coast. *Progress in Oceanography 142* 58–71.

Chang, M. (2015). Reducing microplastics from facial exfoliating cleansers in wastewater through treatment versus consumer product decisions. *Marine Pollution Bulletin 101* 330–333. DOI:10.1016/j.marpolbul.2015.10.074

Chapman, P. M., F. Wang, C. R. Janssen, R. R. Goulet, and C. N. Kamunde. (2003). Conducting ecological risk assessments of inorganic metals and metalloids: Current status. *Human and Ecological Risk Assessment 9* (4): 641–697.

Chase, S., R. Benkert, D. Manning, and S. White. (2005). *Sonoma County Water Agency's Mirabel Rubber Dam/Wohler Pool fish samplling program: Year 5 results, 2004*. Sonoma County Water Agency.

Cheroske, A. G., S. L. Williams, and R. C. Carpenter. (2000). Effects of physical and biological disturbances on algal turfs in Kaneohe Bay, Hawaii. *Journal of Experimental Marine Biology and Ecology 248* 1–34.

Chester, R. (2003). *Marine Geochemistry* (2nd ed.). Malden, MA: Blackwell Publishing Company.

Cheung, W., J. Alder, V. Karpouzi, R. Watson, V. Lam, C. Day, K. Kaschner, and D. Pauly. (2005) Patterns of Species Richness in the High Seas. *Technical Series no. 20*. Montreal, Canada: Secretariat of the Convention on Biological Diversity.

Cheung, W. W. L., R. Watson, T. Morato, T. J. Pitcher, and D. Pauly. (2007). Intrinsic vulnerability in the global fish catch. *Marine Ecology-Progress Series 333* 1–12.

Chimner, R. A., B. Fry, M. Y. Kaneshiro, and N. Cormier. (2006). Current Extent and Historical Expansion of Introduced Mangroves on Oʻahu, Hawaiʻi. *Pacific Science 60* (3): 377-383.

Chivers, S. J., R. W. Baird, D. J. McSweeney, D. L. Webster, N. M. Hedrick, and J. C. Salinas. (2007). Genetic variation and evidence for population structure in eastern North Pacific false killer whales (*Pseudorca crassidens*). *Canadian Journal of Zoology 85* (7): 783–794. DOI:10.1139/z07-059

Chmura, G. L. (2009). Tidal Salt Marshes In D. D. A. Laffoley & G. Grimsditch (Eds.), *The Management of Natural Coastal Carbon Sinks* (pp. 5–11). Gland, Switzerland: International Union for the Conservation of Nature.

Clapham, P. J. (2000). The humpback whale: Seasonal feeding and breeding in a baleen whale. In J. Mann, R. C. Connor, P. L. Tyack, & H. Whitehead (Eds.), *Cetacean Societies: Field Studies of Dolphins and Whales* (pp. 173–196). Chicago, IL: University of Chicago Press.

Clapham, P. J. and D. K. Mattila. (1990). Humpback whale songs as indicators of migration routes. *Marine Mammal Science 6* (2): 155–160.

Clark, C. W., M. W. Brown, and P. Corkeron. (2010). Visual and acoustic surveys for North Atlantic right whales, *Eubalaena glacialis*, in Cape Cod Bay, Massachusetts, 2001–2005: Management implications. *Marine Mammal Science 26* (4): 837–843. DOI:10.111/j.1748-7692.2010.00376

Clark, H. R. and C. J. Gobler. (2016). Diurnal fluctuations in CO_2 and dissolved oxygen concentrations do not provide a refuge from hypoxia and acidification for early-life-stage bivalves. *Marine Ecology Progress Series 558* 1–14. DOI:10.3354/meps11852

Clark, R., W. Morrison, M. J. Allen, and L. Claflin. (2005). *Chapter 3: Biogeography of macroinvertebrates* (A Biogeographic Assessment of the Channel Islands National Marine Sanctuary: A Review of Boundary Expansion Concepts for NOAA's National Marine Sanctuary Program). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Ocean Service, National Centers for Coastal Ocean Science.

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.

Clavero, M., L. Brotons, P. Pons, and D. Sol. (2009). Prominent role of invasive species in avian biodiversity loss. *Biological Conservation 142* (10): 2043–2049. DOI:10.1016/j.biocon.2009.03.034

Clements, J. C., K. D. Woodard, and H. L. Hunt. (2016). Porewater acidification alters the burrowing behavior and post-settlement dispersal of juvenile soft-shell clams (*Mya arenaria*). *Journal of Experimental Marine Biology and Ecology 477* 103–111.

Cohen, A. L., D. C. McCorkle, S. de Putron, G. A. Gaetani, and K. A. Rose. (2009). Morphological and compositional changes in the skeletons of new coral recruits reared in acidified seawater: Insights into the biomineralization response to ocean acidification. *Geochemistry Geophysics Geosystems 10* (7): 1–12. DOI:10.1029/2009GC002411

Cole, M. and T. S. Galloway. (2015). Ingestion of nanoplastics and microplastics by Pacific oyster larvae. *Environmental Science & Technology 49* (24): 14625–14632. DOI:10.1021/acs.est.5b04099

Colella, M., R. Ruzicka, J. Kidney, J. Morrison, and V. Brinkhuis. (2012). Cold-water event of January 2010 results in catastrophic benthic mortality on patch reefs in the Florida Keys. *Coral Reefs* 1–12. DOI:10.1007/s00338-012-0880-5

Coleman, J. M. and D. B. Prior. (1988). Mass wasting on continental margins. *Annual Review of Earth Planet Science 16* 101–119.

Coleman, R. R., M. R. Gaither, B. Kimokeo, F. G. Stanton, B. W. Bowen, and R. J. Toonen. (2014). Large-scale introduction of the Indo-Pacific damselfish *Abudefduf vaigiensis* into Hawai'i promotes genetic swamping of the endemic congener *A. abdominalis*. *Molecular Ecology 23* (22): 5552–5565. DOI:10.1111/mec.12952

Colin, P. L. and A. C. Arneson. (1995a). Molluscs: Phylum *Mollusca*. In *Tropical Pacific Invertebrates: A Field Guide to the Marine Invertebrates Occurring on Tropical Pacific Coral Reefs, Seagrass Beds and Mangroves* (pp. 157–200). Beverly Hills, CA: Coral Reef Press.

Colin, P. L. and A. C. Arneson. (1995b). Sponges: Phylum *Porifera*. In *Tropical Pacific Invertebrates: A Field Guide to the Marine Invertebrates Occurring on Tropical Pacific Coral Reefs, Seagrass Beds and Mangroves* (pp. 17–62). Beverly Hills, CA: Coral Reef Press.

Committee on Taxonomy. (2016). *List of Marine Mammal Species & Subspecies - Society for Marine Mammalogy*. Retrieved from https://www.marinemammalscience.org/species-information/list-of-marine-mammal-species-subspecies/.

Committee on the Status of Endangered Wildlife in Canada. (2003). *COSEWIC Assessment and Status Report on the Short-tailed Albatross Phoebastrie albatrus in Canada*. Ottawa, Canada: Canadian Wildlife Service.

Compagno, L. J. V. (1984). *FAO species catalogue. Sharks of the world. An annotated and illustrated catalogue of shark species known to date. Part 2. Carcharhiniformes* (FAO Fisheries Synopsis No. 125). Tiburon, CA: San Francisco State University.

Conant, T. A., P. H. Dutton, T. Eguchi, S. P. Epperly, C. C. Fahy, M. H. Godfrey, S. L. MacPherson, E. E. Possardt, B. A. Schroeder, J. A. Seminoff, M. L. Snover, C. M. Upite, and B. E. Witherington. (2009). *Loggerhead sea turtle (Caretta caretta) 2009 status review under the U.S. Endangered Species Act* (Report of the loggerhead biological review team to the National Marine Fisheries Service, August 2009). Silver Spring, MD: Loggerhead Biological Review Team.

Congdon, B. C., C. A. Erwin, D. R. Peck, G. B. Baker, M. C. Double, and P. O'Neill. (2007). Vulnerability of seabirds on the Great Barrier Reef to climate change. In J. E. Johnson & P. A. Marshall (Eds.), *Climate Change and the Great Barrier Reef: A Vulnerability Assessment* (pp. 427–463). Townsville, Australia: Great Barrier Reef Marine Park Authority and Australian Greenhouse Office.

Cook, T. R., X. Bonnet, T. Fauvel, R. Shine, and F. Brischoux. (2015). Foraging behaviour and energy budgets of sea snakes: Insights from implanted data loggers. *Journal of Zoology*.

Cook, T. R., M. Hamann, L. Pichegru, F. Bonadonna, D. Grémillet, and P. G. Ryan. (2011). GPS and time-depth loggers reveal underwater foraging plasticity in a flying diver, the Cape Cormorant. *Marine Biology 159* (2): 373–387. DOI:10.1007/s00227-011-1815-3

Cooke, J. G. (2018). *Abundance Estimates for Western North Pacific Gray Whales for Use with Stock Structure Hypotheses of the Range-Wide Review of the Population Structure and Status of North*

Pacific Gray Whales (SC/67B/ASI/02). Impington, United Kingdom: International Whaling Commission.

Cooke, J. G., D. W. Weller, A. L. Bradford, O. Sychenko, A. M. Burdin, A. R. Lang, and R. L. Brownell, Jr. (2015, 22–24 November 2015). *Updated Population Assessment of the Sakhalin Gray Whale Aggregation based on the Russia-U.S. photoidentification study at Piltun, Sakhalin, 1994–2014* [PUBLIC]. Presented at the Western Gray Whale Advisory Panel.

Cooper, D. A. and P. L. Corcoran. (2010). Effects of mechanical and chemical processes on the degradation of plastic beach debris on the island of Kauai, Hawaii. *Marine Pollution Bulletin 60* 650–654. DOI:10.1016/j.marpolbul.2009.12.026

Corcoran, A., M. Dornback, B. Kirkpatrick, and A. Jochens. (2013). *A Primer on Gulf of Mexico Harmful Algal Blooms*. College Station, TX: Gulf of Mexico Alliance and the Gulf of Mexico Coastal Ocean Observing System.

Cordoleani, F., J. Noth, A. S. McHuron, A. J. Ammann, and C. J. Michel. (2019). *Movement and survival of wild Chinook salmon smolts from Butte Creek during their outmigration to the ocean: comparison of a dry versus wet year; Author Manuscript*.

Cordoleani, F., C. C. Phillips, A. Sturrock, G. Whitman, A. Malkassian, P. K. Weber, and R. C. Johnson. (2021a). *Threatened salmon rely on a rare life history strategy in a modified and warming landscape*. Lawrence Livermore National Security, LLC.

Cordoleani, F., C. C. Phillips, G. Whitan, A. Sturrock, and R. C. Johnson. (2021b). *Life History Diversity in Central Valley Butte Creek Spring-run Chinook Salmon Population: Implications for Future Management*. Sacramento, CA: State Water Contractors.

Cordoleani, F., W. H. Satterthwaite, M. E. Daniels, and M. R. Johnson. (2020). *Using Life-Cycle Models to Identify Monitoring Gaps for Central Valley Spring-Run Chinook Salmon* (San Francisco Estuary & Watershed Science).

Cornell Lab of Ornithology. (2009). *All About Birds. Sandpipers, Phalaropes, and Allies (Order: Charadiiformes, Family: Scolopacidae)*. Retrieved 2010, from http://www.allaboutbirds.org/guide/browse_tax.aspx?family=53.

Cortes, N. J. and M. J. Risk. (1985). A reef under siltation stress: Cahuita, Costa Rica. *Bulletin of Marine Science 36* (2): 339–356.

Costa-Pierce, B. A. (2012). Aquaculture in Ancient Hawaii. *Bioscience 37* (5): 320–331.

Costante, D. (2021). *Endangered Species Act: Quantifying Threats Impacting Listing.* (M.S.). William & Mary, East Petersburg, PA. (Paper 1627047881.).

Courbis, S., R. W. Baird , F. Cipriano, and D. Duffield. (2014). Multiple Populations of Pantropical Spotted Dolphins in Hawaiian Waters. *Journal of Heredity 105* (5): 627–641. DOI:10.1093/jhered/esu046

Courtney, M. B., E. A. Miller, A. M. Boustany, K. S. Van Houtan, M. R. Catterson, J. Pawluk, J. Nichols, and A. C. Seitz. (2022). Ocean migration and behavior of steelhead *Oncorhynchus mykiss* kelts from the Situk River, Alaska. *Environmental Biology of Fishes 105* (8): 1081–1097. DOI:10.1007/s10641-022-01323-8

Couture, F., G. Oldford, V. Christensen, L. Barrett-Lennard, and C. Walters. (2022). Requirements and availability of prey for northeastern pacific southern resident killer whales. *PLoS ONE 17* (6). DOI:10.1371/journal.pone.0270523

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. (1979). *Classification of Wetlands and Deepwater Habitats of the United States*. Washington, DC: Northern Prairie Wildlife Research Center Home Page. Retrieved from http://www.npwrc.usgs.gov/resource/1998/classwet/classwet.htm.

Crain, C. M., B. S. Halpern, M. W. Beck, and C. V. Kappel. (2009). Understanding and Managing Human Threats to the Coastal Marine Environment. In R. S. Ostfeld & W. H. Schlesinger (Eds.), *The Year in Ecology and Conservation Biology, 2009* (pp. 39–62). Oxford, United Kingdom: Blackwell Publishing.

Crear, D., D. Lawson, J. Seminoff, T. Eguchi, and R. LeRoux. (2017). Habitat use and behavior of the east Pacific green turtle, *Chelonia mydas* in an urbanized system. *Bulletin of the Southern California Academy of Science 116* (1): 17–32.

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

Crowell, S. E., A. M. Wells-Berlin, C. E. Carr, G. H. Olsen, R. E. Therrien, S. E. Ynnuzzi, and D. R. Ketten. (2015). A comparison of auditory brainstem responses across diving bird species. *Journal of Comparative Physiology A 201* (8): 803–815. DOI:10.1007/s00359-015-1024-5

Crozier, L. G., M. M. McClure, T. Beechie, S. J. Bogard, D. A. Boughton, M. Carr, T. D. Cooney, J. B. Dunham, G. C. M., M. A. Haltuch, E. L. Hazen, D. M. Holzer, D. D. Huff, R. C. Johnson, C. E. Jordan, I. C. Kaplan, S. T. Lindley, N. J. Mantua, P. B. Moyle, J. M. Myers, M. W. Nelson, B. C. Spence, L. A. Weitkamp, T. H. Williams, and E. Willis-Norton. (2019). Climate vulnerability assessment for Pacific salmon and steelhead in the California Current Large Marine Ecosystem. *PLoS ONE 14* (7): 49. DOI:https://doi.org/10.1371/journal.pone.0217711

Culbertson, J. B., I. Valiela, M. Pickart, E. E. Peacock, and C. M. Reddy. (2008). Long-term consequences of residual petroleum on salt marsh grass. *Journal of Applied Ecology 45* (4): 1284–1292. DOI:10.1111/j.1365-2664.2008.01477

Cummings, W. C. (1985). Bryde's whale, *Balaenoptera edeni* Anderson, 1878. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 3, pp. 137–154). San Diego, CA: Academic Press.

Cunning, R., R. Ritson-Williams, and R. D. Gates. (2016). Patterns of bleaching and recovery of Montipora capitata in Kaneohe Bay, Hawaii, USA. *Marine Ecology Progress Series 551* 131–139. DOI:10.3354/meps11733

Currie, J. J., S. H. Stack, and G. D. Kaufman. (2017a). Modelling whale-vessel encounters: The role of speed in mitigating collisions with humpback whales (*Megaptera novaeangliae*). *Journal of Cetacean Research Management 17* 57–63.

Currie, J. J., S. H. Stack, J. A. McCordic, and G. D. Kaufman. (2017b). Quantifying the risk that marine debris poses to cetaceans in coastal waters of the 4-island region of Maui. *Marine Pollution Bulletin 121* (1–2): 69–77. DOI:10.1016/j.marpolbul.2017.05.031

Curtis, K. A., J. Calambokidis, K. Audley, M. G. Castaneda, J. De Weerdt, A. J. García Chávez, F. Garita, P. Martínez-Loustalot, J. D. Palacios-Alfaro, and B. Pérez. (2022). *Abundance of Humpback Whales (Megaptera novaeangliae) wintering in Central America and Southern Mexico from a one-dimensional spatial capture-recapture model* (NOAA Technical Memorandum). La Jolla, CA: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Curtis, K. A., E. A. Falcone, G. S. Schorr, J. E. Moore, D. J. Moretti, J. Barlow, and E. Keene. (2020). *Abundance, survival, and annual rate of change of Cuvier's beaked whales (Ziphius cavirostris) on a Navy sonar range* (Marine Mammal Science).

D'Agnese, E., D. Lambourn, J. Rice, D. Duffield, J. Huggins, T. Spraker, S. Raverty, T. Kuzmina, M. E. Grigg, K. Wilkinson, S. Jeffries, and W. Smith. (2020). Reemergence of Guadalupe fur seals in the U.S. Pacific Northwest: The epidemiology of stranding events during 2005–2016. *Marine Mammal Science 36* (3): 828–845.

D'Amico, A. (2009, December 7–10, 2009). *Deep diving cetacean behavioral response study MED 09*. Presented at the 2009 ONR Marine Mammal Program Review. Alexandria, VA.

D'Amico, A. (2013). *Analysis of Monk Seal Behavior Relative to Navy Activities*. San Diego, CA: Space and Naval Warfare Systems Center Pacific.

Dagit, R., D. Alavrez, A. D. Bella, S. Contreras, B. Demiric, P. House, A. Kahler, R. Kieffer, E. Montgomery, H. Neutzel, and J. C. Garza. (2019). *Steelhead abundance monitoring in the Santa Monica Bay, January 2017 – November 2019*. Topanga, CA: Resource Conservation District of the Santa Monica Mountains.

Dagit, R., M. T. Booth, M. Gomez, T. E. Hovey, S. Howard, S. D. Lewis, S. Jacobson, M. Larson, D. McCanne, and T. H. Robinson. (2020). Occurrences of Steelhead Trout (*Oncorhynchus mykiss*) in Southern California, 1994-2018. *California Fish and Game 106* (1): 38-56.

Dahlheim, M. E. and J. E. Heyning. (1999). Killer whale, *Orcinus orca* (Linnaeus, 1758). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 6, pp. 281–322). San Diego, CA: Academic Press.

Dahlheim, M. E., A. Schulman-Janiger, N. Black, R. Ternullo, D. Ellifrit, and K. C. Balcomb, III. (2008). Eastern temperate North Pacific offshore killer whales (*Orcinus orca*): Occurrence, movements, and insights into feeding ecology. *Marine Mammal Science 24* (3): 719–729. DOI:10.1111/j.1748-7692.2008.00206.x

Dalebout, M. L., J. G. Mead, C. S. Baker, A. N. Baker, and A. L. van Helden. (2002). A new species of beaked whale *Mesoplodon perrini* sp. n. (Cetacea: Ziphiidae) discovered through phylogenetic analyses of mitochondrial DNA sequences. *Marine Mammal Science 18* (3): 577–608.

Dalebout, M. L., G. J. B. Ross, C. S. Baker, R. C. Anderson, P. B. Best, V. G. Cockcroft, H. L. Hinsz, V. Peddemors, and R. L. Pitman. (2003). Appearance, distribution and genetic distinctiveness of Longman's beaked whale, *Indopacetus pacificus*. *Marine Mammal Science 19* (3): 421–461.

Daly-Engel, T. S., K. D. Seraphin, K. N. Holland, J. P. Coffey, H. A. Nance, R. J. Toonen, and B. W. Bowen. (2012). Global phylogeography with mixed-marker analysis reveals male-mediated dispersal in the endangered scalloped hammerhead shark (*Sphyrna lewini*). *PLoS ONE 7* (1): e29986. DOI:10.1371/journal.pone.0029986

Dameron, O. J., M. Parke, M. A. Albins, and R. Brainard. (2007). Marine debris accumulation in the Northwestern Hawaiian Islands: An examination of rates and processes. *Marine Pollution Bulletin 54* 423–433. DOI:10.1016/j.marpolbul.2006.11.019

Davenport, J. (1988). Do diving leatherbacks pursue glowing jelly? *British Herpetological Society Bulletin 24* 20–21.

Davis, G. E., P. L. Haaker, and D. V. Richards. (1998a). The perilous condition of white abalone, *Haliotis sorenseni*, Bartsch, 1940. *Journal of Shellfish Research 17* (3): 871–875.

Davis, R. W., W. E. Evans, and B. Würsig, (Eds.). (2000). *Cetaceans, Sea Turtles and Seabirds in the Northern Gulf of Mexico: Distribution, Abundance and Habitat Associations*. New Orleans, LA: U.S. Geological Survey, Biological Resource Division; and Minerals Management Service, Gulf of Mexico Region.

Davis, R. W., G. S. Fargion, N. May, T. D. Leming, M. Baumgartner, W. E. Evans, L. J. Hansen, and K. Mullin. (1998b). Physical habitat of cetaceans along the continental slope in the north-central and western Gulf of Mexico. *Marine Mammal Science 14* (3): 490–507.

Davoren, G. K., P. Penton, C. Burke, and W. A. Montevecchi. (2012). Water temperature and timing of capelin spawning determine seabird diets. *ICES Journal of Marine Science 69* (7): 1234–1241. DOI:10.1093/icesjms/fss032

Dawes, C. J. (1998). *Marine Botany* (2nd ed.). New York, NY: John Wiley and Sons, Inc.

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. H. and B. A. Cooper. (1995). Patterns of movement of Dark-rumped petrels and Newell's shearwaters on Kauai. *The Condor 97* 1011–1027.

Day, R. H., B. A. Cooper, and R. J. Blaha. (2003). Movement patterns of Hawaiian petrels and Newell's shearwaters on the island of Hawaii. *Pacific Science 57* (2): 147–159.

Day, R. H. and D. A. Nigro. (2000). Feeding ecology of Kittlitz's and marbled murrelets in Prince William Sound, Alaska. *Waterbirds 23* (1): 1–14.

De Carlo, E. H., D. Hoover, C. Young, R. Hoover, and F. Mackenzie. (2007). Impact of storm runoff from tropical watersheds on coastal water quality and productivity. *Applied Geochemistry 22* (8): 1777–1797. DOI:10.1016/j.apgeochem.2007.03.034

De Laubenfels, M. W. (1950). The sponges of Kaneohe Bay, Oahu. *Pacific Science 4* (1): 3–36.

De Laubenfels, M. W. (1951). The sponges of the Island of Hawaii. *Pacific Science 5* (3): 256–271.

Dearborn, D. C., A. D. Anders, and P. G. Parker. (2001). Sexual dimorphism, extrapair fertilizations, and operational sex ratio in great frigatebirds (*Fregata minor*). *Behavioral Ecology 12* (6): 746–752. DOI:10.1093/beheco/12.6.746

Debich, A. J., S. Baumann-Pickering, A. Širović, J. A. Hildebrand, A. L. Alldredge, R. S. Gottlieb, S. T. Herbert, S. C. Johnson, A. C. Rice, L. K. Roche, B. J. Thayre, J. S. Trickey, L. M. Varga, and S. M. Wiggins. (2015a). *Passive Acoustic Monitoring for Marine Mammals in the SOCAL Naval Training Area Dec 2012–Jan 2014* (MPL Technical Memorandum #552). La Jolla, CA: Marine Physical Laboratory, Scripps Institution of Oceanography.

Debich, A. J., S. Baumann-Pickering, A. Širović, J. A. Hildebrand, S. T. Herbert, S. C. Johnson, A. C. Rice, J. S. Trickey, and S. M. Wiggins. (2015b). *Passive Acoustic Monitoring for Marine Mammals in the SOCAL Range Complex January–July 2014* (MPL Techincal Memorandum #554). La Jolla, CA: Marine Physical Laboratory Scripps Institution of Oceanography University of California San Diego.

Defenders of Wildlife. (2015a). *A Petition to List the Giant Manta Ray (Manta birostris), Reef Manta Ray (Manta alfredi), and Caribbean Manta Ray (Manta c.f. birostris) as Endangered, or Alternatively as Threatened, Species Pursuant to the Endangered Species Act and for the Concurrent Designation of Critical Habitat*. Denver, CO: Defenders of Wildlife.

Defenders of Wildlife. (2015b). *A Petition to List the Oceanic Whitetip Shark (Carcharhinus longimanus) as an Endangered, or Alternatively as a Threatened, Species Pursuant to the Endangered Species Act and for the Concurrent Designation of Critical Habitat*. Denver, CO: Defenders of Wildlife.

Defran, R. H., M. Caldwell, E. Morteo, A. Lang, and M. Rice. (2015). *Possible Stock Structure of Coastal Bottlenose Dolphins off Baja California and California Revealed by Photo-Identification Research* (Bulletin of the Southern California Academy of Sciences).

Defran, R. H. and D. W. Weller. (1999). Occurrence, distribution, site fidelity, and school size of bottlenose dolphins (*Tursiops truncatus*) off San Diego, California. *Marine Mammal Science 15* (2): 366–380.

Delean, B. J., V. T. Helker, M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, J. Jannot, and N. C. Young. (2020). *Human-caused mortality and injury of NMFS-managed Alaska marine mammal stocks 2013-2017* (NOAA Technical Memorandum). Springfield, VA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center.

DeMartini, E. E. and J. E. Smith. (2015). Effects of fishing on the fishes and habitat of coral reefs. In C. Mora (Ed.), *Ecology of Fishes on Coral Reefs*. Cambridge, United Kingdom: Cambridge University Press.

Demina, L. L. and S. V. Galkin. (2009). Geochemical features of heavy metal bioaccumulation in the Guaymas Basin of the Gulf of California. *Oceanology 49* (5): 697–706. DOI:10.1134/s0001437009050117

Denkinger, J., M. Parra, J. P. Muñoz, C. Carrasco, J. C. Murillo, E. Espinosa, F. Rubianes, and V. Koch. (2013). Are boat strikes a threat to sea turtles in the Galapagos Marine Reserve? *Ocean & Coastal Management 80* 29–35.

Dentlinger, S. (2023). *Four Decades of Green Turtle (Chelonia mydas) Strandings on Hawaiʻi Island (1983–2022): Identifying Causes and Assessing Trends.* (Bachelor's in Marine Science). University of Hawaiʻi at Hilo, Hilo, HI.

DeRuiter, S. L., B. L. Southall, J. Calambokidis, W. M. Zimmer, D. Sadykova, E. A. Falcone, A. S. Friedlaender, J. E. Joseph, D. Moretti, G. S. Schorr, L. Thomas, and P. L. Tyack. (2013). First direct measurements of behavioural responses by Cuvier's beaked whales to mid-frequency active sonar. *Biology Letters 9* (4): 20130223. DOI:10.1098/rsbl.2013.0223

Desholm, M., A. D. Fox, P. D. L. Beasley, and J. Kahlert. (2006). Remote techniques for counting and estimating the number of bird-wind turbine collisions at sea: A review. *IBIS 148* 76–89. DOI:10.1111/j.1474-919X.2006.00509

DiMatteo, A., G. Lockhart, and S. Barco. (2022). Habitat models and assessment of habitat partitioning for Kemp’s ridley and loggerhead marine turtles foraging in Chesapeake Bay (USA). *Endangered Species Research 47* 91–107. DOI:doi.org/10.3354/esr01168

Division of Aquatic Resources. (2021). *Rapid Response to three species of Invasive Corals in Kāneʻohe Bay, Oʻahu, Hawaiʻi*. Honolulu, HI: Hawai'i Division of Aquatic Resources, Aquatic Invasive Species Team.

Dizon, A. E., W. F. Perrin, and P. A. Akin. (1994). *Stocks of dolphins (Stenella spp. and Delphinus delphis) in the eastern tropical Pacific: A phylogeographic classification*. NOAA.

Dobroski, N., C. Brown, R. Nedelcheva, C. Scianni, and J. Thompson. (2015). *2015 Beinnial Report on the California Marine Invasive Species Program*. Sacramento, CA: California State Lands Commission Marine Facilities Division.

Dodd, C. K., Jr. (1988). *Synopsis of the Biological Data on the Loggerhead Sea Turtle, Caretta caretta (Linnaeus 1758)*. Washington, DC: U.S. Fish and Wildlife Service.

Dodder, N., K. Schiff, A. Latker, and C. L. Tang (2016). *Southern California Bight 2013 Regional Monitoring Program: Volume IV. Sediment Chemistry* (Technical Report 922). Costa Mesa, CA: Southern California Coastal Water Research Project Authority.

Dohl, T. P., R. C. Guess, M. L. Duman, and R. C. Helm. (1983). *Cetaceans of central and northern California, 1980-1983: Status, abundance, and distribution*.

Doi, Y., Y. Kanesawa, N. Tanahashi, and Y. Kumagai. (1992). Biodegradation of microbial polyesters in the marine environment. *Polymer Degradation and Stability 36* 173–177.

Donahue, M. A. and W. L. Perryman. (2008). Pygmy killer whale, *Feresa attenuata*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 938–939). San Diego, CA: Academic Press.

Doney, S. C., M. Ruckelshaus, D. J. Emmett, J. P. Barry, F. Chan, C. A. English, H. M. Galindo, J. M. Grebmeier, A. B. Hollowed, N. Knowlton, J. Polovina, N. N. Rabalais, W. J. Sydeman, and L. D. Talley. (2012). Climate change impacts on marine ecosystems. *Annual Review of Marine Science 4* (1): 11–37. DOI:10.1146/annurev-marine-041911-111611

Donohue, M. J. and D. G. Foley. (2007). Remote sensing reveals links among the endangered Hawaiian monk seal, marine debris, and El Niño. *Marine Mammal Science 23* (2): 468–473. DOI:10.1111/j.1748-7692.2007.00114

Dooling, R. (2002). *Avian Hearing and the Avoidance of Wind Turbines*. College Park, MD: University of Maryland.

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

Doropoulos, C., S. Ward, G. Diaz-Pulido, O. Hoegh-Guldberg, and P. J. Mumby. (2012). Ocean acidification reduces coral recruitment by disrupting intimate larval-algal settlement interactions. *Ecology Letters 15* (4): 338–346. DOI:10.1111/j.1461-0248.2012.01743

Dorsey, E. M., S. J. Stern, A. R. Hoelzel, and J. Jacobsen. (1990). Minke Whales (*Balaenoptera acutorostrata*) from the West Coast of North America: Individual Recognition and Small-Scale Site Fidelity. *Reports of the International Whaling Commission 12* 357–368.

Douglas, A. B., J. Calambokidis, L. M. Munger, M. S. Soldevilla, M. Ferguson, A. M. Havron, D. L. Camacho, G. S. Campbell, and J. A. Hildebrand. (2014a). Seasonal distribution and abundance of cetaceans off Southern California estimated from CalCOFI cruise data from 2004 to 2008. *Fishery Bulletin 112* (2-3): 198–219. DOI:10.7755/FB.112.2-3.7

Douglas, A. B., J. Calambokidis, L. M. Munger, M. S. Soldevilla, M. C. Ferguson, A. M. Havron, D. L. Camacho, G. S. Campbell, and J. A. Hildebrand. (2014b). Seasonal distribution and abundance of cetaceans off Southern California estimated from CalCOFI cruise data from 2004 to 2008. *Fishery Bulletin 112* (2–3): 198–220. DOI:10.7755/fb.112.2-3.7

Downs, C. A., E. Kramarsky-Winter, C. M. Woodley, A. Downs, G. Winters, Y. Loya, and G. K. Ostrander. (2009). Cellular pathology and histopathology of hypo-salinity exposure on the coral *Stylophora pistillata*. *Science of the Total Environment 407* (17): 4838–4851. DOI:10.1016/j.scitotenv.2009.05.015

Drazen, J. C. and B. A. Seibel. (2007). Depth-related trends in metabolism of benthic and benthopelagic deep-sea fishes. *Limnology and Oceanography 52* (5): 2306–2316.

Dreyer, G. D. and W. A. Niering. (1995). Tidal marshes of Long Island Sound: Ecology, history and restoration. *Connecticut Arboretum Bulletin 34* (2): 1–45.

Du, B., C. Wong, K. McLaughlin, and K. Schiff. (2018). *Sediment Chemistry: Southern California Bight 2018 Regional Monitoring Program Volume II*. Costa Mesa, CA: Southern California Coastal Water Research Project.

Dubinsky, Z. and I. Berman-Frank. (2001). Uncoupling primary production from population growth in photosynthesizing organisms in aquatic ecosystems. *Aquatic Sciences 63* 4–17.

Dudzik, K. J., K. M. Baker, and D. W. Weller. (2006). *Mark-Recapture Abundance Estimate of California Coastal Stock Bottlenose Dolphins: February 2004 to April 2005* (NOAA Administrative Report LJ-06-02C). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Dueñas, M.-A., H. J. Ruffheaad, N. H. Wakefield, P. D. Roberts, D. J. Hemming, and H. Diaz-Soltero. (2018). The role played by invasive species in interactions with endangered and threatened species in the United States: a systematic review. *Biodiversity and Conservation 2018* (27): 3171-3183.

Dufour, F., H. Arrizabalaga, X. Irigoien, and J. Santiago. (2010). Climate impacts on albacore and bluefin tunas migrations phenology and spatial distribution. *Progress In Oceanography 86* (1–2): 283–290. DOI:10.1016/j.pocean.2010.04.007

Dugan, J. E., D. M. Hubbard, D. L. Martin, J. M. Engle, D. M. Richards, G. E. Davis, K. D. Lafferty, and R. F. Ambrose. (2000). Macrofauna communities of exposed sandy beaches on the southern California mainland and Channel Islands. In D. R. Browne, K. L. Mitchell, & H. W. Chaney (Eds.), *Proceedings of the Fifth California Islands Symposium, 29 March–1 April 1999 (OCS Study MMS 99-0038)* (pp. 339–346). Santa Barbara, CA: University of Calfornia, Santa Barbara, Marine Science Institute.

Dugan, J. E., D. M. Hubbard, K. J. Nielsen, J. Altstatt, and J. Bursek. (2015). *Baseline Characterization of Sandy Beach Ecosystems Along the South Coast of California. Final Report*. La Jolla, CA: University of California San Diego.

Duncan, K. M. and K. N. Holland. (2006). Habitat use, growth rates and dispersal patterns of juvenile scalloped hammerhead sharks, *Sphyrna lewini,* in a nursery habitat. *Marine Ecology Progress Series 312* 211–221. DOI:10.3354/meps312211

Duncan, K. M., A. P. Martin, B. W. Bowen, and H. G. De Couet. (2006). Global phylogeography of the scalloped hammerhead shark (*Sphyrna lewini*). *Molecular Ecology 15* (8): 2239–2251.

Eakin, C. M., G. Liu, A. M. Gomez, J. L. De La Cour, S. F. Heron, W. J. Skirving, E. F. Geiger, K. V. Tirak, and A. E. Strong. (2016). Global Coral Bleaching 2014–2017. *Reef Encounter 31* 20–26.

Eckert, K. L., B. P. Wallace, J. R. Spotila, and B. A. Bell. (2015). Nesting Ecology and Reproductive Investments of the Leatherback Turtle. *The Leatherback Turtle: Biology and Conservation* 63.

Eckert, S. A., K. L. Eckert, P. Ponganis, and G. L. Kooyman. (1989). Diving and foraging behavior of leatherback sea turtles (*Dermochelys coriacea*). *Canadian Journal of Zoology 67* 2834–2840.

Eckert, S. A. and L. Sarti-Martinez. (1997). Distant fisheries implicated in the loss of the world's largest leatherback nesting population. *Marine Turtle Newsletter 78* 2–7.

Edmonds, N. J., C. J. Firmin, D. Goldsmith, R. C. Faulkner, and D. T. Wood. (2016). A review of crustacean sensitivity to high amplitude underwater noise: Data needs for effective risk assessment in relation to UK commercial species. *Marine Pollution Bulletin 108* 5–11. DOI:10.1016/j.marpolbul.2016.05.006

Edwards, H. H. (2013). Potential impacts of climate change on warmwater megafauna: The Florida manatee example (*Trichechus manatus latirostris*). *Climatic Change 121* (4): 727–738. DOI:10.1007/s10584-013-0921-2

Edwards, K. P., J. A. Hare, F. E. Werner, and B. O. Blanton. (2006). Lagrangian circulation on the Southeast U.S. Continental Shelf: Implications for larval dispersal and retention. *Continental Shelf Research 26* (12–13): 1375–1394. DOI:10.1016/j.csr.2006.01.020

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

Eggleton, J. and K. V. Thomas. (2004). A review of factors affecting the release and bioavailability of contaminants during sediment disturbance events. *Environment International 30* (7): 973–980. DOI:10.1016/j.envint.2004.03.001

Eguchi, T. (2015, 2 November 2015). Southern California sea turtle abundance. Personal communication between Tomoharu Eguchi (National Marine Fisheries Service Southwest Fisheries Science Center) and Mike Zickel (ManTech International) via email.

Eguchi, T., S. R. Benson, D. G. Foley, and K. A. Forney. (2017). Predicting overlap between drift gillnet fishing and leatherback turtle habitat in the California Current Ecosystem. *Fisheries Oceanography 26* (1): 17–33.

Eguchi, T., J. Bredvik, S. Graham, R. LeRoux, B. Saunders, and J. A. Seminoff. (2020). Effects of a power plant closure on home ranges of green turtles in an urban foraging area. *Endangered Species Research 41* 265-277.

Eguchi, T., S. McClatchie, C. Wilson, S. R. Benson, R. A. LeRoux, and J. A. Seminoff. (2018). Loggerhead turtles (*Caretta caretta*) in the California current: Abundance, distribution, and anomalous warming of the North Pacific. *Frontiers in Marine Science 5* (452): 1–15.

Ehrlich, P. R., D. S. Dobkin, and D. Wheye. (1988). *The Birder's Handbook: A Field Guide to the Natural History of North American Birds*. New York, NY: Simon & Schuster, Inc.

Eisenberg, J. F. and J. Frazier. (1983). A leatherback turtle (*Dermochelys coriacea*) feeding in the wild. *Journal of Herpetology 17* (1): 81–82.

Eldredge, L. G. and J. T. Carlton. (2002). Hawaiian marine bioinvasions: A preliminary assessment. *Pacific Science 56* (2): 211–212.

Eldredge, L. G. and C. M. Smith. (2001). *A Guidebook of Introduced Marine Species in Hawaii* (Bishop Museum Technical Report 21). Honolulu, HI: Bishop Museum.

Ellis, M. (2016, July 13). *Disentangling a Whale of a Problem*. Retrieved from http://www.nmfs.noaa.gov/stories/2016/07/whale-entanglement.html.

Elphick, C., J. B. Dunning, Jr., and D. A. E. Sibley. (2001). *National Audubon Society: The Sibley Guide to Bird Life and Behavior*. New York, NY: Alfred A. Knopf, Inc.

Elphick, J. (2007). *Atlas of Bird Migration: Tracing the Great Journeys of the World's Birds*. Buffalo, NY: Firefly Books.

Emmett, R. L., S. A. Hinton, S. L. Stone, and M. E. Monaco. (1991). *Distribution and Abundance of Fishes and Invertebrates in West Coast Estuaries*. Rockville, MD: NOAA/NOS Strategic Environmental Assessments Division.

Engelhaupt, A., J. Aschettino, T. A. Jefferson, D. Engelhaupt, and M. Richlen. (2016). *Occurrence, Distribution, and Density of Marine Mammals Near Naval Station Norfolk and Virginia Beach, Virginia*. Virginia Beach, VA: U.S. Fleet Forces Command.

Enticott, J. and D. Tipling. (1997). *Seabirds of the World: The Complete Reference* (1st ed.). Mechanicsburg, PA: Stackpole Books.

Erbe, C. (2002). Underwater noise of whale-watching boats and potential effects on killer whales (*Orcinus orca*), based on an acoustic impact model. *Marine Mammal Science 18* (2): 394–418.

Erftemeijer, P. L. A., B. Riegl, B. W. Hoeksema, and P. A. Todd. (2012). Environmental impacts of dredging and other sediment disturbances on corals: A review. *Marine Pollution Bulletin 64* 1737–1765.

Erickson, D. L. and J. E. Hightower. (2007). Oceanic distribution and behavior of green sturgeon. *American Fisheries Society Symposium 56* 197–211.

Erickson, R. A., R. A. Hamilton, S. N. G. Howell, P. Pyle, and M. A. Patten. (1995). First record of the marbled murrelet and third record of the ancient murrelet for Mexico. *Western Birds 26* 39–45.

Ericson, P. G. P., I. Envall, M. Irestedt, and J. A. Norman. (2003). Inter-familial relationships of the shorebirds (Aves: Charadriiformes) based on nuclear DNA sequence data. *BMC Evolutionary Biology 3* (16): 1–14.

Estrada, J. A., A. N. Rice, M. E. Lutcavage, and G. B. Skomal. (2003). Predicting trophic position in sharks of the north-west Atlantic ocean using stable isotope analysis. *Journal of the Marine Biological Association of the United Kingdom 83* 1347–1350.

Etnoyer, P. and L. Morgan. (2003). *Occurrences of Habitat-forming Deep Sea Corals in the Northeast Pacific Ocean: A Report to NOAA's Office of Habitat Conservation*. Redmond, WA: Marine Conservation Biology Institute.

Etnoyer, P. and L. E. Morgan. (2005). Habitat-forming deep-sea corals in the Northeast Pacific Ocean. In A. Freiwald & J. M. Roberts (Eds.), *Cold-water Corals and Ecosystems* (pp. 331–343). Berlin, Germany: Springer-Verlag.

Etnoyer, P. J. (2010). Deep-sea corals on seamounts. *Oceanography 23* (1): 128–129.

Fabry, V. J., B. A. Seibel, R. A. Feely, and J. C. Orr. (2008). Impacts of ocean acidification on marine fauna and ecosystem processes. *ICES Journal of Marine Science 65* (3): 414–432.

Fain, M. G. and P. Houde. (2007). Multilocus perspectives on the monophyly and phylogeny of the order Charadriiformes (Aves). *BMC Evolutionary Biology 7* (35): 1–17. DOI:10.1186/1471-2148-7-35

Falcone, E. A., B. Diehl, A. Douglas, and J. Calambokids. (2011). *Photo-Identification of Fin Whales (Balaeanoptera physalus) along the US West Coast, Baja California, and Canada*. Olympia, WA: Cascadia Research Collective.

Falcone, E. A. and G. S. Schorr. (2011). *Distribution and demographics of marine mammals in SOCAL through photo-indentification, genetics, and satellite telemetry: a summary of surveys conducted 15 July 2010 - 24 June 20111*. Monterey, CA: U.S. Navy Postgraduate School.

Falcone, E. A. and G. S. Schorr. (2012). *Distribution and demographics of marine mammals in SOCAL through photo-indentification, genetics, and satellite telemetry: a summary of surveys conducted 1 July 2011 - 15 June 2012*. Monterey, CA: U.S. Navy Postgraduate School.

Falcone, E. A. and G. S. Schorr. (2013). *Distribution and demographics of marine mammals in SOCAL through photo-indentification, genetics, and satellite telemetry: a summary of surveys conducted 1 July 2012–30 June 2013*. Monterey, CA: Naval Postgraduate School.

Falcone, E. A. and G. S. Schorr. (2014). *Distribution and Demographics of Marine Mammals in SOCAL through Photo-Identification, Genetics, and Satellite Telemetry* (Prepared for Chief of Naval Operations Energy and Environmental Readiness Division: NPS-OC-14-005CR). Monterey, CA: Naval Postgraduate School.

Falcone, E. A., G. S. Schorr, A. B. Douglas, J. Calambokidis, E. Henderson, M. F. McKenna, J. Hildebrand, and D. Moretti. (2009). Sighting characteristics and photo-identification of Cuvier's beaked whales (*Ziphius cavirostris*) near San Clemente Island, California: A key area for beaked whales and the military? *Marine Biology 156* 2631–2640.

Fay, C., M. Bartron, S. Craig, A. Hecht, J. Pruden, R. Saunders, T. Sheehan, and J. Trial. (2006). *Status Review for Anadromous Atlantic Salmon (Salmo salar) in the United States*. Silver Spring, MD: U.S. Fish and Wildlife Service; and National Marine Fisheries Service.

Felis, J. J., J. Adams, C. Horton, E. C. Kelsey, and L. M. White. (2022). *Abundance and productivity of marbled murrelets (Brachyramphus marmoratus) off central California during the 2020 and 2021 breeding seasons* (U.S. Geological Survey Data Report 1157). Reston, VA: U.S. Geological Survey.

Felline, S., R. Caricato, A. Cutignano, S. Gorbi, M. G. Lionetto, E. Mollo, F. Regoli, and A. Terlizzi. (2012). Subtle effects of biological invasions: Cellular and physiological responses of fish eating the exotic pest *Caulerpa racemosa*. *PLoS ONE 7* (6): e38763. DOI:10.1371/journal.pone.0038763

Ferguson, M. C. (2005). *Cetacean Population Density in the Eastern Pacific Ocean: Analyzing Patterns With Predictive Spatial Models.* (Unpublished Doctoral Dissertation). University of California, San Diego, La Jolla, CA. Retrieved from http://daytonlab.ucsd.edu.

Ferguson, M. C., J. Barlow, S. B. Reilly, and T. Gerrodette. (2006). Predicting Cuvier's (*Ziphius cavirostris*) and *Mesoplodon* beaked whale population density from habitat characteristics in the eastern tropical Pacific Ocean. *Journal of Cetacean Research and Management 7* (3): 287–299.

Ferguson, M. C., C. Curtice, J. Harrison, and S. M. Van Parijs. (2015). Biologically important areas for cetaceans within U.S. waters – Overview and rationale. *Aquatic Mammals (Special Issue) 41* (1): 2–16. DOI:10.1578/am.41.1.2015.2

Ferrero, R. C. and W. A. Walker. (1996). Age, growth, and reproductive patterns of the Pacific white-sided dolphin (*Lagenorhynchus obliquidens*) taken in high seas drift nets in the central North Pacific Ocean. *Canadian Journal of Zoology 74* 1673–1687.

Ferrero, R. C. and W. A. Walker. (1999). Age, growth, and reproductive patterns of Dall's porpoise (*Phocoenoides dalli*) in the central north Pacific Ocean. *Marine Mammal Science 15* (2).

Fiedler, P. C., E. A. Becker, K. A. Forney, J. Barlow, and J. E. Moore. (2023). Species Distribution Modeling of Deep-diving Cetaceans. *Marine Mammal Science*.

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

Finkbeiner, E. M., B. P. Wallace, J. E. Moore, R. L. Lewison, L. B. Crowder, and A. J. Read. (2011). Cumulative estimates of sea turtle bycatch and mortality in USA fisheries between 1990 and 2007. *Biological Conservation 144* (11): 2719–2727.

Fish, M. R., I. M. Cote, J. A. Horrocks, B. Mulligan, A. R. Watkinson, and A. P. Jones. (2008). Construction setback regulations and sea-level rise: Mitigating sea turtle nesting beach loss. *Ocean & Coastal Management 51* (4): 330–341.

Fisher, F. W. (1994). Past and present status of Central Valley Chinook salmon. *Conservation Biology 8* (3): 870–873.

Fisher, J., M. Trudel, A. Ammann, J. A. Orsi, J. Piccolo, C. Bucher, E. Casillas, J. A. Harding, B. MacFarlane, R. D. Brodeur, J. F. T. Morris, and D. W. Welch. (2007). Comparisons of the coastal distributions and abundances of juvenile Pacific salmon from central California to the northern Gulf of Alaska. *American Fisheries Society Symposium 57* 31–80.

Fisher, J. P., L. A. Weitkamp, D. J. Teel, S. A. Hinton, J. A. Orsi, E. V. Farley Jr., J. F. T. Morris, M. E. Thiess, R. M. Sweeting, and M. Trudel. (2014). Early Ocean Dispersal Patterns of Columbia River Chinook and Coho Salmon. *Transactions of the American Fisheries Society 143* (1): 252–272.

Fisheries and Oceans Canada. (2011). *2011–2015 Integrated Fisheries Management Plan for Atlantic Seals: Harp (Pagophilus groenlandicus), Hooded (Cystophora cristata), Grey (Halichoerus grypus), Ringed (Phoca hispida), Bearded (Erignathus barbatus), Harbour (Phoca vitulina)*. Ottawa, Canada: Fisheries and Oceans Canada.

Fisheries and Oceans Canada. (2016a, 7 April 2016). *Grey Seal Competitive Fleet in Atlantic Canada*. Retrieved 23 August 2016, from http://www.dfo-mpo.gc.ca/decisions/fm-2016-gp/atl-02-eng.htm.

Fisheries and Oceans Canada. (2016b, 7 April 2016). *Harp Seal and Hooded Seal Competitive Fleet in Newfoundland and Labrador, Quebec, Gulf and Maritime Regions*. Retrieved 23 August 2016, from http://www.dfo-mpo.gc.ca/decisions/fm-2016-gp/atl-03-eng.htm.

Fitzgerald, W. F., C. H. Lamborg, and C. R. Hammerschmidt. (2007). Marine biogeochemical cycling of mercury. *Chemical Reviews 107* 641–662.

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.

Fleming, A. H., C. T. Clark, J. Calambokidis, and J. Barlow. (2016). Humpback whale diets respond to variance in ocean climate and ecosystem conditions in the California Current. *Global Change Biology 22* (3): 1214–1224. DOI:10.1111/gcb.13171

Fletcher, N., L. Anderson, K. Melanson, and P. Raimondi. (2021). *Investigations into recruitment enhancement and recovery of black abalone (Haliotis cracherodii) on San Clemente Island*. San Diego, CA: Naval Facilities Engineering Command Southwest.

Florida Fish and Wildlife Conservation Commission. (2015). *Manatee Synoptic Surveys*. Retrieved 14 November 2015, 2015, from http://myfwc.com/research/manatee/research/population-monitoring/synoptic-surveys/.

Food and Agriculture Organization of the United Nations. (2005). *Review of the State of World Marine Fishery Resources*. Rome, Italy: Food and Agriculture Organization Fisheries Department, Fishery Resources Division, Marine Resources Service.

Food and Agriculture Organization of the United Nations. (2009). *The State of World Fisheries and Aquaculture*. Rome, Italy: Food and Agriculture Organization Fisheries Department, Fishery Resources Division, Marine Resources Service.

Food and Agriculture Organization of the United Nations. (2013). *Report of the Fourth FAO Expert Advisory Panel for the Assessment of Proposals to Amend Appendices I and II of CITES Concerning Commercially-Exploited Aquatic Species*. Rome, Italy: Food and Agriculture Organization Fisheries Department, Fishery Resources Division, Marine Resources Service.

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

Ford, J. K. B. and G. M. Ellis. (1999). *Transients: Mammal-Hunting Killer Whales of British Columbia, Washington, and Southeastern Alaska*. Vancouver, Canada and Seattle, WA: UBC Press and University of Washington Press.

Ford, J. K. B., G. M. Ellis, and K. C. I. Balcomb. (2000). *Killer Whales: The Natural History and Genealogy of Orcinus Orca in British Columbia and Washington State (Updated Edition)*. Seattle, WA: University of Washington Press.

Ford, J. K. B., E. H. Stredulinsky, G. M. Ellis, J. W. Durban, and J. F. Pilkington. (2014). *Offshore Killer Whales in Canadian Pacific Waters: Distribution, Seasonality, Foraging Ecology, Population Status and Potential for Recovery*. Ottawa, Canada: Department of Fisheries and Oceans Canada, Canadian Science Advisory, Secretariat.

Forney, K., J. Moore, J. Barlow, and J. Carretta. (2020). A multidecadal Bayesian trend analysis of harbor porpoise (*Phocoena phocoena*) populations off California relative to past fishery bycatch. *Marine Mammal Science*, 1–15. Retrieved February 3, 2023, from https://doi.org/10.1111/mms.12764.

Forney, K. A. (2007). *Preliminary Estimates of Cetacean Abundance Along the U.S. West Coast and Within Four National Marine Sanctuaries During 2005*. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Forney, K. A., R. W. Baird, and E. M. Oleson. (2010). *Rationale for the 2010 Revision of Stock Boundaries for the Hawaii Insular and Pelagic Stocks of False Killer Whales, Pseudorca crassidens*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Forney, K. A. and J. Barlow. (1993). Preliminary winter abundance estimates for cetaceans along the California coast based on a 1991 aerial survey. *Reports of the International Whaling Commission 43* 407–415.

Forney, K. A. and J. Barlow. (1998a). Seasonal patterns in the abundance and distribution of California cetaceans, 1991–1992. *Marine Mammal Science 14* (3): 460–489.

Forney, K. A. and J. Barlow. (1998b). Seasonal patterns in the abundance and distribution of California cetaceans, 1991—1992. *Marine Mammal Science 14* (3): 460–489.

Forney, K. A., J. Barlow, and J. V. Carretta. (1995). The abundance of cetaceans in California waters. Part II: Aerial surveys in winter and spring of 1991 and 1992. *Fishery Bulletin 93* 15–26.

Forney, K. A., E. A. Becker, D. G. Foley, J. Barlow, and E. M. Oleson. (2015). Habitat-based models of cetacean density and distribution in the central North Pacific. *Endangered Species Research 27* 1–20. DOI:10.3354/esr00632

Forney, K. A., M. C. Ferguson, E. A. Becker, P. C. Fiedler, J. V. Redfern, J. Barlow, I. L. Vilchis, and L. T. Ballance. (2012). Habitat-based spatial models of cetacean density in the eastern Pacific Ocean. *Endangered Species Research 16* (2): 113–133. DOI:10.3354/esr00393

Forney, K. A. and P. R. Wade. (2006). Worldwide Distribution and Abundance of Killer Whales. In J. A. Estes, R. L. Brownell, Jr., D. P. DeMaster, D. F. Doak, & T. M. Williams (Eds.), *Whales, Whaling and Ocean Ecosystems* (pp. 145–162). Berkeley, CA: University of California Press.

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.

Foster, M. S. and D. R. Schiel. (1985). *The Ecology of Giant Kelp Forests in California: A Community Profile*. Moss Landing, CA: U.S. Fish and Wildlife Service.

Foster, T., J. L. Falter, M. T. McCulloch, and P. L. Clode. (2016). Ocean acidification causes structural deformities in juvenile coral skeletons. *Science Advances 2* (e1501130): 7. DOI:10.1126/sciadv.1501130

Foster, T. and J. P. Gilmour. (2016). Seeing red: Coral larvae are attracted to healthy-looking reefs. *Marine Ecology Progress Series 559* 65–71. DOI:10.3354/meps11902

Frankel, A. S., C. M. Gabriele, and S. Y. A. S. H. Rickards. (2022). Humpback whale abundance in Hawaii: Temporal trends and response to climatic drivers. *Marine Mammal Science 38(1):118-138. 2022.*

Franklin, E. C., P. L. Jokiel, and M. J. Donahue. (2013). Predictive modeling of coral distribution and abundance in the Hawaiian Islands. *Marine Ecology Progress Series 481* 121–132. DOI:10.3354/meps10252

Freiwald, A., J. H. Fosså, A. Grehan, T. Koslow, and J. M. Roberts. (2004). *Cold-water Coral Reefs: Out of Sight–No Longer Out of Mind*. Cambridge, United Kingdom: United Nations Environmental Program World Conservation Monitoring Center.

Friedlander, A., G. Aeby, S. Balwani, B. Bowen, R. Brainard, A. Clark, J. Kenyon, J. Maragos, C. Meyer, P. Vroom, and J. Zamzow. (2008a). The State of Coral Reef Ecosystems of the Northwestern Hawaiian Islands. In J. E. Waddell & A. M. Clarke (Eds.), *The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States* (pp. 263–306). Silver Spring, MD: National Center for Coastal Ocean Science Center for Coastal Monitoring and Assessment's Biogeography Team.

Friedlander, A., G. Aeby, R. Brainard, E. Brown, K. Chaston, A. Clark, P. McGowan, T. Montgomery, W. Walsh, I. Williams, and W. Wiltse. (2008b). The State of Coral Reef Ecosystems of the Main Hawaiian Islands. In J. E. Waddell & A. M. Clarke (Eds.), *The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States* (pp. 219–262). Silver Spring, MD: National Center for Coastal Ocean Science Center for Coastal Monitoring and Assessment's Biogeography Team.

Friedlander, A., K. Keller, L. Wedding, A. Clark, and M. Monaco. (2009). *A Marine Biogeographic Assessment of the Northwestern Hawaiian Islands*. Silver Spring, MD: Office of National Marine Sanctuaries Papahanaumokuakea Marine National Monument.

Fritz, L., K. Sweeney, R. Towell, and T. Gelatt. (2016). *Aerial and Ship-Based Surveys of Stellar Sea Lions (Eumetopias jubatus) Conducted in Alaska in June–July 2013 through 2015, and an Update on the Status and Trend of the Western Distinct Population Segment in Alaska* (National Oceanic and Atmospheric Administration Technical Memorandum NMFS-AFSC-321). Seattle, WA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center.

Froese, R. and D. Pauly. (2010, January 11, 2010). *FishBase*. Retrieved 2010, from www.fishbase.org.

Froese, R. and D. Pauly. (2016a). *FishBase*. *World Wide Web electronic publication.* Retrieved from www.fishbase.org.

Froese, R. and D. Pauly. (2016b). *FishBase*. Retrieved from www.fishbase.org.

Frost, N. (2016). *California Least Tern Breeding Survey 2015 Season*. San Diego, CA: State of California Natural Resources Agency, Department of Fish and Wildlife Wildlife Branch.

Fuentes, M. M. P. B., D. A. Pike, A. Dimatteo, and B. P. Wallace. (2013). Resilience of marine turtle regional management units to climate change. *Global Change Biology 19* (5): 1399–1406.

Fujii, T. (2012). Climate Change, Sea-level Rise, and Implications for Coastal and Estuarine Shoreline Management with Particular Reference to the Ecology of Intertidal Benthic Macrofauna in Northwest Europe. *Biology 1* 597–616. DOI:10.3390/biology1030597

Fujimori, L. (2002, January 18). Elephant seal visits Hawaii shores. *Honolulu Star-Bulletin Hawaii News*. Retrieved from http://starbulletin.com/2002/01/18/news/story7.html.

Fukuoka, T., H. Suganuma, S. Kondo, and K. Sato. (2022). Long dive capacity of olive ridley turtles (*Lepidochelys olivacea*) at high water temperature during the post-nesting foraging period in the Arafura Sea. *Journal of Experimental Marine Biology and Ecology 546* 151649.

Fukuoka, T., M. Yamane, C. Kinoshita, T. Narazaki, G. J. Marshall, K. J. Abernathy, N. Miyazaki, and K. Sato. (2016). The feeding habit of sea turtles influences their reaction to artificial marine debris. *Scientific Reports 6* 28015. DOI:10.1038/srep28015

Fulling, G. L., P. H. Thorson, and J. Rivers. (2011). Distribution and abundance estimates for cetaceans in the waters off Guam and the Commonwealth of the Northern Mariana Islands. *Pacific Science In Press* 46.

Fusaro, C. and C. Anderson. (1980). First California record: The scalloped hammerhead shark, *Sphyrna lewini*, in coastal Santa Barbara waters. *California Fish and Game 66* (2): 121–123.

Gagnon, K., S. J. Peacock, Y. Jin, and M. A. Lewis. (2015). Modelling the spread of the invasive alga Codium fragile driven by long-distance dispersal of buoyant propagules. *Ecological Modelling 316* 111–121. DOI:10.1016/j.ecolmodel.2015.08.011

Gaither, M. R., G. Aeby, M. Vignon, Y. Meguro, M. Rigby, C. Runyon, R. J. Toonen, C. L. Wood, and B. W. Bowen. (2013). An invasive fish and the time-lagged spread of its parasite across the Hawaiian archipelago. *PLoS ONE 8* (2): e56940. DOI:10.1371/journal.pone.0056940

Gallagher, S. P. (2003). *Juvenile salmonid abundance estimation in the upper Noyo River, California during spring and summer 2002; Annual Report; Project 2a2*. California Department of Fish and Game.

Galloway, S. B., A. W. Bruckner, and C. M. Woodley. (2009). *Coral Health and Disease in the Pacific: Vision for Action*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Ocean Service, National Centers for Coastal Ocean Science.

Gannier, A. (2000). Distribution of cetaceans off the Society Islands (French Polynesia) as obtained from dedicated surveys. *Aquatic Mammals 26* (2): 111–126.

Gannier, A. and E. Praca. (2007). SST fronts and the summer sperm whale distribution in the north-west Mediterranean Sea. *Journal of the Marine Biological Association of the UK 87* (01): 187. DOI:10.1017/s0025315407054689

Gaos, A. R. (2011). *Spatial Ecology of Hawksbill Turtles (Eretmochelys Imbricata) in the Eastern Pacific Ocean.* (Master's thesis). San Diego State University, San Diego, CA. Retrieved from http://sdsu-dspace.calstate.edu/bitstream/handle/10211.10/1272/Gaos_Alexander.pdf?sequence=1.

Gaos, A. R., L. Kurpita, H. Bernard, L. Sundquist, C. S. King, J. H. Browning, E. Naboa, I. K. Kelly, K. Downs, and T. Eguchi. (2021). Hawksbill nesting in Hawaiʻi: 30-year dataset reveals recent positive trend for a small, yet vital population. *Frontiers in Marine Science 8* 770424. DOI:10.3389/fmars.2021.770424

Gaos, A. R. and I. L. Yañez. (2008). ¡CAREY! Where have the Eastern Pacific Hawksbills Gone?! *The State of the World's Sea Turtles III* 18–19.

Garcia-Aguilar, M. C., C. Turrent, F. R. Elorriaga-Verplancken, A. Arias-Del-Razo, and Y. Schramm. (2018). Climate change and the northern elephant seal (*Mirounga angustirostris*) population in Baja California, Mexico. *PLoS ONE 13* (2): e0193211. DOI:10.1371/journal.pone.0193211

Garcia, M. O., J. Caplan-Auerbach, E. H. De Carlo, M. D. Kurz, and N. Becker. (2005). Geology, geochemistry and earthquake history of Loihi Seamount, Hawaii's youngest volcano. *Chemie der Erde-Geochemistry 66* (2): 81–108. DOI:10.1016/j.chemer.2005.09.002

Garcia, M. S., J. Netto, and L. Sowerbutts. (2021). *Analyzing the History and Current Status of Invasive Species Management Efforts in the Hawaiian Islands.* (Bachelor dissertation). Worcester, MA: Worcester Polytechnic Institute.

Garrison, T. (2004). Essentials of Oceanography (3rd ed.). Pacific Grove, CA: Brooks/Cole-Thomas Learning.

Gauthreaux, S. A. and C. G. Belser. (2003). Radar ornithology and biological conservation. *Auk 120* (2): 266–277. DOI:10.1642/0004-8038(2003)120[0266:roabc]2.0.co;2

Gentry, R. L. (2009). Northern fur seal, *Callorhinus ursinus*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 788–791). Cambridge, MA: Academic Press.

Gerrodette, T. and T. Eguchi. (2011). Precautionary design of a marine protected area based on a habitat model. *Endangered Species Research 15* 159–166.

Giddens, J., D. R. Kobayashi, G. N. Mukai, J. Asher, C. Birkeland, M. Fitchett, M. A. Hixon, M. Hutchinson, B. C. Mundy, J. M. O'Malley, M. Sabater, M. Scott, J. Stahl, R. J. Toonen, M. S. Trianni, P. A. Woodworth-Jefcoats, J. L. K. Wren, and M. Nelson. (2022). Assessing the vulnerability of marine life to climate change in the Pacific Islands region. *PLoS ONE 17* (7). DOI:10.1371/journal.pone.0270930

Gilbert, J. R. and N. Guldager. (1998). *Status of Harbor and Gray Seal Populations in Northern New England*. Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Gilmartin, W. G. and J. Forcada. (2009). Monk seals *Monachus monachus, M. tropicalis*, and *M. schauinslandi*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 741–744). Cambridge, MA: Academic Press.

Giorli, G. and W. W. L. Au. (2017). Spatio-temporal variation and seasonality of Odontocetes' foraging activity in the leeward side of the island of Hawaii. *Deep-Sea Research I 121* 202–209. DOI:10.1016/j.dsr.2017.01.013

Giraudeau, J. and L. Beaufort. (2007). Coccolithophores: From Extant Populations to Fossil Assemblages *Developments in Marine Geology* (Vol. 1, pp. 409–439).

Gitschlag, G. R. (1996). Migration and diving behavior of Kemp's ridley (Garman) sea turtles along the U.S. southeastern Atlantic coast. *Journal of Experimental Marine Biology and Ecology 205* 115–135.

Global Invasive Species Database. (2005). *List of invasive marine algae in California*. Retrieved from http://www.issg.org/database/species/search.asp?sts=sss&st=sss&fr=1&sn=&rn=California&hci=8&ei=186&lang=EN&Image1.x=22&Image1.y=12.

Goatley, C. H. R. and D. R. Bellwood. (2009). Morphological structure in a reef fish assemblage. *Coral Reefs 28* (2): 449–457. DOI:10.1007/s00338-009-0477-9

Gochfeld, D. J. (2004). Predation-induced morphological and behavioral defenses in a hard coral: Implications for foraging behavior of coral-feeding butterflyfishes. *Marine Ecology Progress Series 267* 145–158.

Godley, B. J., D. R. Thompson, S. Waldron, and R. W. Furness. (1998). The trophic status of marine turtles as determined by stable isotope analysis. *Marine Ecology Progress Series 166* 277–284.

Godwin, S. and H. Bolick. (2006). *Inventory of Intertidal and Shallow Subtidal Marine Invertebrates at Kalaupapa National Historical Park Moloka'i, Hawai'i*. Honolulu, HI: National Park Service, Pacific Islands Coral Reef Program.

Goertler, P., F. Cordoleani, J. Notch, R. Johnson, and G. Singer. (2020). *Life history variation in Central Valley spring-run Chinook* (Spring-run Workshop Factsheet). Sacramento, CA: Delta Stewardship Council.

Goethel, C. L., J. M. Grebmeier, L. W. Cooper, and T. J. Miller. (2017). Implications of ocean acidification in the Pacific Arctic: Experimental responses of three Arctic bivalves to decreased pH and food availability. *Deep-Sea Research Part II: Topical Studies in Oceanography 144* 112–124.

Goetz, L. C., H. Nuetzel, D. L. Vendrami, A. K. Beulke, E. C. Anderson, J. C. Garza, and D. E. Pearse. (2024). Genetic parentage reveals the (un) natural history of Central Valley hatchery steelhead. *Evolutionary Applications*. DOI:10.1111/eva.13681

Goiran, C., G. P. Brown, and R. Shine. (2020). The behaviour of sea snakes (*Emydocephalus annulatus*) shifts with the tides. *Scientific Reports 10* (1): 11346.

Goncalves, R., M. Scholze, A. M. Ferreira, M. Martins, and A. D. Correia. (2008). The joint effect of polycyclic aromatic hydrocarbons on fish behavior. *Environmental Research 108* 204–213. DOI:10.1016/j.envres.2008.07.008

Good, T. P., R. S. Waples, and P. Adams, (Eds.). (2005). *Updated Status of Federally Listed ESUs of West Coast Salmon and Steelhead*. Seattle, WA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Goodall, C., C. Chapman, and D. Neil. (1990). The Acoustic Response Threshold of the Norway Lobster, *Nephrops norvegicus* (L.), in a Free Sound Field. In K. Weise, W. D. Krenz, J. Tautz, H. Reichert, & B. Mulloney (Eds.), *Frontiers in Crustacean Neurobiology* (pp. 106–113). Basel, Switzerland: Birkhauser-Verlag.

Gosho, M., P. Gearin, R. Jenkinson, J. Laake, L. Mazzuca, D. Kubiak, J. Calambokidis, W. Megill, B. Gisborne, D. Goley, C. Tombach, J. Darling, and V. Deecke. (2011). *Movements and diet of gray whales (Eschrichtius robustus) off Kodiak Island, Alaska, 2002–2005*. Presented at the International Whaling Commission AWMP workshop 28 March–1 April 2011. Washington, DC.

Goulet, T. L., K. P. Shirur, B. D. Ramsby, and R. Iglesias-Prieto. (2017). The effects of elevated seawater temperatures on Caribbean gorgonian corals and their algal symbionts, *Symbiodinium* spp. *PLoS ONE 12* (2): 21 (e0171032). DOI:10.1371/journal.pone.0171032

Graham, S., J. Bredvik, and B. Saunders. (2014). *Rocky Intertidal Community Monitoring At Naval Auxiliary Landing Field, San Clemente Island, 2013 Annual Report*. San Diego, CA: Naval Facilities Engineering Command (NAVFAC) Southwest, California. Environmental Readiness, Navy Marine Ecology Consortium, Space and Naval Warfare Systems Center Pacific.

Graham, S. E. and B. Saunders. (2015). *Occurrence, Distribution, and Population Estimates of Marine Mammals near Silver Strand Training Complex and San Diego Bay, CA* (Prepared for Commander, Pacific Fleet. Submitted to Naval Facilities Engineering Command (NAVFAC) Southwest, California, February 2015). San Diego, CA: SPAWAR Systems Center Pacific.

Grant, G. S. and D. Ferrell. (1993). Leatherback turtle, *Dermochelys coriacea* (Reptilia: *Dermochelidae*): Notes on near-shore feeding behavior and association with cobia. *Brimleyana 19* 77–81.

Gravem, S. A., W. N. Heady, V. R. Saccomanno, K. F. Alvstad, A. L. M. Gehman, T. N. Frierson, and S. L. Hamilton. (2021). *Pycnopodia helianthoides* (IUCN Red List of Threatened Species 2021). Cambridge, United Kingdom: UCN Global Species Programme Red List Unit.

Green, G. A., J. J. Brueggeman, R. A. Grotefendt, C. E. Bowlby, M. L. Bonnell, and K. C. Balcomb, III. (1992). *Cetacean Distribution and Abundance off Oregon and Washington, 1989–1990*. Los Angeles, CA: U.S. Department of the Interior, Minerals Management Service.

Greer, C. D., P. V. Hodson, Z. Li, T. King, and K. Lee. (2012). Toxicity of crude oil chemically dispersed in a wave tank to embryos of Atlantic herring (*Clupea harengus*). *Environmental Toxicology and Chemistry 31* 1–10. DOI:10.1002/etc.1828

Gregory, M. R. (2009). Environmental implications of plastic debris in marine settings—entanglement, ingestion, smothering, hangers-on, hitch-hiking and alien invasions. *Philosophical Transactions of the Royal Society of London 364* 2013–2025. DOI:10.1098/rstb.2008.0265

Grenier, J. J. and S. K. Nelson. (1995). Marbled Murrelet habitat associations in Oregon. In C. J. Ralph, G. L. Hunt, Jr., M. G. Raphael, & J. F. Piatt (Eds.), *Ecology and Conservation of the Marbled Murrelet. USDA Forest Service General Technical Report PSW-152* (pp. 191–204). Washington, DC: U.S. Forest Service.

Griffiths, E. T. and J. Barlow. (2016). Cetacean acoustic detections from free-floating vertical hydrophone arrays in the southern California Current. *JASA Express Letters 140* (5): EL399. DOI:10.1121/1.4967012

Grove, R., K. Zabloudil, T. Norall, and L. Deysher. (2002). Effects of El Niño events on natural kelp beds and artificial reefs in southern California. *ICES Journal of Marine Science 59* S330–S337. DOI:10.1006/jmsc.2002.1290

Grupe, B. M., M. L. Krach, A. L. Pasulka, J. M. Maloney, L. A. Levin, and C. A. Frieder. (2015). Methane seep ecosystem functions and services from a recently discovered southern California seep. *Marine Ecology 36 (Supplement 1)* 91–108. DOI:10.1111/maec.12243

Guest, J. R., J. Low, K. Tun, B. Wilson, C. Ng, D. Raingeard, K. E. Ulstrup, J. T. I. Tanzil, P. A. Todd, T. C. Toh, D. McDougald, L. M. Chou, and P. D. Steinberg. (2016). Coral community response to bleaching on a highly disturbed reef. *Scientific Reports 6* 20717. DOI:10.1038/srep20717

Gulko, D. (1998). The Corallivores: The crown-of-thorns sea star (*Acanthaster planci*) *Hawaiian Coral Reef Ecology* (pp. 101–102). Honolulu, HI: Mutual Publishing.

Gulland, F. M., J. Baker, M. Howe, E. LaBrecque, L. Leach, S. E. Moore, R. R. Reeves, and P. O. Thomas. (2022). A Review of Climate Change Effects on Marine Mammals in United States Waters: Past Predictions, Observed Impacts, Current Research and Conservation Imperatives. *Climate Change Ecology* 100054.

Gustafson, R. G., M. J. Ford, D. Teel, and J. S. Drake. (2010). *Status review of eulachon (Thaleichthys pacificus) in Washington, Oregon, and California* (National Oceanic and Atmospheric Administration Technical Memorandum NMFS-NWFSC-105). Seattle, WA and Port Orchard, WA: Northwest Fisheries Science Center Conservation Biology Division.

Gustafson, R. G., L. Weitkamp, Y.-W. Lee, E. Ward, K. Somers, V. Tuttle, and J. Jannot. (2016a). *Status Review Update of Eulachon (Thaleichthys pacificus) Listed under the Endangered Species Act: Southern Distinct Population Segment*. Seattle, WA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Gustafson, R. G., L. Weitkamp, Y.-W. Lee, E. Ward, K. Somers, V. Tuttle, and J. Jannot. (2016b). *Status Review Update of Eulachon (Thaleichthys pacificus) Listed under the Endangered Species Act: Southern Distinct Population Segment*. Seattle, WA: National Marine Fisheries Service, Northwest Fisheries Science Center.

Haedrich, R. L. (1996). Deep-water fishes: Evolution and adaptation in the earth's largest living spaces. *Journal of Fish Biology 49* (Supplement A): 40–53.

Hale, S. S., H. W. Buffum, J. A. Kiddon, and M. M. Hughes. (2017). Subtidal Benthic Invertebrates Shifting Northward Along the US Atlantic Coast. *Estuaries and Coasts 40* (6): 1744–1756. DOI:10.1007/s12237-017-0236-z

Halpern, B., S. Walbridge, K. A. Selkoe, C. V. Kappel, F. Micheli, C. D'Agrosa, J. F. Bruno, K. S. Casey, C. Ebert, H. E. Fox, R. Fujita, D. Heinemann, H. S. Lenihan, E. M. P. Madin, M. T. Perry, E. R. Selig, M.

Spalding, R. S. Steneck, and R. Watson. (2008a). A global map of human impact on marine ecosystems. *Science 319* (5865): 948–952. DOI:doi: 10.1126/science.1149345

Halpern, B. S., K. L. McLeod, A. A. Rosenberg, and L. B. Crowder. (2008b). 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

Hameedi, M., A. Pait, and R. Warner. (2002). *Environmental Contaminant Monitoring in the Gulf of Maine*. Presented at the Northeast Coastal Monitoring Summit. Durham, NH.

Hamer, T. E. and S. K. Nelson. (1995). *Nesting Chronology of the Marbled Murrelet* (Ecology and Conservation of the Marbled Murrelet). Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture.

Hamilton, S. L., V. R. Saccomanno, W. N. Heady, A. L. Gehman, S. I. Lonhart, R. Beas-Luna, F. T. Francis, L. Lee, L. Rogers-Bennett, A. K. Salomon, and S. A. Gravem. (2021). Disease-driven mass mortality event leads to widespread extirpation and variable recovery potential of a marine predator across the eastern Pacific. *Proceedings of the Royal Society B* (288): e20211195. DOI:https://doi.org/10.1098/rspb.2021.1195

Hamilton, T. A., J. V. Redfern, J. Barlow, L. T. Ballance, T. Gerrodette, R. S. Holt, K. A. Forney, and B. L. Taylor. (2009). *Atlas of Cetacean Sightings for Southwest Fisheries Science Center Cetacean and Ecosystem Surveys: 1986–2005* (NOAA Technical Memorandum NMFS-SWFSC-440). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Haney, J., H. Geiger, and J. Short. (2014a). Bird mortality from the Deepwater Horizon oil spill. I: Exposure probability in the offshore Gulf of Mexico. *Marine Ecology Progress Series 513* 225–237. DOI:10.3354/meps10991

Haney, J., H. Geiger, and J. Short. (2014b). Bird mortality from the Deepwater Horizon oil spill. II: Carcass sampling and exposure probability in the coastal Gulf of Mexico. *Marine Ecology Progress Series 513* 239–252. DOI:10.3354/meps10839

Hansen, K. A., A. Hernandez, T. A. Mooney, M. H. Rasmussen, K. Sorensen, and M. Whalberg. (2020). The common murre (*Uria aalge*), an auk seabird, reacts to underwater sound. *The Journal of the Acoustical Society of America 147* (6): 4069–4074.

Hansen, K. A., A. Maxwell, U. Siebert, O. N. Larsen, and M. Wahlberg. (2017). Great cormorants (*Phalacrocorax carbo*) can detect auditory cues while diving. *The Science of Nature 104* (5–6): 45. DOI:10.1007/s00114-017-1467-3

Hansen, L. P. and M. L. Windsor. (2006). Interactions between aquaculture and wild stocks of Atlantic salmon and other diadromous fish species: Science and management, challenges and solutions. *ICES Journal of Marine Science 63* (7): 1159–1161. DOI:10.1016/J.ICEJMS.2006.05.003

Hanser, S. F., S. H. Smith, and R. K. Uyeyama. (In Prep.). *Estimates of Sea Turtle Presence in and Adjacent to Pearl Harbor, Hawaii and Observations of Green Sea Turtle Behavior*. Pearl Harbor, HI: Naval Facilities Engineering Command, Pacific.

Hanson, M. B., E. J. Ward, C. K. Emmons, and M. M. Holt. (2018). *Modeling the occurrence of endangered killer whales near a U.S. Navy Training Range in Washington State using satellite-tag locations to improve acoustic detection data*. Seattle, WA: National Oceanic and

Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center.

Hanson, M. B., E. J. Ward, C. K. Emmons, M. M. Holt, and D. M. Holzer. (2017). *Assessing the movements and occurrence of Southern Resident Killer Whales relative to the U.S. Navy's Northwest Training Range Complex in the Pacific Northwest*. Seattle, WA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center.

Hanson, M. T. and R. H. Defran. (1993). The behavior and feeding ecology of the Pacific coast bottlenose dolphin, *Tursiops truncatus*. *Aquatic Mammals 19* (3): 127–142.

Harding, J. (2015). *Juvenile Salmon Ocean Ecology R/V Ocean Starr, July 5-24, 2014*. Santa Cruz, CA: National Marine Fisheries Service, Southwest Fisheries Science Center, Fisheries Ecology Division.

Harding, J., E. Dick, N. Mantua, B. Wells, A. Ammann, and S. Hayes. (2021). *Distribution patterns of fish and invertebrates from summer salmon surveys in the Central California Current System 2010-2015* (NOAA Technical Memorandum NMFS-SWFSC-645). NOAA.

Hare, J. A., W. E. Morrison, M. W. Nelson, M. M. Stachura, E. J. Teeters, R. B. Griffis, M. A. Alexander, J. D. Scott, L. Alade, R. J. Bell, A. S. Chute, K. L. Curti, T. H. Curtis, D. Kircheis, J. F. Kocik, S. M. Lucey, C. T. McCandless, L. M. Milke, D. E. Richardson, E. Robillard, H. J. Walsh, M. C. McManus, K. E. Marancik, and C. A. Griswold. (2016). A vulnerability assessment of fish and invertebrates to climate change on the Northeast U.S. Continental Shelf. *PLoS ONE 11* (2): 1–30. DOI:10.1371/journal.pone.0146756

Harley, C. D., A. R. Hughes, K. M. Hultgren, B. G. Miner, C. J. B. Sorte, C. S. Thomber, L. F. Rodriguez, T. L., and S. L. Williams. (2006). The impacts of climate change in coastal marine systems. *Ecology Letters 9* 228–242. DOI:10.1111/j.1461-0248.2005.00871

Harrison, C. S., J. Y. Luo, N. F. Putman, Q. Li, P. Sheevam, K. Krumhardt, J. Stevens, and M. C. Long. (2021). Identifying global favourable habitat for early juvenile loggerhead sea turtles. *Journal of the Royal Society Interface 18* (175): 20200799.

Harrison, J., M. C. Ferguson, L. New, J. Cleary, C. Curtice, S. DeLand, E. Fujioka, P. N. Halpin, R. B. T. Moore, and S. M. Van Parijs. (2023). Biologically Important Areas II for cetaceans within U.S. and adjacent waters - Updates and the application of a new scoring system. *Frontiers in Marine Science 10*. DOI:doi.org/10.3389/fmars.2023.1081893

Harrison, P. (1983). *Seabirds, an Identification Guide*. Boston, MA: Houghton Mifflin Company.

Hart, J. L. (1973). *Pacific Fishes of Canada*. Ottawa, Canada: Canadian Government Publishing Centre.

Harting, A. L., J. D. Baker, and T. C. Johanos. (2017). Estimating Population Size for Hawaiian Monk Seals Using Haulout Data. *The Journal of Wildlife Management 81* (7): 1202–1209.

Harvell, C. D., C. E. Mitchell, J. R. Ward, S. Altizer, A. P. Dobson, R. S. Ostfeld, and M. D. Samuel. (2002). Climate warming and disease risks for terrestrial and marine biota. *Science 296* (5576): 2158–2162. DOI:10.1126/science.1063699

Hatfield, B. B. (2005). *The translocation of sea otters to San Nicolas Island: An update*. Presented at the 6th California Islands Symposium. Arcata, CA.

Hatfield, B. B., M. Kenner, and M. T. Tinker. (2016). *Sea otters beginning to thrive at San Nicolas Island*. Presented at the 9th Channel Islands Symposium. Ventura, CA.

Hatfield, B. B., J. L. Yee, M. C. Kenner, and J. A. Tomoleoni. (2019). *California Sea Otter (Enhydra lutris nereis) Census Results, Spring 2019*. Reston, VA: U.S. Geological Survey.

Hawaii Department of Land and Natural Resources. (1980). *Hawaii Coastal Zone Fisheries Management Study*. Honolulu, HI: Department of Land and Natural Resources, Division of Fish and Game.

Hawaii Department of Land and Natural Resources. (2005). *Hawaii's Comprehensive Wildlife Conservation Strategy: Newell's Shearwater (Puffinus auricularis newelli)*. Honolulu, HI: Hawaii Department of Land and Natural Resources.

Hawaii State Department of Health. (2000). *Hawaii's Implementation Plan for Polluted Runoff Control, Appendix F: Detailed Descriptions of Hawaii's 18 Water Quality Limited Segments*. Honolulu, HI: Hawaii Department of Business and Department of Health.

Hawaiian Monk Seal Research Program. (2015). *Posting Regarding Hawaiian Monk Seal Toxoplasmosis*. Retrieved 11/23/2015, from https://www.facebook.com/HMSRP/posts/963396450367039.

Hawaiisfishes. (2017). *Ribbon worms (nemerteans) of Hawaii*. Retrieved 02/22/2017, 2017, from http://www.hawaiisfishes.com/inverts/ribbon_worms/neat_ribbon_worms.htm.

Hawkins, A. D., A. E. Pembroke, and A. N. Popper. (2015). Information gaps in understanding the effects of noise on fishes and invertebrates. *Reviews in Fish Biology and Fisheries 25* 39–64. DOI:10.1007/s11160-014-9369-3

Hayes, M. O., R. Hoff, J. Michel, D. Scholz, and G. Shigenaka. (1992). *An Introduction to Coastal Habitats and Biological Resources for Oil Spill Response*. Seattle, WA: National Oceanic and Atmospheric Administration, Hazardous Materials Response and Assessment Division.

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.

Hayes, S. A., M. H. Bond, B. K. Wells, C. V. Hanson, A. W. Jones, and R. B. MacFarlane. (2011). *Using archival tags to infer habitat use of Central California steelhead and coho salmon*. Presented at the American Fisheries Symposium 76.

Hays, G. C., J. D. R. Houghton, C. Isaacs, R. S. King, C. Lloyd, and P. Lovell. (2004). First records of oceanic dive profiles for leatherback turtles, *Dermochelys coriacea*, indicate behavioural plasticity associated with long-distance migration. *Animal Behaviour 67* 733–743. DOI:10.1016/j.anbehav.2003.08.011

Hayward, B. W., T. Cedhagen, M. Kaminski, and O. Gross. (2016). *World Foraminifera Database*. Retrieved 06/28/2016, from http://www.marinespecies.org/foraminifera/.

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.

Hazen, E. L., S. Jorgensen, R. R. Rykaczewski, S. J. Bograd, D. G. Foley, I. D. Jonsen, S. A. Shaffer, J. P. Dunne, D. P. Costa, L. B. Crowder, and B. A. Block. (2012). Predicted habitat shifts of Pacific top predators in a changing climate. *Nature Climate Change 3* (3): 234–238. DOI:10.1038/nclimate1686

HDR. (2012). *Summary Report: Compilation of Visual Survey Effort and Sightings for Marine Species Monitoring in the Hawaii Range Complex, 2005–2012* (Prepared for Commander, U.S. Pacific Fleet, Pearl Harbor, Hawaii. Submitted to Naval Facilities Engineering Command Pacific

(NAVFAC), EV2 Environmental Planning, Pearl Harbor, Hawaii, 96860-3134, under contract #N62470-10-D-3011, issued to HDR, Inc.). San Diego, CA: HDR Inc.

Heady, W. N., R. Beas-Luna, M. N. Dawson, N. Eddy, K. Elsmore, F. T. Francis, T. Frierson, A. L. Gehman, T. Gotthardt, S. A. Gravem, J. Grebel, S. L. Hamilton, L. Hannah, C. D. Harvell, J. Hodin, I. Kelmartin, C. Krenz, L. Lee, J. Lorda, D. Lowry, S. Mastrup, E. Meyer, P. T. Raimondi, S. S. Rumrill, V. R. Saccomanno, L. M. Schiebelhut, and C. Siddon. (2022). *Roadmap to Recovery for the Sunflower Sea Star (Pycnopodia helianthoides) along the West Coast of North America*. Sacramento, CA: The Nature Conservancy.

Healey, M. C. (1991). *Life History of Chinook Salmon (Oncorhynchus tshawytscha)* (Pacific Salmon Life Histories). Vancouver, B.C.: UBC Press.

Heatwole, H. (1999). *Sea Snakes*. Malabar, FL: Krieger Publishing Company.

Heberholz, J. and B. A. Schmitz. (2001). Signaling via water currents in behavioral interactions of snapping shrimp (*Alpheus heterochaelis*). *Biological Bulletin 201* 6–16.

Hedges, J. I. and J. M. Oades. (1997). Review Paper: Comparative organic geochemistries of soils and marine sediments. *Organic Geochemistry 27* (7–8): 319–361. DOI:S0146-6380(97)00056-9

Heenehan, H. L., J. A. Tyne, L. Bejder, S. M. Van Parijs, and D. W. Johnston. (2016). Passive acoustic monitoring of coastally associated Hawaiian spinner dolphins, *Stenella longirostris*, ground-truthed through visual surveys. *The Journal of the Acoustical Society of America 140* (1): 206. DOI:10.1121/1.4955094

Heenehan, H. L., S. M. Van Parijs, L. Bejder, J. A. Tyne, and D. W. Johnston. (2017). Using acoustics to prioritize management decisions to protect coastal dolphins: A case study using Hawaiian spinner dolphins. *Marine Policy 75* 84–90. DOI:10.1016/j.marpol.2016.10.015

Heironimus, L. B., M. T. Sturza, and S. S. M. (2022). *Tagging Green Sturgeon with Acoustic Transmitters for Evaluation of Habitat Use Along the Washington Coast*. Seattle, WA: Washington Department of Fish and Wildlife.

Heironimus, L. B., M. T. Sturza, and S. S. M. (2023). *Tagging Green Sturgeon with Acoustic Transmitters for Evaluation of Habitat Use Along the Washington Coast - Final Report*. Ridgefield, WA: Washington Department of Fish and Wildlife.

Heironimus, L. B., M. T. Sturza, and S. M. Schade. (2024). *Tagging Green Sturgeon with Acoustic Transmitters for Evaluation of Habitat Use Along the Washington Coast. Final Summary Report*. Ridgefield, WA: Washington Department of Fish and Wildlife.

Heithaus, M. R. and L. M. Dill. (2008). Feeding strategies and tactics. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 1100–1103). Academic Press.

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.

Helker, V. T., M. M. Muto, K. Savage, S. Teerlink, L. A. Jemison, K. Wilkinson, and J. Jannot. (2017). *Human-Caused Mortality and Injury of NMFS-Managed Alaska Marine Mammal Stocks, 2011–2015* (NOAA Technical Memorandum NMFS-AFSC-354). Seattle, WA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center.

Hemery, L. G., S. R. Marion, C. G. Romsos, A. L. Kurapov, and S. K. Henkel1. (2016). Ecological niche and species distribution modelling of sea stars along the Pacific Northwest continental shelf. *Biodiversity Research* (22): 1314-1327.

Hemminga, M. A. and C. M. Duarte. (2000). Seagrasses in the human environment *Seagrass Ecology* (pp. 248–291). Cambridge, United Kingdom: Cambridge University Press.

Henderson, E. E., K. A. Forney, J. P. Barlow, J. A. Hildebrand, A. B. Douglas, J. Calambokidis, and W. J. Sydeman. (2014). Effects of fluctuations in sea-surface temperature on the occurrence of small cetaceans off Southern California. *Fishery Bulletin 112* (2-3): 159–177. DOI:10.7755/fb.112.2-3.5

Henderson, E. E., R. Manzano-Roth, S. W. Martin, and B. Matsuyama. (2015, 13 December 2013). *Behavioral Responses of Beaked Whales to Mid-Frequency Active Sonar on the Pacific Missile Range Facility, Hawaii*. Presented at the Society for Marine Mammalogy 20th Biennial Conference,. Dunedin, New Zealand.

Henderson, J. R. (2001). A pre-and post-MARPOL Annex V summary of Hawaiian monk seal entanglements and marine debris accumulation in the Northwestern Hawaiian Islands, 1982–1998. *Marine Pollution Bulletin 42* (7): 584-589.

Henderson, J. R. (2019). *Hawaiian Monk Seal Research Program Hawaiian monk seal entanglement data collected in*

the Hawaiian Archipelago, 1974-2018. US National Oceanographic Data Center.

Henkel, L. A., E. E. Burkett, and J. Y. Takekawa. (2004). At-sea activity and diving behavior of a radio-tagged Marbled Murrelet in central California. *Waterbirds 27* (1): 9-12.

Henkel, L. A. and J. T. Harvey. (2008). Abundance and distribution of marine mammals in nearshore waters of Monterey Bay, California. *California Fish and Game 94* (1): 1–17.

Henninger, H. P. and W. H. Watson, III. (2005). Mechanisms underlying the production of carapace vibrations and associated waterborne sounds in the American lobster, *Homarus americanus*. *The Journal of Experimental Biology 208* 3421–3429. DOI:10.1242/jeb.01771

Henry, A., J. Moore, J. Carretta, L. Ballance, J. Barlow, P. Fiedler, B. Hancock-Hanser, T. Joyce, and S. Rankin. (2020). *Report on the California Current Ecosystem Survey: Cetacean and Seabird Data Collection Efforts June 26 – December 4, 2018,* (NOAA Technical Memorandum NMFS-SWFSC-636). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Hernández-Camacho, C. J., L. Pelayo-González, and M. P. Rosas-Hernández. (2021). California Sea Lion (Zalophus californianus, Lesson 1828) *Ecology and Conservation of Pinnipeds in Latin America* (pp. 119-143). Springer.

Heron, S., J. Morgan, M. Eakin, and W. Skirving. (2008). Chapter 3: Hurricanes and Their Effects on Coral Reefs. In C. Wilkinson & D. Souter (Eds.), *Status of Caribbean Coral Reefs After Bleaching and Hurricanes in 2005* (pp. 31–36). Townsville, Australia: Global Coral Reef Monitoring Network, and Reef and Rainforest Research Centre.

Hertel, F. and L. Ballance. (1999). Wing ecomorphology of seabirds from Johnston Atoll. *The Condor 101* 549–556.

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.

Heuer, R. M. and M. Grosell. (2014). Physiological impacts of elevated carbon dioxide and ocean acidification on fish. *American Journal of Physiology - Regulatory, Integrative and Comparative Physiology 307* (9): 1061–1084.

Hewson, I., J. B. Button, B. M. Gudenkauf, B. Miner, A. L. Newton, J. K. Gaydos, J. Wynne, C. L. Groves, G. Hendler, M. Murray, S. Fradkin, M. Breitbart, E. Fahsbender, K. D. Lafferty, A. M. Kilpatrick, C. M. Miner, P. Raimondi, L. Lahner, C. S. Friedman, S. Daniels, M. Haulena, J. Marliave, C. A. Burge, M. E. Eisenlord, and C. D. Harvell. (2014). Densovirus associated with sea-star wasting disease and mass mortality. *Proceedings of the National Academy of Sciences 111* (28): 17278–17283.

Heyning, J. E. and W. F. Perrin. (1994). Evidence for two species of common dolphins (Genus *Delphinus*) from the eastern north Pacific. *Contributions in Science 442* 1–35.

Hilborn, R., S. P. Cox, F. M. D. Gulland, D. G. Hankin, N. T. Hobbs, D. E. Schindler, and A. W. Trites. (2012). *The Effects of Salmon Fisheries on Southern Resident Killer Whales: Final Report of the Independent Science Panel*. Seattle, WA: National Marine Fisheries Service.

Hildebrand, J. A., S. Baumann-Pickering, and M. A. McDonald. (2009). *Beaked Whale Presence, Habitat, and Sound Production in the North Pacific* (Unpublished technical report on file).

Hill, J. E., N. J. Robinson, C. M. King, and F. V. Paladino. (2017). Diving behavior and thermal habitats of gravid hawksbill turtles at St. Croix, USA. *Marine Biology 164* (1): 17.

Hill, M. (1998). Spongivory on Caribbean reefs releases corals from competition with sponges. *Oecologia 117* (1–2): 143–150.

Hill, M., A. D. Ligon, M. H. Deakos, U. Adam, E. Norris, and E. M. Oleson. (2011). *Cetacean Surveys of Guam and CNMI Waters: August–September, 2011* (MIRC Survey Report FY2011). Honolulu, HI: Pacific Islands Fisheries Science Center.

Hindell, M. A. and W. F. Perrin. (2009). Elephant seals, *Mirounga angustirostris* and *M. leonina*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 364–368). Cambridge, MA: Academic Press.

Hinke, J. T., G. M. Watters, G. W. Boehlert, and P. Zedonius. (2005). Ocean habitat use in autumn by Chinook salmon in coastal waters of Oregon and California. *Marine Ecological Progress Series 285* 181–192.

Ho, T.-Y., L.-S. Wen, C.-F. You, and D.-C. Lee. (2007). The trace metal composition of size-fractionated plankton in the South China Sea: Biotic versus abiotic sources. *Limnology and Oceanography 52* (5): 1776–1788. DOI:10.4319/lo.2007.52.5.1776

Hoadley, K. D., D. T. Pettay, A. G. Grottoli, W.-J. Cai, T. F. Melman, V. Schoepf, X. Hu, Q. Li, H. Xu, Y. Wang, Y. Matsui, J. H. Baumann, and M. E. Warner. (2015). Physiological response to elevated temperature and pCO_2 varies across four Pacific coral species: Understanding the unique host+symbiont response. *Scientific Reports 5:18371* 1–15. DOI:10.1038/srep18371

Hobday, A. J. and M. J. Tegner. (2000). *Status Review of White Abalone (Haliotis sorenseni) throughout its Range in California and Mexico*. Long Beach, CA: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Region Office.

Hobday, A. J., M. J. Tegner, and P. L. Haaker. (2001). Over-exploitation of a broadcast spawning marine invertebrate: Decline of the white abalone. *Reviews in Fish Biology and Fisheries 10* 493–514.

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.

Hodder, J., R. F. Brown, and C. Cziesla. (1998). The northern elephant seal in Oregon: A pupping range extension and onshore occurrence. *Marine Mammal Science 14* (4): 873–881.

Hoelzel, A. R. (1999). Impact of population bottlenecks on genetic variation and the importance of life-history; a case study of the northern elephant seal. *Biological Journal of the Linnean Society 68* 23–29.

Hoelzel, A. R. (2002). *Marine Mammal Biology: An Evolutionary Approach*. Malden, MA: Blackwell Publishing.

Hoelzel, A. R., J. Hey, M. E. Dahlheim, C. Nicholson, V. Burkanov, and N. Black. (2007). Evolution of population structure in a highly social top predator, the killer whale. *Molecular Biology and Evolution 24* (6): 1407–1415. DOI:10.1093/molbev/msm063

Hoff, R., P. Hensel, E. C. Proffitt, P. Delgado, G. Shigenaka, R. Yender, R. Hoff, and A. J. Mearns. (2002). *Oil Spills in Mangroves: Planning & Response Considerations*. Silver Spring, MD: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Ocean Service, Office of Response and Restoration.

Hohn, A. A., D. Rotstein, C. Harms, and B. Southall. (2006). *Report on Marine Mammal Unusual Mortality Event UMESE0501Sp: Multispecies Mass Stranding of Pilot Whales (Globicephala macrorhynchus), Minke Whale (Balaenoptera acutorostrata) and Dwarf Sperm Whales (Kogia sima) in North Carolina on 15-16 January 2005* (National Oceanic and Atmospheric Administration Technical Memorandum NMFS-SEFSC-537). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Holland, K. T. and P. A. Elmore. (2008). A review of heterogeneous sediments in coastal environments. *Earth-Science Reviews 89* (3–4): 116–134. DOI:10.1016/j.earscirev.2008.03.003

Holloway-Adkins, K. G. (2006). Juvenile green turtles (*Chelonia mydas*) forage on high-energy, shallow reef on the east coast of Florida. In M. Frick, A. Panagopoulou, A. F. Rees, & K. Williams (Eds.), *Twenty-sixth Annual Symposium on Sea Turtle Biology and Conservation: Book of Abstracts*. Athens, Greece: National Marine Fisheries Service Southeast Fisheries Science Center, International Sea Turtle Society.

Holt, M. M., M. B. Hanson, D. A. Giles, C. K. Emmons, and J. T. Hogan. (2017). Noise levels received by endangered killer whales *Orcinus orca* before and after implementation of vessel regulations. *Endangered Species Research 34* 15–26. DOI:10.3354/esr00841

Holt, M. M., D. P. Noren, and C. K. Emmons. (2011). Effects of noise levels and call types on the source levels of killer whale calls. *The Journal of the Acoustical Society of America 130* (5): 3100–3106. DOI:10.1121/1.3641446

Holt, M. M., D. P. Noren, V. Veirs, C. K. Emmons, and S. Veirs. (2008). Speaking up: Killer whales (*Orcinus orca*) increase their call amplitude in response to vessel noise. *The Journal of the Acoustical Society of America 125* (1): EL27–EL32. DOI:10.1121/1.3040028

Hong, B. C., J. J. Curran, M. Hanna, and E. Pollard. (2023a). *Rocky Intertidal Community Monitoring At Naval Auxiliary Landing Field, San Clemente Island, Spring and Fall 2022*. San Diego, CA: Naval Facilities Engineering Systems Command (NAVFAC) Southwest.

Hong, B. C., J. J. Curran, B. P. Saunders, M. E. Hanna, and E. A. Pollard. (2023b). *Rocky Intertidal Community Monitoring at Naval Outlying Field, San Nicolas Island*. San Diego, CA: Naval Facilities Engineering Command (NAVFAC) Southwest.

Hoover, J. P. (1998a). Echinoderms: Phylum Echinodermata. In *Hawai'i's Sea Creatures: A Guide to Hawai'i's Marine Invertebrates* (pp. 290–335). Honolulu, HI: Mutual Publishing.

Hoover, J. P. (1998b). *Hawaii's Sea Creatures A Guide to Hawaii's Marine Invertebrates*. Seoul, Republic of Korea: Mutual Publishing.

Horwood, J. (2009). Sei whale, *Balaenoptera borealis*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 1001–1003). Cambridge, MA: Academic Press.

Horwood, J. W. (1987). *The Sei Whale: Population Biology, Ecology, and Management*. New York, NY: Croom Helm.

Houck, W. J. and T. A. Jefferson. (1999). Dall's Porpoise, *Phocoenoides dalli* (True, 1885). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals Vol 6: The second book of dolphins and porpoises* (pp. 443–472). San Diego, CA: Academic Press.

Houghton, J. D. R., M. J. Callow, and G. C. Hays. (2003). Habitat utilization by juvenile hawksbill turtles (*Eretmochelys imbricata*, Linnaeus, 1766) around a shallow water coral reef. *Journal of Natural History 37* 1269–1280. DOI:10.1080/00222930110104276

Hourigan, T. F., P. J. Etnoyer, and S. D. Cairns. (2017a). Introduction to the State of Deep-Sea Coral and Sponge Ecosystems of the United States. In T. F. Hourigan, P. J. Etnoyer, & S. D. Cairns (Eds.), *The State of Deep-Sea Coral and Sponge Ecosystems of the United States* (Vol. National Oceanic and Atmospheric Administration Technical Memorandum NMFS-OHC-3, pp. 1–34). Silver Spring, MD: National Oceanic and Atmospheric Administration.

Hourigan, T. F., J. Reed, S. Pomponi, S. W. Ross, A. W. David, and S. Harter. (2017b). State of the Deep-Sea Coral and Sponge Ecosystems off the Southeast United States *The State of Deep-Sea Coral and Sponge Ecosystems in the United States* (pp. 13-11–13-61). Silver Spring, MD: National Oceanic and Atmospheric Administration.

Houser, D. S., S. W. Martin, and J. J. Finneran. (2013). Behavioral responses of California sea lions to mid-frequency (3250-3450 Hz) sonar signals. *Marine Environmental Research 92* 268–278. DOI:10.1016/j.marenvres.2013.10.007

Hu, M. Y., H. Y. Yan, W. S. Chung, J. C. Shiao, and P. P. Hwang. (2009). Acoustically evoked potentials in two cephalopods inferred using the auditory brainstem response (ABR) approach. *Comparative Biochemistry and Physiology, Part A 153* 278–283.

Huber, E. R., R. E. Ryan, R. C. Johnson, A. M. Sturrock, R. A. Lusardi, and S. M. Carlson. (2024). Seventy years of diminishing biocomplexity of California Central Valley hatchery steelhead, *Oncorhynchus mykiss*. *Canadian Journal of Fisheries and Aquatic Sciences 81* (4): 355-367.

Huff, D. D., S. T. Lindley, P. S. Rankin, and E. A. Mora. (2011). Green sturgeon physical habitat use in the coastal Pacific Ocean. *PLoS ONE 6* (9): e25156.

Huff, D. D., S. T. Lindley, B. K. Wells, and F. Chai. (2012). Green sturgeon distribution in the Pacific Ocean estimated from modeled oceanographic features and migration behavior. *PLoS ONE 7* (9): e45852.

Hughes, T. P., A. H. Baird, D. R. Bellwood, M. Card, S. R. Connolly, C. Folke, R. Grosberg, O. Hoegh-Guldberg, J. B. C. Jackson, J. Kleypas, J. M. Lough, P. A. Marshall, M. Nystrom, S. R. Palumbi, J. M. Pandolfi, B. Rosen, and J. Roughgarden. (2003). Climate change, human impacts, and the resilience of coral reefs. *Science 301* (5635): 929–933.

Hughes, T. P., M. J. Rodrigues, D. R. Bellwood, D. Ceccarelli, O. Hoegh-Guldberg, L. McCook, N. Moltschaniwskyj, M. S. Pratchett, R. S. Steneck, and B. Willis. (2007). Phase shifts, herbivory, and the resilience of coral reefs to climate change. *Current Biology 17* (4): 360–365. DOI:10.1016/j.cub.2006.12.049

Hui, C. A. (1985). Undersea topography and the comparative distribution of two pelagic cetaceans. *Fishery Bulletin 83* (3): 472–475.

Humber, F., B. J. Godley, and A. C. Broderick. (2014). So excellent a fishe: A global overview of legal marine turtle fisheries. *Diversity and Distributions 20* (5): 579–590. DOI:10.1111/ddi.12183

Hunt, G. L., Jr. and J. L. Butler. (1980). Reproductive ecology of Western gulls and Xantus' murrelets with respect to food resources in the Southern California Bight. *CalCOFI Reports XXI* 62–67.

Huntingford, F., C. Adams, V. A. Braithwaite, S. Kadri, T. G. Pottinger, P. Sandoe, and J. F. Turnball. (2006). Review paper: Current issues in fish welfare. *Journal of Fish Biology 70* (4): 1311–1316.

Hutchinson, M., M. Scott, R. Bauer, J. Anderson, D. M. Coffey, K. Holland, C. Meyer, J. O'Sullivan, and M. Royer. (2023). Habitat use and movement patterns of adult male and juvenile scalloped hammerhead sharks *Sphyrna lewini* throughout the Hawaiian archipelago. *Endangered Species Research 52* 41–64. DOI:https://doi.org/10.3354/esr01267

Hwang, A., R. H. Defran, M. Bearzi, D. Maldini, C. A. Saylan, A. R. Lang, K. J. Dudzik, O. R. Guzon-Zatarain, D. L. Kelly, and D. W. Weller. (2014). Coastal Range and Movements of Common Bottlenose Dolphins off California and Baja California, Mexico. *Bulletin of the Southern California Academy of Science 113* (1): 1–13.

Ilyashenko, V. and K. Zharikov. (2014). *Aboriginal Harvest of Gray and Bowhead Whales in the Russian Federation In 2013*

Ingvarsdottir, A., C. Bjorkblom, E. Ravagnan, B. F. Godal, M. Arnberg, D. L. Joachim, and S. Sanni. (2012). Effects of different concentrations of crude oil on first feeding larvae of Atlantic herring (*Clupea harengus*). *Journal of Marine Systems 93* 69–76. DOI:10.1016/j.jmarsys.2011.10.014

Inman, M. (2005). Fish moved by warming waters. *Science 308* 937.

INP. (2006). *Sustentabilidad y Pesca Responsable en México*. Instituto Nacional de la Pesca.

Intergovernmental Panel on Climate Change. (2014). *IPCC Fifth Assessment Report*. Geneva, Switzerland: Intergovernmental Panel on Climate Change.

International Union for Conservation of Nature. (2011). *IUCN Red List of Threatened Species. Version 2011.2*. Retrieved from http://www.iucnredlist.org.

International Union for Conservation of Nature. (2017). *Leporillus conditor*. Retrieved May 15, 2017, from http://www.iucnredlist.org/details/11634/0.

International Union for Conservation of Nature and Natural Resources. (2010a). *The IUCN Red List of Threatened Species Version 2010.1*. Retrieved from http://www.iucnredlist.org/.

International Union for Conservation of Nature and Natural Resources. (2010b). *IUCN Red List of Threatened Species. Version 2010.4*. Retrieved from www.iucnredlist.org.

International Union for Conservation of Nature and Natural Resources. (2010c). *Phoebastria albatrus. International Union for Conservation of Nature 2010. International Union for Conservation of Nature Red List of Threatened Species. Version 2010.3.* Retrieved from http://www.iucnredlist.org.

International Union for the Conservation of Nature and Natural Resources. (2010). *Oceanodroma castro. IUCN Red List of Threatened Species. Version 2010.3*. Retrieved from http://www.iucnredlist.org/.

International Whaling Commission. (2014). *Report of the Workshop on the Rangewide Review of the Population Structure and Status of North Pacific Gray Whales* [Type]. Presented at the 14th Meeting of the Western Gray Whale Advisory Panel.

Irvine, L. M., B. R. Mate, M. H. Winsor, D. M. Palacios, S. J. Bograd, D. P. Costa, and H. Bailey. (2014). Spatial and temporal occurrence of blue whales off the U.S. west coast, with implications for management. *PLoS ONE 9* (7): e102959. DOI:10.1371/journal.pone.0102959

Israel, J. A., K. J. Bando, E. C. Anderson, and B. May. (2009). Polyploid microsatellite data reveal stock complexity among estuarine North American green sturgeon (*Acipenser medirostris*). *Canadian Journal of Fish Aquatic Science 66* 1491–1504. DOI:10.1139/F09-091

Jackson, J. B. C., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, J. M. Erlandson, J. A. Estes, T. P. Hughes, S. Kidwell, C. B. Lange, H. S. Lenihan, J. M. Pandolfi, C. H. Peterson, R. S. Steneck, M. J. Tegner, and R. R. Warner. (2001). Historical overfishing and the recent collapse of coastal ecosystems. *Science 293* 629–638.

Jacobson, S., J. Marshall, D. Dalrymple, F. Kawasaki, D. Pearse, A. Abadía-Cardoso, and J. C. Garza. (2014). *Genetic Analysis of Trout (Oncorhynchus mykiss) in Southern California Coastal Rivers and Streams*: NOAA Southwest Fisheries Science Center and University of California Santa Cruz.

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.

James, M. C., R. A. Myers, and C. A. Ottensmeyer. (2005). Behaviour of leatherback sea turtles, *Dermochelys coriacea*, during the migratory cycle. *Proceedings of the Royal Society B: Biological Sciences 272* 1547–1555. DOI:10.1098/rspb.2005.3110

Jannot, J. E., K. A. Somers, V. Tuttle, J. McVeigh, J. V. Carretta, and V. Helker. (2018). *Observed and Estimated Marine Mammal Bycatch in US West Coast Groundfish Fisheries, 2002–16* (NWFSC Processed Report 2018-03). Seattle, WA: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center.

Jefferson, T. A. (1991). Observations on the distribution and behavior of Dall's porpoise (*Phocoenoides dalli*) in Monterey Bay, California. *Aquatic Mammals 17* (1): 12–19.

Jefferson, T. A., D. Fertl, M. Michael, and T. D. Fagin. (2006). An unusual encounter with a mixed school of melon-headed whales (*Peponocephala electra*) and rough-toothed dolphins (*Steno bredanensis*) at Rota, Northern Mariana Islands. *Micronesica 38* (2): 23–244.

Jefferson, T. A., S. Leatherwood, and M. A. Webber. (1993). *Marine mammals of the world: FAO species identification guide*. Rome, Italy: Food and Agriculture Organization of the United Nations.

Jefferson, T. A. and S. K. Lynn. (1994). Marine mammal sightings in the Gulf of Mexico and Caribbean Sea, Summer 1991. *Caribbean Journal of Science 30* (1–2): 83–89.

Jefferson, T. A. and A. Schulman-Janiger. (2018). Investigating the disappearance of short-finned pilot whales (*Globicephala macrorhynchus*) from Southern California: Did fisheries play a role? *Bulletin of the Southern California Academy of Sciences 117* (1): 29–51.

Jefferson, T. A., M. A. Smultea, and C. A. Bacon. (2014). Southern California Bight marine mammal density and abundance from aerial surveys, 2008–2013. *Journal of Marine Animals and Their Ecology 7* (2): 14–30.

Jefferson, T. A., M. A. Smultea, S. S. Courbis, and G. S. Campbell. (2016). Harbor porpoise (*Phocoena phocoena*) recovery in the inland waters of Washington: Estimates of density and abundance from aerial surveys, 2013–2015. *Canadian Journal of Zoology 94* (7): 505–515. DOI:10.1139/cjz-2015-0236

Jefferson, T. A. and K. Van Waerebeek. (2002). The taxonomic status of the nominal dolphin species *Delphinus tropicalis* van Bree, 1971. *Marine Mammal Science 18* (4): 787–818.

Jefferson, T. A., M. A. Webber, and R. L. Pitman. (2008). *Marine Mammals of the World: A Comprehensive Guide to their Identification*. London, UK: Elsevier.

Jefferson, T. A., M. A. Webber, and R. L. Pitman. (2015). *Marine Mammals of the World: A Comprehensive Guide to Their Identification* (2nd ed.). Cambridge, MA: Academic Press.

Jeffries, S. (2014). *Aerial Surveys of Pinniped Haulout Sites in Pacific Northwest Inland Waters. Final Report* (Prepared for Commander, U.S. Pacific Fleet, Pearl Harbor, Hawaii). Silverdale, WA: Naval Facilities Engineering Command (NAVFAC) Northwest.

Jeffs, A., N. Tolimieri, and J. C. Montgomery. (2003). Crabs on cue for the coast: The use of underwater sound for orientation by pelagic crab stages. *Marine Freshwater Resources 54* 841–845.

Jemison, L. A., G. W. Pendleton, L. W. Fritz, K. K. Hastings, J. M. Maniscalco, A. W. Trites, and T. S. Gelatt. (2013). Inter-population movements of Steller sea lions in Alaska with implications for population separation. *PLoS ONE 8* (8): e70167.

Jensen, M. P., C. D. Allen, T. Eguchi, I. P. Bell, E. L. LaCasella, W. A. Hilton, C. A. M. Hof, and P. H. Dutton. (2018). Environmental warming and feminization of one of the largest sea turtle populations in the world. *Current Biology 28* (1): 154–159. DOI:https://doi.org/10.1016/j.cub.2017.11.057

Jepson, P. D., R. Deaville, J. L. Barber, A. Aguilar, A. Borrell, S. Murphy, J. Barry, A. Brownlow, J. Barnett, S. Berrow, A. A. Cunningham, N. J. Davison, M. Ten Doeschate, R. Esteban, M. Ferreira, A. D. Foote, T. Genov, J. Gimenez, J. Loveridge, A. Llavona, V. Martin, D. L. Maxwell, A. Papachlimitzou, R. Penrose, M. W. Perkins, B. Smith, R. de Stephanis, N. Tregenza, P. Verborgh, A. Fernandez, and R. J. Law. (2016). PCB pollution continues to impact populations of orcas and other dolphins in European waters. *Scientific Reports 6* 18573. DOI:10.1038/srep18573

Jessup, D. A., M. A. Miller, J. P. Ryan, H. M. Nevins, H. A. Kerkering, A. Mekebri, D. B. Crane, T. A. Johnson, and R. M. Kudela. (2009). Mass stranding of marine birds caused by a surfactant-producing red tide. *PLoS ONE 4* (2): e4550. DOI:10.1371/journal.pone.0004550

Jiménez, S., A. Domingo, M. Abreu, and A. Brazeiro. (2012). Bycatch susceptibility in pelagic longline fisheries: Are albatrosses affected by the diving behaviour of medium-sized petrels? *Aquatic Conservation: Marine and Freshwater Ecosystems 22* (4): 436–445. DOI:10.1002/aqc.2242

Jodice, P. G. R. and M. W. Collopy. (1999). *Diving and foraging patt*erns of Marbled Murrelets (Brachyramphus Marmoratus): Testing predicitions from optimal-breathing models. *Canadian Journal of Zoology 77* (9): 1049–1418.

Johanos, T. C., A. L. Harting, T. A. Wurth, and J. D. Baker. (2014). Range-wide movement patterns of Hawaiian monk seals. *Marine Mammal Science 30* (3): 1165–1174. DOI:10.1111/mms.12084

Johansen, S., O. N. Larsen, J. Christensen-Dalsgaard, L. Seidelin, T. Huulvej, K. Jensen, S. G. Lunneryd, M. Bostrom, and M. Wahlberg. (2016). In-air and underwater hearing in the great cormorant (*Phalacrocorax carbo sinensis*). *Advances in Experimental Medicine and Biology 875* 505–512. DOI:10.1007/978-1-4939-2981-8_61

Johnson, J. E., V. Allain, B. Basel, J. D. Bell, A. Chin, L. X. Dutra, E. Hooper, D. Loubser, J. Lough, B. R. Moore, and S. Nicol. (2020). Impacts of climate change on marine resources in the Pacific Island region. In J. Dodson (Ed.), *Climate Change and Impacts in the Pacific* (pp. 359-402). Cham, Switzerland: Springer Climate.

Johnston, D. W., M. E. Chapla, L. E. Williams, and D. K. Matthila. (2007). Identification of humpback whale Megaptera novaeangliae wintering habitat in the Northwestern Hawaiian Islands using spatial habitat modeling. *Endangered Species Research*. DOI:10.3354/esr00049

Johnstone, R. A. and R. Bshary. (2004). Evolution of spite through indirect reciprocity. *Proceeding of the Royal Society B 271* 1917–1922.

Jokiel, P. L. (2008). Biology and Ecological Functioning of Coral Reefs in the Main Hawaiian Islands. In B. Riegl & R. E. Dodge (Eds.), *Coral Reefs of the USA*. Berlin, Germany: Springer Science.

Jokiel, P. L. and E. K. Borwn. (2004). Global warming, regional trends and inshore environmental conditions influence coral bleaching in Hawaii. *Global Change Biology 10* 1627–1641. DOI:10.1111/j.1365-2486.2004.00836

Jones, C. L., T. W. Anderson, and M. S. Edwards. (2013). Evaluating eelgrass site quality by the settlement, performance, and survival of a marine fish. *Journal of Experimental Marine Biology and Ecology 445* 61–68. DOI:10.1016/j.jembe.2013.04.003

Jones, M. L. and S. L. Swartz. (2009). Gray whale, *Eschrichtius robustus*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 503–511). Cambridge, MA: Academic Press.

Julian, F. and M. Beeson. (1998). Estimates of marine mammal, turtle, and seabird mortality for two California gillnet fisheries: 1990–1995. *Fishery Bulletin 96* (2): 271–284.

Jumars, P. A., K. M. Dorgan, and S. M. Lindsay. (2015). Diet of worms emended: An update of polychaete feeding guilds. *Annual Review of Marine Science 7* 497–520. DOI:10.1146/annurev-marine-010814-020007

Júnior, J. C. R., J. B. Pfaller, R. Corbetta, and L. Veríssimo. (2015). Parasitic isopods associated with sea turtles nesting in Brazil. *Journal of the Marine Biological Association of the United Kingdom 95* (5): 973–981.

Kahng, S. E. (2006). *Ecology and Ecological Impact of an Alien Octocoral, Carijoa riisei, in Hawai'i.* (Doctoral dissertation). University of Hawaii at Manoa, Honolulu, HI.

Kahng, S. E. and R. W. Grigg. (2005). Impact of an alien octocoral, *Carijoa riisei*, on black corals in Hawaii. *Coral Reefs 24* 556–562. DOI:10.1007/s00338-005-0026-0

Kahng, S. E. and J. E. Maragos. (2006). The deepest, zooxanthellate scleractinian corals in the world? *Coral Reefs 25* 254. DOI:10.1007/s00338-006-0098-5

Kaifu, K., T. Akamatsu, and S. Segawa. (2008). Underwater sound detection by cephalopod statocyst. *Fisheries Science 74* 781–786. DOI:10.1111/j.1444-2906.2008.01589

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

Kamezaki, N., Y. Matsuzawa, O. Abe, H. Asakawa, T. Fujii, K. Goto, S. Hagino, M. Hayami, M. Ishii, T. Iwamoto, T. Kamata, H. Kato, J. Kodama, Y. Kondo, I. Miyawaki, K. Mizobuchi, Y. Nakamura, Y. Nakashima, H. Naruse, K. Omuta, M. Samejima, H. Suganuma, H. Takeshita, T. Tanaka, T. Toji, M. Uematsu, A. Yamamoto, T. Yamato, and I. Wakabayashi. (2003). Loggerhead turtles nesting in Japan. In A. B. Bolten & B. E. Witherington (Eds.), *In Loggerhead Sea Turtles* (pp. 210–217). Washington, DC: Smithsonian Books.

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

Kaplanis, N. J., J. Harris, and J. Smith. (2016). Distribution patterns of the non-native seaweeds *Sargassum horneri* (Turner) C. Agardh and *Undaria pinnatifida* (Harvey) Suringar on the San Diego and Pacific coast of North America. *Aquatic Invasions 11* (2): 111–124. DOI:10.3391/ai.2016.11.2.01

Kappel, C. V. (2005). Losing pieces of the puzzle: Threats to marine, estuarine, and diadromous species. *Frontiers in Ecology and the Environment 3* (5): 275–282.

Karleskint, G., Jr. , R. Turner, and J. W. Small, Jr. (2006). *Introduction to Marine Biology* (2nd ed.). Belmont, CA: Thomson Brooks/Cole.

Kashiwagi, T., A. D. Marshall, M. B. Bennett, and J. R. Ovenden. (2011). Habitat segregation and mosaic sympatry of the two species of manta ray in the Indian and Pacific Oceans: *Manta alfredi* and *M. birostris*. *Marine Biodiversity Records 4*. DOI:10.1017/S1755267211000479

Kasuya, T. (2009). Giant beaked whales, *Berardius bairdii* and *B. arnuxii*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 498–500). Amsterdam, Netherlands: Academic Press.

Kasuya, T. and T. Miyashita. (1997). Distribution of Baird's beaked whales off Japan. *Reports of the International Whaling Commission 47* 963–968.

Kauai Endangered Seabird Recovery Project. (2017). *Band-rumped Storm-Petrel*. Retrieved May 31, 2017, from https://kauaiseabirdproject.org/band-rumped-storm-petrel/.

Kauparinen, A. and J. Merila. (2007). Detecting and managing fisheries-induced evolution. *Trends in Ecology & Evolution 22* (12): 652–659. DOI:10.1016/j.tree.2007.08.11

Keller, A. A., L. Ciannelli, W. W. Wakefield, V. Simon, J. A. Barth, and S. D. Pierce. (2015a). Occurrence of demersal fishes in relation to near-bottom oxygen levels within the California Current large marine ecosystem. *Fisheries Oceanography 24* (2): 162–176. DOI:10.1111/fog.12100

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

Keller, A. A., W. W. Wakefield, V. Simon, J. A. Barth, and S. D. Pierce. (2015b). Occurrence of demersal fishes in relation to near-bottom oxygen levels within the California Current large marine ecosystem. *Fisheries Oceanography 24* (2): 162–176.

Kemp, N. J. (1996). Habitat loss and degradation. In M. P. Simmonds & J. D. Hutchinson (Eds.), *The Conservation of Whales and Dolphins* (pp. 263–280). New York, NY: John Wiley & Sons.

Kendall-Bar, J. M., D. W. Weller, H. Fearnbach, S. Shane, G. S. Schorr, E. A. Falcone, J. Calambokidis, A. Schulman-Janiger, and J. Barlow. (2016). Movement and Occurrence Patterns of Short-Finned Pilot Whales (*Globicephala macrorhynchus*) in the Eastern North Pacific. *Aquatic Mammals 42* (3): 300–305. DOI:10.1578/AM.42.3.2016.300

Kenner, M. and J. L. Yee. (2022). *Black Abalone surveys at Naval Base Ventura County, San Nicolas Island, California—2021 Annual Report* (Open-File Report 2022–1107). Reston, VA: U.S. Geological Survey.

Kenner, M. C. and J. A. Tomoleoni. (2021). *Kelp Forest Monitoring at Naval Base Ventura County, San Nicolas Island, California: Fall 2019, Sixth Annual Report* (Open File Report 2021-1081). Reston, VA: U.S. Geological Survey.

Kennett, J. P. (1982). *Marine Geology*. New York, NY: Prentice-Hall.

Kenworthy, W. J., M. J. Durako, S. M. R. Fatemy, H. Valavi, and G. W. Thayer. (1993). Ecology of seagrasses in northeastern Saudi Arabia one year after the Gulf War oil spill. *Marine Pollution Bulletin 27* 213–222. DOI:10.1016/0025-326x(93)90027-h

Kerosky, S. M., A. Širović, L. K. Roche, S. Baumann-Pickering, S. M. Wiggins, and J. A. Hildebrand. (2012). Bryde's whale seasonal range expansion and increasing presence in the Southern California Bight from 2000 to 2010. *Deep Sea Research Part I: Oceanographic Research Papers 65* 125–132. DOI:10.1016/j.dsr.2012.03.013

Killam, D. and B. Mache. (2018). *Salmonid Populations of the Upper Sacramento River Basin In 2017*. Red Bluff, CA: California Department of Fish and Wildlife, Pacific States Marine Fisheries Commission.

Kiyota, M., N. Baba, and M. Mouri. (1992). Occurrence of an elephant seal in Japan. *Marine Mammal Science 8* (4): 433.

Klinck, H., S. L. Nieukirk, S. Fregosi, D. K. Mellinger, S. Lastuka, G. B. Shilling, and J. C. Luby. (2015). *Cetacean Studies on the Hawaii Range Complex in December 2014–January 2015: Passive Acoustic Monitoring of Marine Mammals using Gliders. Final Report* (Prepared for Naval Facilities Engineering Command Pacific under HDR Environmental, Operations and Construction, Inc. Contract No. N62470-10-D-3011, CTO KB25). Honolulu, HI: HDR Inc.

Knopf, F. L. and R. M. Evans. (2004). American White Pelican (*Pelecanus erythrorhynchos*). *The Birds of North America Online 57* 6. DOI:10.2173/bna.57

Kohler, N. E. and P. A. Turner. (2001). Shark tagging: A review of conventional methods and studies. *Environmental Biology of Fishes 60* (1–3): 191–223. DOI:10.1023/a:1007679303082

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

Konar, B., T. J. Mitchell, K. Iken, H. Coletti, T. Dean, D. Esler, M. Lindeberg, B. Pister, and B. Weitzman. (2019). Wasting disease and static environmental variables drive sea star assemblages in the

Northern Gulf of Alaska. *Journal of Experimental Marine Biology and Ecology* (520): 151209. DOI:10.1016/j.jembe.2019.151209

Koski, W. R., J. W. Lawson, D. H. Thomson, and W. J. Richardson. (1998). *Point Mugu Sea Range Marine Mammal Technical Report*. San Diego, CA: Naval Air Warfare Center, Weapons Division and Southwest Division, Naval Facilities Engineering Command.

Koslow, J. A. (1996). Energetic and life-history patterns of deep-sea benthic, benthopelagic and seamount-associated fish. *Journal of Fish Biology 49* (Supplement A): 54–74.

Krahn, M. M., M. B. Hanson, R. W. Baird, R. H. Boyer, D. G. Burrows, C. K. Emmons, J. K. Ford, L. L. Jones, D. P. Noren, P. S. Ross, G. S. Schorr, and T. K. Collier. (2007). Persistent organic pollutants and stable isotopes in biopsy samples (2004/2006) from Southern Resident killer whales. *Marine Pollution Bulletin 54* (12): 1903–1911. DOI:10.1016/j.marpolbul.2007.08.015

Kratofil, M. A., A. E. Harnish, S. D. Mahaffy, E. E. Henderson, A. L. Bradford, S. W. Martin, B. A. Lagerquist, D. M. Palacios, E. M. Oleson, and R. W. Baird. (2023). Biologically Important Areas II for cetaceans within U.S. and adjacent waters – Hawaiʻi Region. *Frontiers in Marine Science 10*. DOI:10.3389/fmars.2023.1053581

Kroeker, K. J., R. L. Kordas, R. Crim, I. E. Hendriks, L. Ramajo, G. S. Singh, C. M. Duarte, and J.-P. Gattuso. (2013). Impacts of ocean acidification on marine organisms: Quantifying sensitivities and interaction with warming. *Global Change Biology 19* 1884–1896.

Kruse, S., D. K. Caldwell, and M. C. Caldwell. (1999). Risso's dolphin, *Grampus griseus* (G. Cuvier, 1812). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 6, pp. 183–212). San Diego, CA: Academic Press.

Kszos, L. A., J. J. Beauchamp, and A. J. Stewart. (2003). Toxicity of lithium to three freshwater organisms and the antagonistic effect of sodium. *Ecotoxicology 12* (5): 427–437.

Kudela, R., W. Cochlan, and A. Roberts. (2004). Spatial and temporal patterns of *Pseudo-nitzschia* spp. in central California related to regional oceanography. In K. A. Steidinger, J. H. Landsberg, C. R. Tomas, & G. A. Vargo (Eds.), *Harmful Algae 2002* (pp. 347–349). St. Petersburg, FL: Florida Fish and Wildlife Conservation Commission, Florida Institute of Oceanography, and Intergovernmental Oceanographic Commission of UNESCO.

Kudela, R. M. and W. P. Cochlan. (2000). Nitrogen and carbon uptake kinetics and the influence of irradiance for a red tide bloom off southern California. *Aquatic Microbial Ecology 21* 31–47.

Kueffer, C., C. C. Daehler, C. W. Torres-Santana, C. Lavergne, J. Y. Meyer, R. Otto, and L. Silva. (2010). A global comparison of plant invasions on oceanic islands. *Perspectives in Plant Ecology, Evolution and Systematics 12* (2): 145–161.

Kuhn, C. E. and D. P. Costa. (2014). Interannual variation in the at-sea behavior of California sea lions (*Zalophus californianus*). *Marine Mammal Science 30* (4): 1297–1319. DOI:10.1111/mms.12110

Kuker, K. J., J. A. Thomson, and U. Tscherter. (2005). Novel surface feeding tactics of minke whales, *Balaenoptera acutorostrata*, in the Saguenay-St. Lawrence National Marine Park. *Canadian Field-Naturalist 119* (2): 214–218.

Kuschke, S., J. Wyneken, C. Cray, E. Turla, M. Kinsella, and D. Miller. (2023). Fusarium spp. an emerging fungal threat to leatherback (*Dermochelys coriacea*) eggs and neonates. *Frontiers in Marine Science 10* 1–12. DOI:10.3389/fmars.2023.1170376

Laake, J. L., M. S. Lowry, R. L. DeLong, S. R. Melin, and J. V. Carretta. (2018). Population Growth and Status of California Sea Lions. *Journal of Wildlife Management 82* (3): 583–595. DOI:10.1002/jwmg.21405

Lafferty, K. D. and M. T. Tinker. (2014). Sea otters are recolonizing southern California in fits and starts. *Ecosphere 5* (5): 1-11.

Laffoley, D. and G. Grimsditch. (2009). Introduction. In D. Laffoley & G. Grimsditch (Eds.), *The Management of Natural Coastal Carbon Sinks* (pp. 1–3). Gland, Switzerland: International Union for the Conservation of Nature.

Lam, T., Lingxu, S. Takahashi, and E. A. Burgess. (2011). *Market Forces: An Examination of Marine Turtle Trade in China and Japan*. Hong Kong, China: TRAFFIC East Asia.

Lambert, P. (2000). *Sea Stars of British Columbia, Southeast Alaska and Puget Sound* (2nd ed.). Victoria, Canada: The Royal British Columbia Museum; 2nd edition.

Lammers, M. O. (2004). Occurence and behavior of Hawaiian spinner dolphins (*Stenella longirostris*) along Oahu's leeward and south shores. *Aquatic Mammals 30* (2): 237–250.

Lammers, M. O., L. M. Munger, J. N. Oswald, and T. M. Yack. (2015). *Passive Acoustic Monitoring of Cetaceans in the Hawaii Range Complex Using Ecological Acoustic Recorders (EARs)*. Pearl Harbor, HI: U.S Navy Pacific Fleet.

Larsen, O. N., M. Wahlberg, and J. Christensen-Dalsgaard. (2020). Amphibious hearing in a diving bird, the great cormorant (*Phalacrocorax carbo sinensis*). *Journal of Experimental Biology 223* (6). DOI:10.1242/jeb.217265

Latha, G., S. Senthilvadivu, R. Venkatesan, and V. Rajendran. (2005). Sound of shallow and deep water lobsters: Measurements, analysis, and characterization. *The Journal of the Acoustical Society of America 117* (5): 2720–2723. DOI:10.1121/1.1893525

Law, K. L., S. Moret-Ferguson, N. A. Maximenko, G. Proskurowski, E. E. Peacock, J. Hafner, and C. M. Reddy. (2010). Plastic accumulation in the North Atlantic Subtropical Gyre. *Sciencexpress 329* 1–8. DOI:10.1126/science.1192321

Law, K. L., S. E. Moret-Ferguson, D. S. Goodwin, E. R. Zettler, E. Deforce, T. Kukulka, and G. Proskurowski. (2014). Distribution of surface plastic debris in the eastern Pacific Ocean from an 11-year data set. *Environmental Science & Technology 48* (9): 4732–4738. DOI:10.1021/es4053076

Law, K. L. and R. C. Thompson. (2014). Microplastics in the Sea. *Science 345* (6193): 144–145. DOI:10.1002/2014EF000240/

Le Boeuf, B. J., D. E. Crocker, D. P. Costa, S. B. Blackwell, P. M. Webb, and D. S. Houser. (2000). Foraging ecology of northern elephant seals. *Ecological Monographs 70* (3): 353–382.

Le Boeuf, B. J. and R. M. Laws. (1994). *Elephant seals: an introduction to the genus*: University of California Press: Berkeley/Los Angeles, CA.

Le Boeuf, B. J., P. A. Morris, S. B. Blackwell, D. E. Crocker, and D. P. Costa. (1996). Diving behavior of juvenile northern elephant seals. *Canadian Journal of Zoology 74* 1632–1644.

Leal, M. C., J. Puga, J. Serodio, N. C. Gomes, and R. Calado. (2012). Trends in the discovery of new marine natural products from invertebrates over the last two decades – where and what are we bioprospecting? *PLoS ONE 7* (1): e30580. DOI:10.1371/journal.pone.0030580

Leatherwood, S., R. R. Reeves, A. E. Bowles, B. S. Stewart, and K. R. Goodrich. (1984). Distribution, seasonal movements and abundance of Pacific white-sided dolphins in the eastern North Pacific. *The Scientific Reports of the Whales Research Institute 35* 129–157.

Leet, W. S., C. M. Dewees, R. Klingbeil, and E. J. Larson. (2001). *California's Living Marine Resources: A Status Report*. Sacramento, CA: California Department of Fish and Game.

Lefebvre, K. A., L. Quakenbush, E. Frame, K. B. Huntington, G. Sheffield, R. Stimmelmayr, A. Bryan, P. Kendrick, H. Ziel, T. Goldstein, J. A. Snyder, T. Gelatt, F. Gulland, b. Dickerson, and V. Gill. (2016). Prevalence of algal toxins in Alaskan marine mammals foraging in a changing arctic and subarctic environment. *Harmful Algae 55* (2016): 13–24.

Lehmköster, J. (2015). Climate change impacts on marine ecosystems. In T. Schröder (Ed.), *World Ocean Review* (pp. 102–117). Hamburg, Germany: Mareverlag Publishing House.

Leon, Y. M. and K. A. Bjorndal. (2002). Selective feeding in the hawksbill turtle, an important predator in coral reef ecosystems. *Marine Ecology Progress Series 245* 249–258.

Leslie, M. S., A. Batibasaga, D. S. Weber, D. Olson, and H. C. Rosenbaum. (2005). First record of Blainville's beaked whale, *Mesoplodon densirostris,* in Fiji. *Pacific Conservation Biology 11* (4): 302–304.

Levinton, J. S. (2009a). *Marine Biology: Function, Biodiversity, Ecology* (3rd ed.). New York, NY: Oxford University Press.

Levinton, J. S. (2009b). *Marine Biology: Function, Biodiversity, Ecology* (3rd ed.). New York, NY: Oxford University Press.

Levinton, J. S. (2013). Plankton *Marine Biology: Function, Biodiversity, Ecology* (4th ed., pp. 141–162). New York, NY: Oxford University Press.

Lewison, R. L., L. B. Crowder, B. P. Wallace, J. E. Moore, T. Cox, R. Zydelis, S. McDonald, A. DiMatteo, D. C. Dunn, C. Y. Kot, R. Bjorkland, S. Kelez, C. Soykan, K. R. Stewart, M. Sims, A. Boustany, A. J. Read, P. Halpin, W. J. Nichols, and C. Safina. (2014). Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. *PNAS 111* (14): 5271–5276.

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

Libes, S. M. (2009). *Introduction to Marine Biogeochemistry* (2nd ed.). Burlington, MA, San Diego, CA, and London, United Kingdom: Academic Press.

Lillis, A., D. B. Eggleston, and D. R. Bohnenstiehl. (2013). Oyster larvae settle in response to habitat-associated underwater sounds. *PLoS ONE 8* (10): e79337. DOI:10.1371/journal.pone.0079337

Lillywhite, H. B., C. M. S. Iii, F. Brischoux, and J. B. Pfaller. (2015). On the abundance of a pelagic sea snake. *Journal of Herpetology 49* (2): 184–189.

Lillywhite, H. B., C. M. Sheehy, F. Brischoux, and A. Grech. (2014). Pelagic sea snakes dehydrate at sea. *Proceedings of the Royal Society of London B: Biological Sciences 281* (1782).

Limpus, C. J. (1992). The hawksbill turtle, *Eretmochelys imbricata*, in Queensland: Population structure within a southern Great Barrier Reef ground. *Wildlife Research 19* 489–506.

Limpus, C. J., I. Bell, and J. D. Miller. (2009). Mixed stocks of green turtles foraging on Clack Reef, northern Great Barrier Reef identified from long term tagging studies. *Marine Turtle Newsletter 123* 3–5.

Lindley, S., R. S. Schick, A. Agrawal, M. Goslin, T. E. Pearson, E. Mora, J. Anderson, B. May, S. Greene, C. Hanson, A. Low, D. McEwan, R. Macfarlane, C. Swanson, and J. G. Williams. (2006). Historical population structure of Central Valley steelhead and its alteration by dams. *San Francisco Estuary and Watershed Science 4* (1). DOI:10.15447/sfews.2006v4iss1art3

Lindley, S. T., M. L. Moser, D. L. Erickson, M. Belchik, D. W. Welch, E. L. Rechisky, J. T. Kelly, J. Heublein, and A. P. Kimley. (2008). Marine Migration of North American Green Sturgeon. *Transactions of the American Fisheries Society 137* 182–194.

Lindley, S. T., R. Schick, B. P. May, J. J. Anderson, S. Greene, C. E. Hanson, A. Low, D. McEwan, R. B. MacFarlane, C. Swanson, and J. G. Williams. (2004). *Population Structure of Threatened and Endangered Chinook Salmon ESUs in California's Central Valley Basin*. La Jolla, CA: National Marine Fisheries Service, Southwest Fisheries Science Center.

Lindley, S. T., R. S. Schick, E. Mora, P. B. Adams, J. J. Anderson, S. Greene, C. E. Hanson, B. P. May, D. R. McEwan, R. B. MacFarlane, C. Swanson, and J. G. Williams. (2007). Framework for Assessing Viability of Threatened and Endangered Chinook Salmon and Steelhead in the Sacramento-San Joaquin Basin. *San Francisco Estuary & Watershed Science 5* (1): 1–26.

Lirman, D., S. Schopmeyer, D. Manzello, L. J. Gramer, W. F. Precht, F. Muller-Karger, K. Banks, B. Barnes, E. Bartels, A. Bourque, J. Byrne, S. Donahue, J. Duquesnel, L. Fisher, D. Gilliam, J. Hendee, M. Johnson, K. Maxwell, E. McDevitt, J. Monty, D. Rueda, R. Ruzicka, and S. Thanner. (2011). Severe 2010 cold-water event caused unprecedented mortality to corals of the Florida reef tract and reversed previous survivorship patterns. *PLoS ONE 6* (8): e23047. DOI:10.1371/journal.pone.0023047

Littnan, C. L. (2011). *Habitat Use and Behavioral Monitoring of Hawaiian Monk Seals in Proximity to the Navy Hawaii Range Complex*.

Lobel, L. K. and P. Lobel. (2000). *Coral Reef Protection Implementation Plan*. Washington, DC: U.S. Department of Defense, U.S. Department of the Navy, U.S. Coral Reef Task Force.

Long, E. R., D. D. MacDonald, S. L. Smith, and F. D. Calder. (1995). Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environmental Management 19* (1): 81–97.

Lott, D., E. Bowlby, D. Howard, K. Higgason, K. Grimmer, L. Francis, L. Krop, R. Feely, and L. Jewett. (2011). *National Marine Sanctuaries of the West Coast Ocean Acidification Action Plan*. Monterey, CA: National Oceanic and Atmospheric Administration, National Ocean Service, National Marine Sanctuary Program.

Lough, J. M. and M. J. H. van Oppen. (2009). Coral bleaching: Patterns, processes, causes and consequences. In M. J. H. van Oppen & J. M. Lough (Eds.), *Coral Bleaching* (Vol. Ecological Studies 205). Berlin, Germany: Springer-Verlag.

Love, M. S., M. M. Yoklavich, B. A. Black, and A. H. Andrews. (2007). Age of black coral (*Antipathes dendrochristos*) colonies, with notes on associated invertebrate species. *Bulletin of Marine Science 80* (2): 391–400.

Lovell, J. M., M. M. Findlay, R. M. Moate, and H. Y. Yan. (2005). The hearing abilities of the prawn, *Palaemon serratus*. *Comparative Biochemistry and Physiology, Part A 140* 89–100.

Lovell, J. M., R. M. Moate, L. Christiansen, and M. M. Findlay. (2006). The relationship between body size and evoked potentials from the statocysts of the prawn, *Palaemon serratus*. *The Journal of Experimental Biology 209* 2480–2485. DOI:10.1242/jeb.02211

Lowry, D., S. Wright, M. Neuman, D. Stevenson, J. Hyde, M. Lindeberg, N. Tolimieri, S. Lonhart, S. Traiger, and R. Gustafson. (2022). *Endangered Species Act Status Review Report: Sunflower Sea Star (Pycnopodia helianthoides)*. Seattle, WA: National Marine Fisheries Service, Office of Protected Resources.

Lowry, M. S., J. V. Carretta, and K. A. Forney. (2008). Pacific harbor seal census in California during May-July 2002 and 2004. *California Fish and Game 94* (4): 180–193.

Lowry, M. S., R. Condit, B. Hatfield, S. G. Allen, R. Berger, P. A. Morris, B. J. Le Boeuf, and J. Reiter. (2014). Abundance, distribution, and population growth of the northern elephant seal (*Mirounga angustirostris*) in the United States from 1991 to 2010. *Aquatic Mammals 40* (1): 20–31. DOI:10.1578/am.40.1.2014.20

Lowry, M. S. and K. A. Forney. (2005). Abundance and distribution of California sea lions (*Zalophus californianus*) in central and northern California during 1998 and summer 1999. *Fishery Bulletin 103* (2): 331–343.

Lowry, M. S., E. M. Jaime, and J. E. Moore. (2021). Abundance and distribution of pinnipeds at the Channel Islands in southern California, central and northern California, and southern Oregon during summer 2016–2019. DOI:https://doi.org/10.25923/6qhf-0z55

Lowry, M. S., S. E. Nehasil, and E. M. Jaime. (2017). *Distribution of California Sea Lions, Northern Elephant Seals, Pacific Harbor Seals, and Steller Sea Lions at the Channel Islands During July 2011–2015* (NOAA Technical Memorandum NMFS-SWFSC-578). Springfield, VA: Southwest Fisheries Science Center.

Lumsden, S. E., T. F. Hourigan, A. W. Bruckner, and G. Dorr. (2007). *The State of Deep Coral Ecosystems of the United States: 2007*. Silver Spring, MD: National Oceanic and Atmospheric Administration.

Lunden, J. J., C. G. McNicholl, C. R. Sears, C. L. Morrison, and E. E. Cordes. (2014). Acute survivorship of the deep-sea coral *Lophelia pertusa* from the Gulf of Mexico under acidification, warming, and deoxygenation. *Frontiers in Marine Science 1* (78). DOI:10.3389/fmars.2014.00078

Lusher, A. L., C. O'Donnell, R. Officer, and I. O'Connor. (2016). Microplastic interactions with North Atlantic mesopelagic fish. *ICES Journal of Marine Science 73* (4): 1214–1225. DOI:10.1093/icesjms/fsv241

Lutcavage, M. E. and P. L. Lutz. (1997). Diving Physiology. In P. L. Lutz & J. A. Musick (Eds.), *The Biology of Sea Turtles* (pp. 277–296). Boca Raton, FL: CRC Press.

Lutcavage, M. E., P. Plotkin, B. Witherington, and P. L. Lutz. (1997). Human impacts on sea turtle survival. In P. L. Lutz & J. A. Musick (Eds.), *The Biology of Sea Turtles* (pp. 387–409). New York, NY: CRC Press.

Lux, C. A., A. S. Costa, and A. E. Dizon. (1997). Mitochondrial DNA population structure of the Pacific white-sided dolphin. *Reports of the International Whaling Commission 47* 645–652.

Lynch, J. F., G. G. Glen, Y. Lin, T. F. Duda, and A. E. Newhall. (2018). Impacts of ocean warming on acoustic propogation over continental shelf and slope regions. *Oceanography 31* 174–181. DOI:10.5670/oceanog.2018.219

MacDonald, B. D., R. L. Lewison, S. V. Madrak, J. A. Seminoff, and T. Eguchi. (2012). Home ranges of East Pacific green turtles, *Chelonia mydas*, in a highly urbanized temperate foraging ground. *Marine Ecology Progess Series 461* 211–221.

MacDonald, B. D., S. V. Madrak, R. L. Lewison, J. A. Seminoff, and T. Eguchi. (2013). Fine scale diel movement of the east Pacific green turtle, *Chelonia mydas*, in a highly urbanized foraging environment. *Journal of Experimental Marine Biology and Ecology 443* 56–64. DOI:10.1016/j.jembe.2013.02.033

Machernis, A. F., S. H. Stack, G. L. Olson, F. A. Sullivan, and J. J. Currie. (2021). External scarring as an indicator of fisheries interactions with bottlenose (*Tursiops truncatus*) and pantropical spotted (*Stenella attenuata*) dolphins in Maui Nui, Hawai ʻi. *Aquatic Mammals 47* (5): 482-498.

Mackie, G. O. and C. L. Singla. (2003). The capsular organ of *Chelyosoma productum* (Ascidiacea: Corellidae): A new tunicate hydrodynamic sense organ. *Brain, Behavior and Evolution 61* 45–58.

MacLeod, C. D. and A. D'Amico. (2006). A review of beaked whale behaviour and ecology in relation to assessing and mitigating impacts of anthropogenic noise. *Journal of Cetacean Research and Management 7* (3): 211–222.

MacLeod, C. D., N. Hauser, and H. Peckham. (2004). Diversity, relative density and structure of the cetacean community in summer months east of Great Abaco, Bahamas. *Journal of the Marine Biological Association of the United Kingdom 84* 469–474.

MacLeod, C. D. and G. Mitchell. (2006). Key areas for beaked whales worldwide. *Journal of Cetacean Research and Management 7* (3): 309–322.

MacLeod, C. D., W. F. Perrin, R. Pitman, J. Barlow, L. Ballance, A. D'Amico, T. Gerrodette, G. Joyce, K. D. Mullin, D. L. Palka, and G. T. Waring. (2006). Known and inferred distributions of beaked whale species (Ziphiidae: Cetacea). *Journal of Cetacean Research and Management 7* (3): 271–286.

Macpherson, E. (2002). Large-scale species-richness gradients in the Atlantic Ocean. *Proceedings of the Royal Society of Biology 269* (1501): 1715–1720. DOI:10.1098/rspb.2002.2091

Madej, M. A., A. Torregrosa, and A. Woodward. (2012). *Linking Physical Monitoring to Coho and Chinook Salmon Populations in the Redwood Creek Watershed, California—Summary of May 3–4, 2012 Workshop* (Open-File Report 2012-1245). U.S. Geological Survey.

Madl, P. and M. Yip. (2000). *Cartilagenous fish: Colloquial Meeting of Chondrichthyes; Essay about the Electric Organ Discharge (EOD)*. Salzburg, Austria. Retrieved from http://biophysics.sbg.ac.at/ray/eod.htm.

Madsen, P. T., M. Wahlberg, J. Tougaard, K. Lucke, and P. Tyack. (2006). Wind turbine underwater noise and marine mammals: Implications of current knowledge and data needs. *Marine Ecology Progress Series 309* 279–295. DOI:10.3354/meps309279

Mah, C. L. and D. B. Blake. (2012). Global diversity and phylogeny of the Asteroidea (Echinodermata). *PLoS ONE 7* (4): e35644. DOI:10.1371/journal.pone.0035644

Mahaffy, S. D., R. W. Baird, D. J. McSweeney, D. L. Webster, and G. S. Schorr. (2015). High site fidelity, strong associations, and long-term bonds: Short-finned pilot whales off the island of Hawaiʻi. *Marine Mammal Science 31* (4): 1427–1451. DOI:10.1111/mms.12234

Maison, K. A., I. K. Kelly, and K. P. Frutchey. (2010). *Green Turtle Nesting Sites and Sea Turtle Legislation throughout Oceania* (National Oceanic and Atmospheric Administration Technical Memorandum NMFS-F/SPO-110). Silver Spring, MD: Scientific Publications Office.

Maldini, D., L. Mazzuca, and S. Atkinson. (2005). Odontocete stranding patterns in the main Hawaiian islands (1937–2002): How do they compare with live animal surveys? *Pacific Science 59* (1): 55–67.

Mangels, K. F. and T. Gerrodette. (1994). *Report of Cetacean Sightings During a Marine Mammal Survey in the Eastern Pacific Ocean and the Gulf of California Aboard the NOAA Ships Mcarthur and David Starr Jordan, July 28–November 6, 1993*: Southwest Fisheries Science Center.

Maniscalco, J. M., K. Wynne, K. W. Pitcher, M. B. Hanson, S. R. Melin, and S. Atkinson. (2004). The occurrence of California sea lions (*Zalophus californianus*) in Alaska. *Aquatic Mammals 30* (3): 427–433. DOI:10.1578/AM.30.3.2004.427

Mann, K. H. and J. R. N. Lazier. (1996). *Dynamics of Marine Ecosystems: Biological-Physical Interactions in the Oceans* (2nd ed.). Boston, MA: Blackwell Scientific Publications.

Manta Trust. (2017). *Natural Predation in Manta Rays*. Retrieved April 14, 2017, from http://www.mantatrust.org/threats/natural-predation/.

Manzano-Roth, R., E. E. Henderson, S. W. Martin, C. Martin, and B. M. Matsuyama. (2016). Impacts of U.S. Navy training events on Blainville's beaked whale (*Mesoplodon densirostris*) foraging dives in Hawaiian waters. *Aquatic Mammals 42* (4): 507–518. DOI:10.1578/AM.42.4.2016.507

Manzano-Roth, R. A., E. A. Henderson, S. W. Martin, and B. Matsuyama. (2013). *Impacts of a U.S. Navy training event on beaked whale dives in Hawaiian waters*. Pearl Harbor, HI: U.S. Navy Pacific Fleet.

Maragos, J. E. (2000). Hawaiian Islands (U.S.A.). In C. R. C. Sheppard (Ed.), *Seas at the Millennium: An Environmental Evaluation* (Vol. 2: Regional chapters: The Indian Ocean to the Pacific, pp. 791–812). Amsterdam, Netherlands: Pergamon Press.

Maragos, J. E., D. C. Potts, G. Aeby, D. Gulko, J. Kenyon, D. Siciliano, and D. VanRavenswaay. (2004). 2000–2002 Rapid ecological assessment of corals (Anthozoa) on shallow reefs of the northwestern Hawaiian Islands. Part 1: Species and distribution. *Pacific Science 58* (2): 211–230.

Marine Corps Base Camp Pendleton. (2012). *Integrated Natural Resources Management Plan for Marine Corps Base Camp Pendleton*. Washington, DC: U.S. Department of Defense, U.S. Marine Corps.

Marine Corps Base Hawaii. (2011). *Marine Corps Base Hawaii Integrated Natural Resources Management Plan (MCB Hawaii INRMP) Update (2012–2016)*. Kailua, HI: Sustainable Resources Group International, Inc.

Marks, L. M., P. Salinas-Ruiz, D. C. Reed, S. J. Holbrook, C. S. Culver, J. M. Engle, G. J. Kushner, J. E. Caselle, J. Freiwald, J. P. Williams, J. R. Smith, L. E. Aguilar-Rosas, and N. J. Kaplanis. (2015). Range expansion of a non-native, invasive macroalga *Sargassum horneri* (Turner) C. Agardh, 1820 in the eastern Pacific. *BioInvasions Records 4* (4): 243–248. DOI:http://dx.doi.org/10.3391/bir.2015.4.4.02

Marras, S., R. S. Batty, and P. Domenici. (2012). Information transfer and antipredator maneuvers in schooling herring. *Adaptive Behavior 20* (1): 44–56. DOI:10.1177/1059712311426799

Marsh, H. E. (1989). Mass stranding of dugongs by a tropical cyclone in northern Australia. *Marine Mammal Science 5* (1): 78–84.

Marshall, N. J. (1996). Vision and sensory physiology: The lateral line systems of three deep-sea fish. *Journal of Fish Biology 49* (Supplement A): 239–258.

Martien, K. K., R. W. Baird, N. M. Hedrick, A. M. Gorgone, J. L. Thieleking, D. J. McSweeney, K. M. Robertson, and D. L. Webster. (2012). Population structure of island-associated dolphins: Evidence from mitochondrial and microsatellite markers for common bottlenose dolphins (*Tursiops truncatus*) around the main Hawaiian Islands. *Marine Mammal Science 28* (3): E208–E232. DOI:10.1111/j.1748-7692.2011.00506

Martien, K. K., A. R. Lang, B. L. Taylor, P. E. Rosel, S. E. Simmons, E. M. Oleson, P. L. Boveng, and M. B. Hanson. (2019). The DIP delineation handbook: a guide to using multiple lines of evidence to delineate demographically independent populations of marine mammals.

Martien, K. K., B. L. Taylor, F. I. Archer, K. Audley, J. Calambokidis, T. Cheeseman, J. De Weerdt, A. Frisch Jordán, P. Martínez-Loustalot, and C. D. Ortega-Ortiz. (2021). Evaluation of Mexico Distinct Population Segment of Humpback Whales as units under the Marine Mammal Protection Act.

Martien, K. K., B. L. Taylor, A. R. Lang, P. J. Clapham, D. W. Weller, F. I. Archer, and J. Calambokidis. (2023). The migratory whale herd concept: A novel unit to conserve under the ecological paradigm. *Marine Mammal Science 39* (4): 1267-1292.

Martin, C. R., R. A. Guazzo, T. A. Helble, G. C. Alongi, I. N. Durbach, S. W. Martin, B. M. Matsuyama, and E. E. Henderson. (2022). North Pacific minke whales call rapidly when calling conspecifics are nearby. *Frontiers in Marine Science 9* 897298.

Martin, C. R., S. W. Martin, E. E. Henderson, T. A. Helble, R. A. Manzano-Roth, and B. M. Matsuyama. (2016). *SSC Pacific FY15 annual report on PMRF Marine Mammal Monitoring* (Submitted in Support of the U.S. Navy's 2015 Annual Marine Species Monitoring Report for the Pacific). San Diego, CA: National Marine Mammal Foundation.

Martin, C. R., S. W. Martin, E. E. Henderson, T. A. Helble, R. A. Manzano-Roth, B. M. Matsuyama, and G. C. Alongi. (2017). *SSC Pacific FY16 annual report on PMRF Marine Mammal Monitoring. Final Report*. San Diego, CA: National Marine Mammal Foundation; and Space and Naval Warfare Systems Center Pacific.

Martin, S. L., Z. Siders, T. Eguchi, B. J. Langseth, A. Yau, J. D. Baker, R. Ahrens, and T. T. Jones. (2020). Assessing the population level impacts of North Pacific loggerhead and western Pacific leatherback interactions in the Hawaii-based shallow set longline fishery.

Martínez-Abraín, A. (2008). Statistical significance and biological relevance: A call for a more cautious interpretation of results in ecology. *Acta Oecologica 34* (1): 9–11. DOI:10.1016/j.actao.2008.02.004

Martinez, B., F. Arenas, M. Rubal, S. Burgues, R. Esteban, I. Garcia-Plazaola, F. L. Figueroa, R. Pereira, L. Saldana, I. Sousa-Pinto, A. Trilla, and R. M. Viejo. (2012). Physical factors driving intertidal macroalgae distribution: Physiological stress of a dominant fucoid at its southern limit. *Oecologia 170* (2): 341–353. DOI:10.1007/s00442-012-2324-x

Masaki, Y. (1976). Biological studies on the North Pacific sei whale. *Bulletin of the Far Seas Fisheries Research Laboratory 14* 1–104

Masaki, Y. (1977). The separation of the stock units of sei whales in the North Pacific. *Reports of the International Whaling Commission Special Issue 1* 71–79.

Massey, B. and J. Fancher. (1989). Renesting by California Least Terns. *Journal of Field Ornithology 60* (3): 350–357.

Massey, L. M., S. Penna, E. Zahn, D. Lawson, and C. M. Davis. (2023). Monitoring Green Sea Turtles in the San Gabriel River of Southern California. *Animals 13* (3): 434. DOI:10.3390/ani13030434

Mate, B., A. Bradford, G. Tsidulko, and V. Ilyashenko. (2013). *Late-Feeding Season Movements of a Western North Pacific Gray Whale off Sakhalin Island, Russia and Subsequent Migration into the Eastern North Pacific* (Paper SC/63/BRG23 presented to the International Whaling Commission Scientific Committee).

Mate, B., D. M. Palacios, L. M. Irvine, B. Lagerquist, T. Follett, M. Winsor, and C. Hayslip. (2015a). *Baleen (Blue & Fin) Whale Tagging in Southern California in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas (SOCAL, NWTRC, GOA)*. U.S. Navy Pacific Fleet.

Mate, B. R., R. Gisiner, and J. Mobley. (1997). Local and migratory movements of Hawaiian humpback whales tracked by satellite telemetry. *Canadian Journal of Zoology 76* 863–868.

Mate, B. R., R. Gisiner, and J. Mobley. (1998). Local and migratory movements of Hawaiian humpback whales tracked by satellite telemetry. *Canadian Journal of Zoology 76* (5): 863–868.

Mate, B. R., V. Y. Ilyashenko, A. L. Bradford, V. V. Vertyankin, G. A. Tsidulko, V. V. Rozhnov, and L. M. Irvine. (2015b). Critically endangered western gray whales migrate to the eastern North Pacific. *Biology Letters 11* (4): 1–4. DOI:10.1098/rsbl.2015.0071

Mate, B. R., D. M. Palacios, C. S. Baker, B. A. Lagerquist, L. M. Irvine, T. Follett, and D. Steel. (2018). *Humpback Whale Tagging in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas in the Pacific Ocean: Preliminary Summary of Field Tagging Efforts off the Pacific Northwest in Summer 2018*. Newport, OR: Marine Mammal Institute, Oregon State University.

Mate, B. R., D. M. Palacios, C. S. Baker, B. A. Lagerquist, L. M. Irvine, T. Follett, D. Steel, C. Hayslip, and M. H. Winsor. (2016a). *Baleen (Blue and Fin) Whale Tagging in Southern California in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas* (Prepared for Naval Facilities Engineering Pacific for Commander, U.S. Pacific Fleet under Contract Nos. 62470-10-D-3011 (KB29) and N62470-15-D-8006 (KB01) issued to HDR, Inc.). San Diego, CA: HDR Inc.

Mate, B. R., D. M. Palacios, C. S. Baker, B. A. Lagerquist, L. M. Irvine, T. Follett, D. Steel, C. Hayslip, and M. H. Winsor. (2016b). *Baleen (Blue and Fin) Whale Tagging in Southern California in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas. Preliminary Summary* (Prepared for Naval Facilities Engineering Command Pacific for Commander, U.S. Pacific Fleet under Contract Nos. N62470-10-D-3011 (KB29) and N62470-15-D-8006 (KB01) issued to HDR, Inc.). San Diego, CA: HDR Inc.

Mate, B. R., D. M. Palacios, C. S. Baker, B. A. Lagerquist, L. M. Irvine, T. Follett, D. Steel, C. Hayslip, and M. H. Winsor. (2017). *Baleen Whale Tagging in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas Covering the Years 2014, 2015, and 2016. Final Report* (Submitted to Naval Facilities Engineering Command Pacific, Pearl Harbor, Hawaii under Contract No. N62470-15-8006 (FZN1) issued to HDR, Inc.). San Diego, CA: Oregon State University Marine Mammal Institute.

Mate, B. R., D. M. Palacios, C. S. Baker, B. A. Lagerquist, L. M. Irvine, T. M. Follett, D. Steel, and C. E. Hayslip. (2019). *Humpback Whale Tagging in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas in the Pacific Ocean: Preliminary Summary of Field Tagging Effort* *in Hawaii in March 2019*. Newport, OR: Oregon State University, Marine Mammal Institute, Hatfield Marine Science Center.

Mate, B. R., D. M. Palacios, L. M. Irvine, B. A. Lagerquist, T. Follett, M. H. Winsor, and C. Hayslip. (2015c). *Baleen (Blue & Fin) Whale Tagging in Southern California in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas (SOCAL, NWTRC, GOA); Final Report* (Prepared for Naval Facilities Engineering Command Pacific for Commander, U.S. Pacific Fleet under Contract No. N62470-10-D-3011, Task Orders JP03 and KB27 issued to HDR, Inc.). Honolulu, HI: HDR, Inc.

Mate, B. R. and J. Urban-Ramirez. (2003). A note on the route and speed of a gray whale on its northern migration from Mexico to central California, tracked by satellite-monitored radio tag. *Journal of Cetacean Research and Management 5* (2): 155–157.

Mato, Y., T. Isobe, H. Takada, H. Kanehiro, C. Ohtake, and T. Kaminuma. (2001). Plastic resin pellets as a transport medium for toxic chemicals in the marine environment. *Environmental Science Technology 35* 318–324.

Maui Forest Bird Recovery Project. (2017). *Avian Disease*. Retrieved May 31, 2017, from http://mauiforestbirds.org/articles/17.

Mauzey, K. P., C. Birkland, and P. K. Dayton. (1968). Feeding behavior of asteroids and escape responses of their prey in the Puget Sound region. *Ecology 49* (4): 603-619.

McAlpine, D. F. (2009). Pygmy and dwarf sperm whales, *Kogia breviceps* and *K. sima*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 936–938). Academic Press.

McCaskie, G. and K. Garrett. (2001). North American Birds: Southern Pacific Coast. *North American Birds 55* (2): 226–230.

McCaskie, G. and K. Garrett. (2002). North American Birds: Southern Pacific Coast. *North American Birds 56* 222–226.

McClanahan, T. R. (2017). Changes in coral sensitivity to thermal anomalies. *Marine Ecology Progress Series 570* 71–85.

McClanahan, T. R., S. D. Donner, J. A. Maynard, M. A. MacNeil, N. A. Graham, J. Maina, A. C. Baker, I. J. Alemu, M. Beger, S. J. Campbell, E. S. Darling, C. M. Eakin, S. F. Heron, S. D. Jupiter, C. J. Lundquist, E. McLeod, P. J. Mumby, M. J. Paddack, E. R. Selig, and R. van Woesik. (2012). Prioritizing key resilience indicators to support coral reef management in a changing climate. *PLoS ONE 7* (8): e42884. DOI:10.1371/journal.pone.0042884

McCorkle, S. L. (2016). *Fibropapillomatosis in Marine Turtles: A Comprehensive Review.* (Unpublished master's thesis). Nova Southeastern University, Fort Lauderdale, FL.

McCracken, M. L. (2000). *Estimation of Sea Turtle Take and Mortality in the Hawaiian Longline Fisheries* (Administrative Report H-00-06). Honolulu, HI: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

McCracken, M. L. (2014). *Estimation of Incidental Interactions with Sea Turtles and Seabirds in the 2013 Hawaii Deep-Set Longline Fishery*. Honolulu, HI: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

McCracken, M. L. (2019). *Assessment of Incidental Interactions with Marine Mammals in the Hawaii Longline Deep and Shallow Set Fisheries from 2014 through 2018*. Honolulu, HI: Pacific Islands Fisheries Science Center.

McDermott, J. J. (2001). Status of the Nemertea as prey in marine ecosystems. *Hydrobiologia 456* (1): 7–20.

McEwan, D. R. (2001). Central Valley steelhead. *Fish Bulletin 179* (1): 1–43.

McGrew, K. A., S. E. Crowell, J. L. Fiely, A. M. Berlin, G. H. Olsen, J. James, H. Hopkins, and C. K. Williams. (2022). Underwater hearing in sea ducks with applications for reducing gillnet bycatch through acoustic deterrence. *The Journal of Experimental Biology 225* (20). DOI:10.1242/jeb.243953

McIver, W. R., S. F. Pearson, C. Strong, M. M. Lance, J. Baldwin, D. Lynch, M. G. Raphael, R. D. Young, and N. Johnson. (2021). *Status and trend of marbled murrelet populations in the Northwest Forest Plan area, 2000 to 2018*. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

McKenna, M. F., J. Calambokidis, E. M. Oleson, D. W. Laist, and J. A. Goldbogen. (2015). Simultaneous tracking of blue whales and large ships demonstrates limited behavioral responses for avoiding collision. *Endangered Species Research 27* (3): 219–232. DOI:10.3354/esr00666

McLaughlin, K., R. Mazor, K. Schiff, and L. Hampton. (2022). *Southern California Bight 2018 Regional Monitoring Program: Volume IX. Trash and Marine Debris*. Costa Mesa, CA: Southern California Coastal Water Research Project.

McMurray, S., J. Vicente, K. Jabanoski, and T. Lewis. (2012). Spawning of the basket star, *Astrophyton muricatum,* in the Bahamas. *Coral Reefs 1* 1. DOI:10.1007/s00338-012-0884-1

McSweeney, D. J., R. W. Baird, and S. D. Mahaffy. (2007). Site fidelity, associations, and movements of Cuvier's (*Ziphius Cavirostris*) and Blainville's (*Mesoplodon Densirostris*) beaked whales off the island of Hawaii. *Marine Mammal Science 23* (3): 666–687. DOI:10.1111/j.1748-7692.2007.00135

McSweeney, D. J., R. W. Baird, S. D. Mahaffy, D. L. Webster, and G. S. Schorr. (2009). Site fidelity and association patterns of a rare species: Pygmy killer whales (*Feresa attenuata*) in the main Hawaiian Islands. *Marine Mammal Science 25* (3): 557–572. DOI:10.1111/j.1748-7692.2008.00267

Mead, J. G. (1981). First records of *Mesoplodon hectori* (Ziphiidae) from the Northern Hemisphere and a description of the adult male. *Journal of Mammalogy 62* (2): 430–432.

Mead, J. G. (1989). Beaked whales of the genus *Mesoplodon*. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 4, pp. 349–430). San Diego, CA: Academic Press.

Mearns, A. J., D. J. Reish, P. S. Oshida, T. Ginn, and M. A. Rempel-Hester. (2011). Effects of Pollution on Marine Organisms. *Water Environment Research 83* (10): 1789–1852.

Mélançon, J., M. Levasseur, M. Lizotte, M. Scarratt, J. É. Tremblay, P. Tortell, G.-P. Yang, G.-Y. Shi, H. Gao, D. Semeniuk, M. Robert, M. Arychuk, K. F. Johnson, N. Sutherland, M. Davelaar, N. Nemcek, A. Peña, and W. Richardson. (2016). Impact of ocean acidification on phytoplankton assemblage, growth, and DMS production following Fe-dust additions in the NE Pacific high-nutrient, low-chlorophyll waters. *Biogeosciences 13* (5): 1677–1692. DOI:10.5194/bg-13-1677-2016

Melillo, J. M., T. Richmond, and G. W. Yohe. (2014). *Climate Change Impacts in the United States: The Third National Climate Assessment*. Washington, DC: U.S. Global Change Research Program.

Melin, S. R. and R. L. DeLong. (1999). Observations of a Guadalupe fur seal (*Arctocephalus townsendi*) female and pup at San Miguel Island, California. *Marine Mammal Science 15* (3): 885–887.

Melin, S. R. and R. L. DeLong. (2000). *At-sea distribution and diving behavior of California sea lion females from San Miguel Island, California* (Proceedings of the Fifth California Islands Symposium). Santa Barbara, CA: U.S. Department of the Interior, Minerals Management Service.

Melin, S. R., R. L. DeLong, and D. B. Siniff. (2008). The effects of El Niño on the foraging behavior of lactating California sea lions (*Zalophus californianus californianus*) during the nonbreeding season. *Canadian Journal of Zoology 86* (3): 192–206. DOI:10.1139/z07-132

Melin, S. R., J. T. Sterling, R. R. Ream, R. G. Towell, T. Zeppelin, A. J. Orr, B. Dickerson, N. Pelland, and C. E. Kuhn. (2012). *A Tale of Two Stocks: Studies of Northern Fur Seals Breeding at the Northern and Southern Extent of the Range*: Alaska Fisheries Science Center (AFSC).

Menge, B. A. and G. M. Branch. (2001). Rocky intertidal communities. In M. D. Bertness, S. D. Gaines, & M. E. Hay (Eds.), *Marine Community Ecology* (pp. 221–252). Sunderland, MA: Sinauer Associates, Inc.

Mercer, T. (2023). *Summary of documented human-caused mortality, serious injury and non-serious injury in Hawaiian monk seals: Supporting documentation for the 2023 Stock Assessment Report*. Honolulu, HI: Joint Institute for Marine and Atmospheric Research, University of Hawaiʻi at Manoa.

Mercer, T. A. (2021). Summary of documented human-caused mortality, serious injury and non-serious injury in Hawaiian monk seals: supporting documentation for the 2021 Stock Assessment Report.

Merkel & Associates Inc. (2014). *2014 San Diego Bay Eelgrass Inventory Update* (Prepared for U.S. Navy Region Southwest Naval Facilities Engineering Command and San Diego Unified Port District). San Diego, CA: Merkel & Associates, Inc.

Merkel, F. R. and K. L. Johansen. (2011). Light-induced bird strikes on vessels in Southwest Greenland. *Marine Pollution Bulletin 62* (11): 2330–2336.

Mesnick, S. L., B. L. Taylor, F. I. Archer, K. K. Martien, S. E. Trevino, B. L. Hancock-Hanser, S. C. M. Medina, V. L. Pease, K. M. Robertson, J. M. Straley, R. W. Baird, J. Calambokidis, G. S. Schorr, P. Wade, V. Burkanov, C. R. Lunsford, L. Rendell, and P. A. Morin. (2011). Sperm whale population structure in the eastern and central North Pacific inferred by the use of single-nucleotide polymorphisms, microsatellites and mitochondrial DNA. *Molecular Ecology Resources 11 (Supplement 1)* 278–298. DOI:10.1111/j.1755-0998.02973

Meyer, M., R. R. Fay, and A. N. Popper. (2010). Frequency tuning and intensity coding of sound in the auditory periphery of the lake sturgeon, *Acipenser fulvescens*. *The Journal of Experimental Biology 213* 1567–1578. DOI:10.1242/jeb.031757

Meylan, A. (1988). Spongivory in hawksbill turtles: A diet of glass. *Science 239* (4838): 393–395.

Mignucci-Giannoni, A. A. (1998). Zoogeography of cetaceans off Puerto Rico and the Virgin Islands. *Caribbean Journal of Science 34* (3–4): 173–190.

Miller, E. A., G. P. Singer, M. L. Peterson, E. D. Chapman, M. E. Johnston, M. J. Thomas, R. D. Battleson, M. Gingras, and A. P. Klimley. (2020a). Spatio-temporal distribution of green sturgeon (*Acipenser medirostris*) and white sturgeon (*A. transmontanus*) in the San Francisco estuary and

Sacramento River, California. *Environmental Biology of Fishes 103* 577–603. DOI:10.1007/s10641-020-00972-x

Miller, J., S. Dollar, A. Millan, S. Peltier, P. Andersen, M. Ericksen, C. Conger, and G. Tumino. (2023). *Aquatic Invasive Species Report, Phase 2, Marine Mitigation in Support of P-209 Dry Dock 3 Replacement Joint Base Pearl Harbor-Hickam, Hawaii*. Joint Base Pearl Harbor-Hickam, HI: Naval Facilities Engineering Systems Command Pacific for Commander, Navy Region Hawaii.

Miller, K. J., A. A. Rowden, A. Williams, and V. Häussermann. (2011). Out of their depth? Isolated deep populations of the cosmopolitan coral *Desmophyllum dianthus* may be highly vulnerable to environmental change. *PLoS ONE 6* (5): e19004. DOI:10.1371/journal.pone.0019004

Miller, K. W. and V. B. Scheffer. (1986). False killer whale. In D. Haley (Ed.), *Marine Mammals of the Eastern North Pacific and Arctic Waters* (pp. 148–151). Seattle, WA: Pacific Search Press.

Miller, L. J., A. D. Mackey, T. Hoffland, and S. A. Kuczaj, II. (2010). Potential effects of a major hurricane on Atlantic bottlenose dolphin (*Tursiops truncatus*) reproduction in the Mississippi Sound. *Marine Mammal Science 26* (3): 707–715. DOI:10.1111/j.1748-7692.2010.00371

Miller, M. A., P. J. Duignan, E. Dodd, F. Batac, M. Staedler, J. A. Tomoleoni, M. Murray, H. Harris, and C. Gardiner. (2020b). Emergence of a Zoonotic Pathogen in a Coastal Marine Sentinel: *Capillaria hepatica* (syn. *Calodium hepaticum*)-Associated Hepatitis in Southern Sea Otters (*Enhydra lutris nereis*). *Frontiers in Marine Science 7* (335).

Miller, M. A., M. E. Moriarty, L. Henkel, M. T. Tinker, T. L. Burgess, F. I. Batac, E. Dodd, C. Young, M. D. Harris, D. A. Jessup, J. Ames, P. A. Conrad, A. E. Packham, and C. K. Johnson. (2020c). Predators, disease, and environmental change in the Nearshore Ecosystem: Mortality in Southern Sea Otters (*Enhydra lutris nereis*) From 1998–2012. *Frontiers in Marine Science, 19*. Retrieved from https://doi.org/10.3389/fmars.2020.00582.

Miller, M. H., J. Carlson, P. Cooper, D. Kobayashi, M. Nammack, and J. Wilson. (2014). *Status Review Report: Scalloped Hammerhead Shark (Sphyrna lewini)*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

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.

Miller, S. L. and C. J. Ralph. (1995). *Relationship of marbled murrelets with habitat characteristics at inland sites in California* (Ecology and Conservation of the Marbled Murrelet). Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture.

Miloslavich, P., E. Klein, J. M. Díaz, C. E. Hernández, G. Bigatti, L. Campos, F. Artigas, J. Castillo, P. E. Penchaszadeh, P. E. Neill, A. Carranza, M. V. Retana, J. M. Díaz de Astarloa, M. Lewis, P. Yorio, M. L. Piriz, D. Rodríguez, Y. Yoneshigue-Valentin, L. Gamboa, and A. Martín. (2011). Marine Biodiversity in the Atlantic and Pacific Coasts of South America: Knowledge and Gaps. *PLoS ONE 6* (1): e14631. DOI:10.1371/journal.pone.0014631

Milstein, M. (2015, October 2015). *Oncoming El Niño Likely to Continue Species Shakeup in Pacific*. Retrieved 2/18/2016, 2016, from http://www.nwfsc.noaa.gov/news/features/el_nino/index.cfm.

Milton, D. A. (2001). Assessing the susceptibility to fishing of populations of rare trawl bycatch: Sea snakes caught by Australia’s Northern Prawn Fishery. *Biological Conservation 101* (3): 281–290.

Miner, C. M., J. L. Burnaford, R. F. Ambrose, L. Antrim, H. Bohlmann, C. A. Blanchette, J. M. Engle, S. C. Fradkin, R. Gaddam, C. D. G. Harley, B. G. Miner, S. N. Murray, J. R. Smith, S. G. Whitaker, and P. T. Raimondi. (2018). Large-scale impacts of sea star wasting disease (SSWD) on intertidal sea stars and implications for recovery. *PLoS ONE 13* (3): e0192870. DOI:https://doi.org/10.1371/journal.pone.0192870

Minton, D. and M. Carnevale. (2006). *A Preliminary Survey of the Marine and Intertidal Invertebrates of Kalaupapa National Historical Park*. Molokai, HI: National Park Service Report submitted to Kalaupapa National Historical Park, Natural Resources Division.

Mitchell, C. P. J. and C. C. Gilmour. (2008). Methylmercury production in a Chesapeake Bay salt marsh. *Journal of Geophysical Research 113* (G2): G00C04. DOI:10.1029/2008jg000765

Mitsch, W. J. and J. G. Gosselink. (2007). *Wetlands* (Fourth ed.). Hoboken, NJ: John Wiley & Sons, Inc.

Mitsch, W. J., J. G. Gosselink, C. J. Anderson, and L. Zhang. (2009a). *Wetland Ecosystems*. Hoboken, NJ: John Wiley & Sons, Inc.

Mitsch, W. J., J. G. Gosselink, C. J. Anderson, and L. Zhang. (2009b). *Wetland Ecosystems*. Hoboken, NJ: John Wiley & Sons, Inc.

Miyashita, T., T. Kishiro, N. Higashi, F. Sato, K. Mori, and H. Kato. (1996). Winter distribution of cetaceans in the western North Pacific inferred from sighting cruises 1993—1995. *Reports of the International Whaling Commission 46* 437–442.

Miyazaki, N. and S. Wada. (1978). Fraser's dolphin, *Lagenodelphis hosei,* in the western North Pacific. *The Scientific Reports of the Whales Research Institute 30* 231–244.

Mizroch, S. A., D. W. Rice, D. Zwiefelhofer, J. Waite, and W. L. Perryman. (2009). Distribution and movements of fin whales in the North Pacific Ocean. *Mammal Review 39* (3): 193–227. DOI:10.1111/j.1365-2907.2009.00147.x

Mobley, J. R. (2004). *Results of Marine Mammal Surveys on U.S. Navy Underwater Ranges in Hawaii and Bahamas*. Arlington, VA: Office of Naval Research.

Mobley, J. R. (2005). Assessing responses of humpback whales to North Pacific Acoustic Laboratory (NPAL) transmissions: Results of 2001–2003 aerial surveys north of Kauai. *The Journal of Acoustical Society of America 117* 1666–1773. DOI:10.1121/1.1854475

Mobley, J. R., G. B. Bauer, and L. M. Herman. (1999). Changes over a ten-year interval in the distribution and relative abundance of humpback whales (*Megaptera novaeangliae*) wintering in Hawaiian waters. *Aquatic Mammals 25* (2): 63–72.

Mobley, J. R., M. H. Deakos, S. W. Martin, and R. Manzano-Roth. (2015). *Aerial Survey Monitoring for Marine Mammals and Sea Turtles in the Hawaii Range Complex in Conjunction with a Navy Training Event: SCC 18–20 February 2014, Final Report* (Prepared for Naval Facilities Engineering Command Pacific for Commander, U.S. Pacific Fleet under Contract No. N62470-10-D-3011, CTO KB26, issued to HDR, Inc.). Honolulu, HI: HDR Inc.

Mobley, J. R., L. Mazzuca, A. S. Craig, M. W. Newcomer, and S. S. Spitz. (2001a). Killer whales (*Orcinus orca*) sighted west of Niihau, Hawaii. *Pacific Science 55* (3): 301–303.

Mobley, J. R. and A. F. Pacini. (2013). *Aerial Survey Monitoring for Marine Mammals and Sea Turtles in the Hawaii Range Complex in Conjunction with a Navy Training Event, SCC February 19-21 and August 12-13, 2013, Final Field Report*. Pearl Harbor, HI: U.S. Navy Pacific Fleet.

Mobley, J. R., S. Spitz, and R. Grotefendt. (2001b). *Abundance of Humpback Whales in Hawaiian Waters: Results of 1993–2000 Aerial Surveys*. Honolulu, HI: Hawaiian Islands Humpback Whale National Marine Sanctuary, Department of Land and Natural Resources, State of Hawaii.

Mobley, J. R., S. S. Spitz, K. A. Forney, R. Grotefendt, and P. H. Forestell. (2000). *Distribution and Abundance of Odontocete Species in Hawaiian Waters: Preliminary Results of 1993–98 Aerial Surveys*. Pearl City, HI; Colorado Springs, CO; La Jolla, CA; North Bend, WA; and Southampton, NY: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Monnahan, C. (2013). *Population Trends of the Eastern North Pacific Blue Whale.* University of Washington.

Monnahan, C. C., T. A. Branch, and A. E. Punt. (2015). Do ship strikes threaten the recovery of endangered eastern North Pacific blue whales? *Marine Mammal Science 31* (1): 279–297. DOI:10.1111/mms.12157

Monnahan, C. C., T. A. Branch, K. M. Stafford, Y. V. Ivashchenko, and E. M. Oleson. (2014). Estimating historical eastern North Pacific blue whale catches using spatial calling patterns. *PLoS ONE 9* (6): e98974. DOI:10.1371/journal.pone.0098974

Monterey Bay Aquarium Research Institute. (2009, 05 February 2009). *Marine Flora of Monterey*. Retrieved from http://www.mbari.org/staff/conn/botany/flora/mflora.htm.

Montero, J. T., R. O. Martinez-Rincon, S. S. Heppell, M. Hall, and M. Ewal. (2016). Characterizing environmental and spatial variables associated with the incidental catch of olive ridley (*Lepidochelys olivacea*) in the Eastern Tropical Pacific purse-seine fishery. *Fisheries Oceanography 25* (1): 1–14. DOI:10.1111/fog.12130

Montgomery, J. C., A. Jeffs, S. D. Simpson, M. Meekan, and C. Tindle. (2006). Sound as an orientation cue for the pelagic larvae of reef fishes and decapod crustaceans. *Advances in Marine Biology 51* 143–196.

Mooney, T. A., R. T. Hanlon, J. Christensen-Dalsgaard, P. T. Madsen, D. Ketten, and P. E. Nachtigall. (2010). Sound detection by the longfin squid (*Loligo pealeii*) studied with auditory evoked potentials: Sensitivity to low-frequency particle motion and not pressure. *The Journal of Experimental Biology 213* 3748–3759.

Mooney, T. A., A. Smith, O. N. Larsen, K. A. Hansen, and M. Rasmussen. (2020). A field study of auditory sensitivity of the Atlantic puffin, *Fratercula arctica*. *Journal of Experimental Biology 223*. DOI:10.1242/jeb.228270

Mooney, T. A., A. Smith, O. N. Larsen, K. A. Hansen, M. Wahlberg, and M. H. Rasmussen. (2019). Field-based hearing measurements of two seabird species. *Journal of Experimental Biology 222* 1–7. DOI:DOI:10.1242/jeb.190710

Moore, C. J. (2008). Synthetic polymers in the marine environment: A rapidly increasing, long-term threat. *Environmental Research 108* (2): 131–139. DOI:10.1016/j.envres.1008.07.025

Moore, J. and J. Barlow. (2017a). *Population Abundance and Trend Estimates for Beaked Whales and Sperm Whales in the California Current from Ship-Based Visual Line-Transect Survey Data, 1991–2014* (National Oceanic and Atmospheric Administration Technical Memorandum NMFS-SWFSC-585). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Moore, J. and J. Barlow. (2017b). *Population Abundance and Trend Estimates for Beaked Whales and Sperm Whales in the California Current from Ship-Based Visual Line-Transect Survey Data, 1991–2014* (NOAA Technical Memorandum NMFS-SWFSC-585). La Jolla, CA: Southwest Fisheries Science Center.

Moore, J. C. (1972). More skull characters of the beaked whale, *Indopacetus pacificus,* and comparative measurements of austral relatives. *Fieldiana Zoology 62* (1): 1–19.

Moore, J. E. and J. Barlow. (2011). Bayesian state-space model of fin whale abundance trends from a 1991–2008 time series of line-transect surveys in the California Current. *Journal of Applied Ecology* 1–11. DOI:10.1111/j.1365-2664.2011.02018.x

Moore, J. E. and J. P. Barlow. (2013). Declining abundance of beaked whales (Family Ziphiidae) in the California Current Large Marine Ecosystem. *PLoS ONE 8* (1): e52770. DOI:10.1371/journal.pone.0052770

Moore, J. E. and J. P. Barlow. (2014). Improved abundance and trend estimates for sperm whales in the eastern North Pacific from Bayesian hierarchical modeling. *Endangered Species Research 25* (2): 141–150. DOI:10.3354/esr00633

Moore, M. E. and B. A. Berejikian. (2022). Coastal infrastructure alters behavior and increases predation mortality of threatened Puget Sound steelhead smolts. *Ecosphere 13* (4). DOI:10.1002/ecs2.4022

Moore, S., M. Sutula, T. Von Bitner, G. Lattin, and K. Schiff. (2016). *Southern California Bight 2013 Regional Monitoring Program: Volume III. Trash and Marine Debris*. Costa Mesa, CA: Southern California Coastal Water Research Project.

Moore, S. K., V. L. Trainer, N. J. Mantua, M. S. Parker, E. A. Laws, L. C. Backer, and L. E. Fleming. (2008). Impacts of climate variability and future climate change on harmful algal blooms and human health. *Environmental Health 7* (Supplement 2): S4. DOI:10.1186/1476-069X-7-S2-S4

Mora, E. A., R. D. Battleson, S. T. Lindley, M. J. Thomas, R. Bellmer, L. J. Zarri, and A. P. Klimley. (2018). Estimating the annual spawning run size and population size of the southern distinct population segment of Green Sturgeon. *Transactions of the American Fisheries Society 147* 195–203.

Morales-Zárate, M., J. López-Ramírez, and C. Salinas-Zavala. (2021). Loggerhead marine turtle (Caretta caretta) ecological facts from a trophic relationship model in a hot spot fishery area: Gulf of Ulloa, Mexico. *Ecological Modelling 439* 109327.

Morejohn, G. V. (1979). The Natural History of Dall's Porpoise in the North Pacific Ocean (pp. 45–83).

Morel, F. M. M. and N. M. Price. (2003). The biogeochemical cycles of trace metals in the oceans. *Science 300* (5621): 944–947. DOI:10.1126/science.1083545

Moretti, D. (2016). *Marine Mammal Monitoring on Navy Ranges (M3R) Passive Acoustic Monitoring of Abundance on the Pacific Missile Range Facility (PMFR) and southern California Offshore Range (SCORE)* (Submitted in Support of the U.S. Navy's 2015 Annual Marine Species Monitoring Report for the Pacific). Newport, RI: Naval Undersea Warfare Center.

Moriarty, M. E., M. T. Tinker, M. A. Miller, J. A. Tomoleoni, M. M. Staedler, J. A. Fujii, F. I. Batac, E. M. Dodd, R. M. Kudela, V. Zubkousky-White, and C. K. Johnson. (2021). Exposure to domoic acid is an ecological driver of cardiac disease in southern sea otters. *Harmful Algae 101*.

Morin, P. A., M. L. McCarthy, C. W. Fung, J. W. Durban, K. M. Parsons, W. F. Perrin, B. L. Taylor, T. A. Jefferson, and F. I. Archer. (2024). Revised taxonomy of eastern North Pacific killer whales

(Orcinus orca): Bigg's and resident ecotypes deserve species status. *Royal Society Open Science 11* (3): 231368.

Morris, J. F. T., M. Trudel, M. E. Thiess, R. M. Sweeting, J. Fisher, S. A. Hinton, E. A. Ferguson, J. A. Orsi, E. V. Farley Jr., and D. W. Welch. (2007). Stock-specific migrations of juvenile Coho Salmon derived from coded-wire tag recoveries on the continental shelf of Western North America. *American Fisheries Society Symposium 57* (81–104).

Moser, M. L. and S. T. Lindley. (2007). Use of Washington estuaries by subadult and adult green sturgeon. *Environmental Biology of Fishes 79* 243–253.

Moyle, P. B. (2002). *Inland Fishes of California*. Los Angeles, CA: University of California Press.

Moyle, P. B. and J. J. Cech, Jr. (2004). *Fishes: An Introduction to Ichthyology* (5th ed.). London, United Kingdom: Pearson Educational, Inc.

Moyle, P. B., R. A. Lusardi, P. J. Samuel, and J. V. E. Katz. (2017). *State of the Salmonids: Status of California's Emblematic Fishes 2017*. Davis, CA: UC Davis Center for Watershed Sciences.

Mrosovsky, N., G. D. Ryan, and M. C. James. (2009). Leatherback turtles: The menace of plastic. *Marine Pollution Bulletin 58* (2): 287–289. DOI:10.1016/j.marpolbul.2008.10.018

Muir, J. E., L. Ainsworth, R. Racca, Y. Bychkov, G. Gailey, V. Vladimirov, S. Starodymov, and K. Bröker. (2016). Gray whale densities during a seismic survey off Sakhalin Island, Russia. *Endangered Species Research 29* (3): 211–227. DOI:10.3354/esr00709

Muller, E. M., N. M. Leporacci, K. J. Macartney, A. G. Shea, R. E. Crane, E. R. Hall, and K. B. Ritchie. (2017). Low pH reduces the virulence of black band disease on *Orbicella faveolata*. *PLoS ONE 12* (6).

Munday, P. L., D. L. Dixson, J. M. Donelson, G. P. Jones, M. S. Pratchett, G. V. Devitsina, and K. B. Doving. (2009). Ocean acidification impairs olfactory discrimination and homing ability of a marine fish. *Proceedings of the National Academy of Sciences of the United States of America 106* (6): 1848–1852.

Murase, H., T. Tamura, S. Otani, and S. Nishiwaki. (2015). Satellite tracking of Bryde's whales *Balaenoptera edeni* in the offshore western North Pacific in summer 2006 and 2008. *Fisheries Science 82* (1): 35–45. DOI:10.1007/s12562-015-0946-8

Murphy, J. C. (2012). Marine invasions by non-sea snakes, with thoughts on terrestrial-aquatic-marine transitions. *Integrative and Comparative Biology 52* (2): 217–226.

Murray, C. C., A. Bychkov, T. Therriault, H. Maki, and N. Wallace. (2015). The impact of Japanese tsunami debris on North America. *PICES Press 23* (1): 28.

Murray, F., S. Hennige, J. M. Roberts, E. Anagnostou, A. Gori, L. Wicks, J. Stewart, and G. Foster. (2016, October 19-21, 2016). *Cold-water-corals in a high CO_2 ocean: Behaviour, physiology and growth in Desmophyllum dianthus*. Presented at the Annual Science Meeting. University of Strathclyde, Glasgow, Scotland.

Musick, J. A. and S. L. Fowler. (2007). Sphyrna lewini. *IUCN Red List of Threatened Species 2014.2.*

Musick, J. A., M. M. Harbin, S. A. Berkeley, G. H. Burgess, A. M. Eklund, L. Findley, R. G. Gilmore, J. T. Golden, D. S. Ha, G. R. Huntsman, J. C. McGovern, S. J. Parker, S. G. Poss, E. Sala, T. W. Schmidt, G. R. Sedberry, H. Weeks, and S. G. Wright. (2000). Marine, estuarine, and diadromous fish stocks at risk of extinction in North America (exclusive of Pacific salmonids). *Fisheries 25* (11): 6–30.

Musick, J. A. and C. J. Limpus. (1997). Habitat utilization and migration of juvenile sea turtles. In P. L. Lutz & J. A. Musick (Eds.), *The Biology of Sea Turtles* (pp. 137–163). Boca Raton, FL: CRC Press.

Mussi, B., A. Miragliuolo, T. De Pippo, M. C. Gambi, and D. Chiota. (2004). The submarine canyon of Cuma (southern Tyrrhenian Sea, Italy), a cetacean key area to protect. *European Research on Cetaceans 15* 178–179.

Muto, M. M. and R. P. Angliss. (2016). *Alaska Marine Mammal Stock Assessments, 2015*. Seattle, WA.

Muto, M. M., V. T. Helker, R. P. Angliss, B. A. Allen, P. L. Boveng, J. M. Breiwick, M. F. Cameron, P. J. Clapham, S. P. Dahle, M. E. Dahlheim, B. S. Fadely, M. C. Ferguson, L. W. Fritz, R. C. Hobbs, Y. V. Ivashchenko, A. S. Kennedy, J. M. London, S. A. Mizroch, R. R. Ream, E. L. Richmond, K. E. W. Shelden, R. G. Towell, P. R. Wade, J. M. Waite, and A. R. Zerbini. (2017). *Alaska Marine Mammal Stock Assessments, 2016* (NOAA Technical Memorandum NMFS-AFSC-323). Seattle, WA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center.

Muto, M. M., V. T. Helker, R. P. Angliss, P. L. Boveng, J. M. Breiwick, M. F. Cameron, P. J. Clapham, S. P. Dahle, M. E. Dahlheim, B. S. Fadely, M. C. Ferguson, L. W. Fritz, R. C. Hobbs, Y. V. Ivashchenko, A. S. Kennedy, J. M. London, S. A. Mizroch, R. R. Ream, E. L. Richmond, K. E. W. Shelden, K. L. Sweeney, R. G. Towell, P. R. Wade, J. M. Waite, and A. N. Zerbini. (2018). *Alaska Marine Mammal Stock Assessments, 2018. Draft*. Seattle, WA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center.

Muto, M. M., V. T. Helker, B. J. Delean, R. P. Angliss, P. L. Boveng, J. M. Breiwick, B. M. Brost, M. F. Cameron, P. J. Clapham, S. P. Dahle, M. E. Dahlheim, B. S. Fadely, M. C. Ferguson, L. W. Fritz, R. C. Hobbs, Y. V. Ivashchenko, A. S. Kennedy, J. M. London, S. A. Mizroch, R. R. Ream, E. L. Richmond, K. E. W. Shelden, K. L. Sweeney, R. G. Towell, P. R. Wade, J. M. Waite, and A. N. Zerbini. (2020). *Alaska Marine Mammal Stock Assessments, 2019* (NOAA Technical Memorandum NMFS-AFSC-404). Juneau, AK: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center.

Myers, J. M., R. G. Kope, G. J. Bryant, D. Teel, L. J. Lierheimer, T. C. Wainwright, W. S. Grant, F. W. Waknitz, K. Neely, S. T. Lindley, and R. S. Waples. (1998). *Status Review of Chinook Salmon from Washington, Idaho, Oregon, and California*. Seattle, WA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Myers, K. W. (2018). Ocean Ecology of Steelhead. In R. J. Beamish (Ed.), *The Ocean Ecology of Pacific Salmon and Trout* (pp. 779–904). Bethesda, MD: American Fisheries Society.

Myers, K. W., K. Y. Aydin, R. V. Walker, S. Fowler, and M. L. Dahlberg. (1996). *Known ocean ranges of stocks of Pacific salmon and steelhead as shown by tagging experiments, 1956-1995* (NPAFC Doc. 192.). Seattle, WA: University of Washington, Fisheries Research Institute.

Nadeem, K., J. E. Moore, Y. Zhang, and H. Chipman. (2016). Integrating population dynamics models and distance sampling data: A spatial hierarchial state-space approach. *Ecology 97* (7): 1735–1745.

Nagaoka, S., A. Martins, R. dos Santos, M. Tognella, E. de Oliveira Filho, and J. Seminoff. (2012). Diet of juvenile green turtles (*Chelonia mydas*) associating with artisanal fishing traps in a subtropical estuary in Brazil. *Marine Biology 159* (3): 573–581. DOI:10.1007/s00227-011-1836-y

Nakamachi, T., H. Ishida, and N. Hirohashi. (2015). Sound production in the aquatic isopod *Cymodoce japonica* (Crustacea: Peracarida). *The Biological Bulletin 229* (2): 167–172. DOI:10.1086/BBLv229n2p167

Nance, H., P. Klimley, F. Galván-Magaña, J. Martínez-Ortíz, and P. Marko. (2011). Demographic Processes Underlying Subtle Patterns of Population Structure in the Scalloped Hammerhead Shark, *Phyrna lewini*. *PLoS ONE 6* (7): e21459. DOI:10.1371/journal.pone.0021459

Napper, I. E., A. Bakir, S. J. Rowland, and R. C. Thompson. (2015). Characterisation, quantity and sorptive properties of microplastics extracted from cosmetics. *Marine Pollution Bulletin 99* (1–2): 178–185. DOI:10.1016/j.marpolbul.2015.07.029

Narazaki, T., K. Sato, K. J. Abernathy, G. J. Marshall, and N. Miyazaki. (2013). Loggerhead turtles (*Caretta caretta*) use vision to forage on gelatinous prey in mid-water. *PLoS ONE 8* (6): e66043. DOI:10.1371/journal.pone.0066043

Naslund, N. (1993). Why do marbled murrelets attend old-growth forest nesting areas year-round? *The Auk 110* (3): 594–602.

National Centers for Coastal Ocean Science. (2010). *Economic Impacts of Harmful Algal Blooms*. Silver Spring, MD: Center for Sponsored Coastal Ocean Research. Retrieved from http://www.cop.noaa.gov/stressors/extremeevents/hab/current/econimpact_08.pdf.

National Geographic. (2016). *Information on Deep-vent Eelpouts*. Retrieved September 22, 2016, from http://environment.nationalgeographic.com/environment/habitats/deep-sea-vents/.

National Marine Fisheries Service. (1976). *Hawaiian Monk Seal Final Regulations* (Federal Register).

National Marine Fisheries Service. (1991). *Final recovery plan for the humpback whale (Megaptera novaeangliae)*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

National Marine Fisheries Service. (2002). *Final Recovery Plan for Johnson's Seagrass (Halophia johnsonni)*. Silver Spring, MD: National Marine Fisheries Service.

National Marine Fisheries Service. (2005). *Green Sturgeon (Acipenser medirostris) Status Review Update*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

National Marine Fisheries Service. (2006). Notice of Availability of New Criteria for Designation of Marine Mammal Unusual Mortality Events (UMEs). *Federal Register 71 FR 75234*.

National Marine Fisheries Service. (2007a). *Conservation Plan for the Eastern Pacific Stock of Northern Fur Seal (Callorhinus ursinus)*. Juneau, AK: NMFS Protected Resources Division, Alaska Region.

National Marine Fisheries Service. (2007b). *Recovery Plan for the Hawaiian Monk Seal (Monachus schauinslandi)*. Silver Spring, MD: National Marine Fisheries Service. Retrieved from www.nmfs.noaa.gov/pr/species/mammals/pinnipeds/hawaiianmonkseal.htm.

National Marine Fisheries Service. (2007c). *Report to Congress on the Impact of Hurricanes Katrina, Rita, and Wilman on Commercial and Recreational Fisheries Habitat of Alabama, Florida, Louisiana, Mississippi, and Texas*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service. (2008). *Recovery Plan for Southern Resident Killer Whales (Orcinus orca)*. Seattle, WA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Region.

National Marine Fisheries Service. (2009a). *Endangered and Threatened Species: 12–Month Finding for a Petition to Revise Critical Habitat for Hawaiian Monk Seal* (Federal Register).

National Marine Fisheries Service. (2009b). *Sperm Whale (Physeter macrocephalus): 5-Year Review: Summary and Evaluation*. Silver Spring, MD: National Marine Fisheries Service Office of Protected Resources.

National Marine Fisheries Service. (2010a). *Cruise Report: Oscar Elton Sette, Cruise SE-10-01 (SE-77)* (PIFSC Cruise Report CR-10-006). Honolulu, HI: Pacific Island Fisheries Science Center.

National Marine Fisheries Service. (2010b). *Final Recovery Plan for the Fin Whale (Balaenoptera physalus)*. Silver Spring, MD: National Marine Fisheries Service.

National Marine Fisheries Service. (2010c). *Final Recovery Plan for the Sperm Whale (Physeter macrocephalus)*. Silver Spring, MD.

National Marine Fisheries Service. (2010d). *Hawaiian Monk Seal Population and Location*. Retrieved from http://www.fpir.noaa.gov/Library/PRD/Hawaiian%20monk%20seal/Fact%20Sheets/HMS-populationMAY2010.pdf).

National Marine Fisheries Service. (2010e). *Hawaiian Monk Seal Top Threats*. Retrieved from http://www.fpir.noaa.gov/Library/PRD/Hawaiian%20monk%20seal/Fact%20Sheets/HMS-top_threats.MAY2010.pdf.

National Marine Fisheries Service. (2010f). *Notice of Intent to Prepare a Programmatic Environmental Impact Statement on Implementing Recovery Actions for Hawaiian Monk Seals* (Federal Register).

National Marine Fisheries Service. (2010g). *Steelhead Trout (Oncorhynchus mykiss)*. Silver Spring, MD: National Oceanic and Atmospheric Administration Fisheries Office of Protected Resources. Retrieved from http://www.nmfs.noaa.gov/pr/species/fish/steelhead-trout.htm.

National Marine Fisheries Service. (2011a). *Final Recovery Plan for the Sei Whale (Balaenoptera borealis)*. Silver Spring, MD: National Marine Fisheries Service Office of Protected Resources.

National Marine Fisheries Service. (2011b). *Hawaiian Monk Seal Recovery; 2009–2010 Program Update and Accomplishments Report* (NOAA Pacific Service; Pacific Islands Region).

National Marine Fisheries Service. (2011c). *Pacific Science Center Stranding Data. Excel file containing stranding from the Hawaiian Islands, manuscript on file*.

National Marine Fisheries Service. (2011d). *Petition to List the Scalloped Hammerhead Shark (Sphyrna lewini) Under the U.S. Endangered Species Act Either Worldwide or as One or More Distinct Population Segments*. Silver Spring, MD: National Marine Fisheries Service.

National Marine Fisheries Service. (2011e). *Sea Turtles and the Gulf of Mexico Oil Spill*. Retrieved from http://www.nmfs.noaa.gov/pr/health/oilspill/turtles.htm.

National Marine Fisheries Service. (2012a). Endangered and Threatened Species: Final Rule to Revise the Critical Habitat Designation for the Endangered Leatherback Sea Turtle. *Federal Register 77* (17): 4170–4201.

National Marine Fisheries Service. (2012b). *Final Recovery Plan for Central California Coast coho salmon Evolutionarily Significant Unit*. Santa Rosa, CA: National Marine Fisheries Service, Southwest Region.

National Marine Fisheries Service. (2012c). *Southern California Steelhead Recovery Plan Summary*. Long Beach, CA.

National Marine Fisheries Service. (2013a). *Occurrence of Western Distinct Population Segment Steller Sea Lions East of 144° W. Longitude*. Silver Spring, MD: National Oceanic and Atmospheric Administration.

National Marine Fisheries Service. (2013b). *South-Central California Coast Steelhead Recovery Plan*. Long Beach, CA: National Marine Fisheries Service, West Coast Region, California Coastal Area Office.

National Marine Fisheries Service. (2014a). *California Eelgrass Mitigation Policy and Implementing Guidelines*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2014b). *Deepwater Horizon Oil Spill 2010: Sea Turtles, Dolphins, and Whales*. Retrieved from https://www.fisheries.noaa.gov/national/marine-life-distress/deepwater-horizon-oil-spill-2010-sea-turtles-dolphins-and-whales.

National Marine Fisheries Service. (2014c). *Final Recovery Plan for the Southern Oregon/Northern California Coast Evolutionarily Significant Unit of Coho Salmon (Oncorhynchus kisutch)*. Arcata, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2014d). *Recovery Plan for the Evolutionarily Significant Units of Sacramento River Winter-Run Chinook Salmon and Central Valley Spring-Run Chinook Salmon and the Distinct Population Segment of California Central Valley Steelhead*. Sacramento, CA: National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2014e). *Recovery Plan for the Evolutionarily Significant Units of Sacramento River Winter-Run Chinook Salmon and Central Valley Spring-Run Chinook Salmon and the Distinct Population Segment of California Central Valley Steelhead*. Sacramento, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2014f). *Reinitiated Biological Opinion on Navy activities on the Northwest Training Range Complex and NMFS's Issuance of an MMPA Letter of Authorization*.

National Marine Fisheries Service. (2015a). *Guadalupe Fur Seal Unusual Mortality Event in California*. Retrieved 1/12/2016, from http://www.nmfs.noaa.gov/pr/health/mmume/guadalupefurseals2015.html.

National Marine Fisheries Service. (2015b). *Status Review Update for Pacific Salmon and Steelhead Listed Under the Endangered Species Act: Pacific Northwest*. Seattle, WA: Northwest Fisheries Science Center.

National Marine Fisheries Service. (2015c). *Stranding Data for Hawaii through 2015*.

National Marine Fisheries Service. (2016a). *5-Year Review: Summary & Evaluation of Central California Coast Coho Salmon*. Santa Rosa, CA: National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2016b). *5-Year Review: Summary & Evaluation of Oregon Coast Coho Salmon*. Portland, OR: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2016c). *5-Year Review: Summary and Evaluation California Central Valley Steelhead Distinct Population Segment*. Portland, OR: National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2016d). *5-Year Review: Summary and Evaluation of Central Valley Spring-run Chinook Salmon Evolutionarily Significant Unit*. Portland, OR: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2016e). *5-Year Review: Summary and Evaluation of South-Central California Coast Steelhead Distinct Population Segment*. Santa Rosa, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Region California Coastal Office, Santa Rosa, California.

National Marine Fisheries Service. (2016f). *5-Year Status Review: Summary and Evaluation of Sacramento River Winter-Run Chinook Salmon ESU*. Sacramento, CA: National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2016g). *2016 5-Year Review: Summary & Evaluation of California Coastal Chinook Salmon and Northern California Steelhead*. Arcata, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2016h). *2016 5-Year Review: Summary & Evaluation of Southern Oregon/Northern California Coast Coho Salmon*. Arcata, CA: National Marine Fisheries Service West Coast Region.

National Marine Fisheries Service. (2016i). *Coastal Multispecies Recovery Plan - Vol III - Northern California Steelhead*. Santa Rosa, CA: National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2016j). Endangered and Threatened Species; Identification of 14 Distinct Population Segments of the Humpback Whale (*Megaptera novaeangliae*) and Revision of Species-Wide Listing. *Federal Register 81* (174): 62260–62320.

National Marine Fisheries Service. (2016k). *FAQs: Whale, Dolphin, Seal, and Sea Lion (Marine Mammal) Strandings*. Retrieved 2/27/2017, from http://www.nmfs.noaa.gov/pr/health/faq.htm.

National Marine Fisheries Service. (2016l). *Final Coastal Multispecies Recovery Plan: California Coastal Chinook Salmon, Northern California Steelhead, Central California Coast Steelhead*. Sacramento, CA: National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2016m). *Final ESA Recovery Plan for Oregon Coast Coho Salmon (Oncorhynchus kisutch)*. Portland, OR: National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2016n). *Final ESA Recovery Plan for Oregon Coast Coho Salmon (Oncorhynchus kisutch)*. Portland, OR: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2016o). *Guidelines for Preparing Stock Assessment Reports Pursuant to Section 117 of the Marine Mammal Protection Act*. Silver Spring, MD: National Oceanic and Atmospheric Administration.

National Marine Fisheries Service. (2016p). *Main Hawaiian Islands Monk Seal Management Plan*. Honolulu, HI: National Marine Fisheries Service, Pacific Islands Region.

National Marine Fisheries Service. (2016q). *National Marine Fisheries Service, Alaska Region Occurrence of Endangered Species Act (ESA) Listed Humpback Whales off Alaska*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Region.

National Marine Fisheries Service. (2016r). *Salmon & Steelhead 2016 Status Reviews*. Retrieved from http://www.westcoast.fisheries.noaa.gov/publications/status_reviews/salmon_steelhead/2016_status_review.html.

National Marine Fisheries Service. (2016s). *South-Central/Southern California Coast Steelhead Recovery Planning Domain: 5-Year Review: Summary and Evaluation of Southern California Coast Steelhead Distinct Population Segment*. Long Beach, CA.

National Marine Fisheries Service. (2016t). *Southern Resident Killer Whales (Orcinus orca) 5-Year Review: Summary and Evaluation*. Seattle, WA: National Oceanic and Atmospheric Administration, West Coast Region.

National Marine Fisheries Service. (2016u). *Species in the Spotlight: Pacific Leatherback 5-Year Action Plan*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service. (2016v). *Status of Stocks 2015* (Annual Report to Congress on the Status of U.S. Fisheries). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service. (2017). *Endangered Species Act Recovery Plan for the Southern Distinct Population Segment of Eulachon (Thaleichthys pacificus)*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2018a). *Black Abalone (Haliotis cracherodii) Five-Year Status Review: Summary and Evaluation*. Long Beach, CA: National Marine Fisheries Service, West Coast Region, Protected Resources Division.

National Marine Fisheries Service. (2018b). *Recovery Plan for the Southern Distinct Population Segment of North American Green Sturgeon (Acipenser medirostris)*. Sacramento, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Region, California Central Valley Office.

National Marine Fisheries Service. (2018c). *Sea Lion Breeding Shifts North to San Francisco Bay Area Islands*. Retrieved from https://swfsc.noaa.gov/news.aspx?ParentMenuId=147&id=22976&utm_medium=email&utm_source=govdelivery.

National Marine Fisheries Service. (2018d). *Status of Stocks 2017* (Annual Report to Congress on the Status of U.S. Fisheries). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service. (2018e). *White Abalone (Haliotis sorenseni) Five-Year Status Review: Summary and Evaluation*. Long Beach, CA: National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2019a). *2015–2019 Guadalupe Fur Seal Unusual Mortality Event in California*. Retrieved from www.fisheries.noaa.gov/national/marine-life-distress/2015-2018-guadalupe-fur-seal-unusual-mortality-event-california.

National Marine Fisheries Service. (2019b, June 3). *2019 Gray Whale Unusual Mortality Event Along the West Coast*. Retrieved June 5, 2019, from https://www.fisheries.noaa.gov/national/marine-life-distress/2019-gray-whale-unusual-mortality-event-along-west-coast.

National Marine Fisheries Service. (2020a). *Central Gulf of Alaska Marine Heatwave Watch*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service. (2020b). *Final Endangered Species Act (ESA) Recovery Plan for Black Abalone (Haliotis cracherodii)*. Long Beach, CA: National Marine Fisheries Service, West Coast Region, Protected Resources Division.

National Marine Fisheries Service. (2020c). *Recovery Plan for the Blue Whale (Balaenoptera musculus) - First Revision*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

National Marine Fisheries Service. (2021a). *Final Endangered Species Act Recovery Plan for the Main Hawaiian Islands Insular False Killer Whale (Pseudorca crassidens) Distinct Population Segment*. Honolulu, HI: National Oceanic and Atmospheric Administration, National Marina Fisheries Service, Pacific Islands Regional Office.

National Marine Fisheries Service. (2021b). *Southern Distinct Population Segment of North American Green Sturgeon (Acipenser medirostris), 5-Year Review: Summary and Evaluation*. Sacramento, CA: National Marine Fisheries Service, California Central Valley Office.

National Marine Fisheries Service. (2021c). *Status of Stocks 2020: Annual Report to Congress on the Status of U.S. Fisheries*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service. (2022a). *5-Year Review: Summary & Evaluation of Eulachon, Southern DPS*. Long Beach, CA: National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2022b). *A Cautionary Tale: the 2019 Coral Bleaching Event in Hawaii*. Retrieved July 1, 2024, from https://www.fisheries.noaa.gov/feature-story/cautionary-tale-2019-coral-bleaching-event-hawaii.

National Marine Fisheries Service. (2023a). *5-Year Review: Central California Coast Coho Salmon*. Portland, OR: National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2023b). *5-Year Review: Summary & Evaluation of South-Central California Coast Steelhead*. Long Beach, CA: National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2023c). *5-Year Review: Summary & Evaluation of Southern California Steelhead*. Long Beach, CA: National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2023d). *Draft Recovery Plan for the Oceanic Whitetip Shark (Carcharhinus longimanus)*. Silver Spring, MD: National Marine Fisheries Service, Office of Protected Resources.

National Marine Fisheries Service. (2023e). *Viability assessment for Pacific salmon and steelhead listed under the Endangered Species Act: Southwest*. Santa Cruz, CA: National Marine Fisheries Service, Southwest Fisheries Science Center.

National Marine Fisheries Service. (2024a). *5-Year Review- Summary & Evaluation of Northern California Steelhead*. Portland, OR: National Marine Fisheries Service, West Coast Region.

National Marine Fisheries Service. (2024b). *5-Year Reviews for 15 Species of Indo-Pacific CoralsListed under the Endangered Species Act*. National Marine Fisheries Service, Pacific Islands Regional Office.

National Marine Fisheries Service. (2024c). *Chinook Salmon*. Retrieved January 4, 2024, from https://www.fisheries.noaa.gov/species/chinook-salmon.

National Marine Fisheries Service. (2024d). *Closure of 2019–2023 Eastern North Pacific Gray Whale Unusual Mortality Event*. Retrieved March 21, 2024, from https://www.fisheries.noaa.gov/feature-story/closure-2019-2023-eastern-north-pacific-gray-whale-unusual-mortality-event?utm_medium=email&utm_source=govdelivery.

National Marine Fisheries Service. (2024e). *ESA Recovery Plan for the Oceanic Whitetip Shark (Carcharhinus longimanus)*. Silver Spring, MD: Office of Protected Resources, National Marine Fisheries Service.

National Marine Fisheries Service. (2024f). *Green Turtle*. Retrieved January 15, 2024, from https://www.fisheries.noaa.gov/species/green-turtle.

National Marine Fisheries Service. (2024g). *Hawaiian Monk Seal*. Retrieved May 14, 2024, from https://www.fisheries.noaa.gov/species/hawaiian-monk-seal.

National Marine Fisheries Service. (2024h). *White Abalone Conservation and Management*. Retrieved January 28, 2024, from https://www.fisheries.noaa.gov/species/white-abalone/conservation-management.

National Marine Fisheries Service and Southwest Regional Office. (2008a). *White Abalone Recovery Plan (Haliotis sorenseni)*. Long Beach, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

National Marine Fisheries Service and Southwest Regional Office. (2008b). *White Abalone Recovery Plan (Haliotis sorenseni)*. Long Beach, CA: U.S. Department of Commerce, National Oceanic and Atmospheric Administration.

National Marine Fisheries Service and U. S. Fish and Wildlife Service. (1998). *Recovery Plan for U.S. Pacific Populations of the Leatherback Turtle (Dermochelys coriacea)*. Silver Spring, MD: National Marine Fisheries Service.

National Marine Fisheries Service and U. S. Fish and Wildlife Service. (2007). *Green Sea Turtle (Chelonia mydas) 5-year Review: Summary and Evaluation*. Silver Spring, MD: National Marine Fisheries Service.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (1991). *Recovery Plan for U.S. Populations of Atlantic Green Turtle (Chelonia mydas)*. Washington, DC: National Marine Fisheries Service.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (1993). *Recovery Plan for the Hawksbill Turtle (Eretmochelys imbricata) in the U.S. Caribbean, Atlantic and Gulf of Mexico* St. Petersburg, FL: National Marine Fisheries Service.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (1998). *Recovery Plan for U.S. Pacific Populations of the Olive Ridley Turtle (Lepidochelys olivacea)*. Silver Spring, MD: National Marine Fisheries Service.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2005). *Final Recovery Plan for the Gulf of Maine Distinct Population Segment of Atlantic Salmon (Salmo salar)*. Silver Spring, MD: National Marine Fisheries Service.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2011). *Bi-National Recovery Plan for the Kemp's Ridley Sea Turtle (Lepidochelys kempii)*. Silver Spring, MD: U.S. Department of Commerce, National Oceanic and Atmospheric Administration.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2013). *Leatherback Turtle (Dermochelys coriacea) 5-Year Review: Summary and Evaluation*. Silver Spring, MD: National Marine Fisheries Service Office of Protected Resources and U.S. Fish and Wildlife Service Southeast Region.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2014). *Olive Ridley Sea Turtle (Lepidochelys olivacea) 5-Year Review: Summary and Evaluation*. Silver Spring, MD: National Marine Fisheries Service Office of Protected Resources and U.S. Fish and Wildlife Service Southeast Region.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2020a). *5-Year Review: Loggerhead sea turtle, North Pacific Ocean DPS (Caretta caretta)*. Silver Spring, MD, and Jacksonville, FL: NOAA and USFWS.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2020b). *Endangered Species Act status review of the leatherback turtle (Dermochelys coriacea)* (Report to the National Marine Fisheries Service Office of Protected Resources and U.S. Fish and Wildlife Service). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service; and the U.S. Fish and Wildlife Service.

National Ocean and Atmospheric Administration. (2016, February 25, 2016). *Deep-Sea Coral Ecosystems*. Retrieved June 13, 2016, from https://www.fisheries.noaa.gov/national/habitat-conservation/deep-sea-coral-habitat.

National Ocean Service. (2016a, March 17, 2016). *What is coral bleaching?* Retrieved 02/15/2017, 2017, from http://oceanservice.noaa.gov/facts/coral_bleach.html.

National Ocean Service. (2016b). *Where Are Reef Building Corals Found*. Retrieved March 20, 2017, 2017, from http://oceanservice.noaa.gov/education/tutorial_corals/coral05_distribution.html.

National Oceanic and Atmospheric Administration. (1985). *Threatened Fish and Wildlife; Guadalupe Fur Seal Final Rule* (Federal Register).

National Oceanic and Atmospheric Administration. (1999). *Sediment Quality Guidelines Developed for the National Status and Trends Program*. Silver Spring, MD: National Oceanic and Atmospheric Administration.

National Oceanic and Atmospheric Administration. (2008a, March 25, 2008). *Corals: Anthropogenic Threats to Corals*. Retrieved June 21, 2016, from http://oceanservice.noaa.gov/education/kits/corals/coral09_humanthreats.html.

National Oceanic and Atmospheric Administration. (2008b, March 25, 2008). *Corals: Natural Threats to Coral Reefs*. Retrieved June 21, 2016, from http://oceanservice.noaa.gov/education/kits/corals/coral08_naturalthreats.html.

National Oceanic and Atmospheric Administration. (2011). *Marine Aquaculture Policy*. Silver Spring, MD: National Oceanic and Atmospheric Administration. Retrieved from https://www.avma.org/Advocacy/National/Federal/Documents/noaa_aquaculture_policy_factsheet2011.pdf.

National Oceanic and Atmospheric Administration. (2012). *Taking of Marine Mammals Incidental to Commercial Fishing Operations; False Killer Whale Take Reduction Plan; Final Rule*: Federal Register.

National Oceanic and Atmospheric Administration. (2014). *Crude oil causes developmental abnormalities in large marine fish: Study shows Deepwater Horizon oil disrupts heart development in tunas*. Retrieved from http://www.noaanews.noaa.gov/stories2014/20140324_dwh_fishimpact.html.

National Oceanic and Atmospheric Administration. (2015a). *Endangered and Threatened Species: Final Rulemaking To Revise Critical Habitat for Hawaiian Monk Seals; Final Rule*: Federal Register.

National Oceanic and Atmospheric Administration. (2015b, 10/8/2015). *NOAA declares third ever global coral bleaching event*. Retrieved 08/23/2016, 2016, from http://www.noaanews.noaa.gov/stories2015/100815-noaa-declares-third-ever-global-coral-bleaching-event.html.

National Oceanic and Atmospheric Administration. (2015c). Takes of Marine Mammals Incidental to Specified Activities; Taking Marine Mammals Incidental to a Pier Maintenance Project. *Federal Register 80* (228): 74076–74085.

National Oceanic and Atmospheric Administration. (2015d). *Vaccination of Hawaiian Monk Seals*. Honolulu, HI: National Marine Fisheries Service.

National Oceanic and Atmospheric Administration. (2016a). *About the National Marine Aquaculture Initiative*. Retrieved March 9, 2016, from http://www.nmfs.noaa.gov/aquaculture/funding/nmai.html.

National Oceanic and Atmospheric Administration. (2016b). *Discover the Issue: Marine Debris*. Retrieved September 12, 2016, from https://marinedebris.noaa.gov/discover-issue.

National Oceanic and Atmospheric Administration. (2016c). *Harmful Algal Blooms Observing System*. Retrieved March 24, 2016, from http://habsos.noaa.gov/.

National Oceanic and Atmospheric Administration. (2016d, June 28). *Manta rays (Manta spp.)*. Retrieved July 26, 2016, from https://www.fisheries.noaa.gov/species/giant-manta-ray.

National Oceanic and Atmospheric Administration. (2016e, January 12). *Oceanic Whitetip Shark (Carcharhinus longimanus)*. Retrieved July 21, 2016, from https://www.fisheries.noaa.gov/species/oceanic-whitetip-shark.

National Oceanic and Atmospheric Administration. (2017a, August 15). *2013–2017 California Sea Lion Unusual Mortality Event in California*. Retrieved May 1, 2018, from https://www.fisheries.noaa.gov/national/marine-life-distress/2013-2017-california-sea-lion-unusual-mortality-event-california.

National Oceanic and Atmospheric Administration. (2017b). *Maritime Heritage in Papahanaumokuakea Marine National Monument*. Retrieved from https://www.papahanaumokuakea.gov/maritime/.

National Oceanic and Atmospheric Administration. (2018a, January 4). *2015–2018 Guadalupe Fur Seal Unusual Mortality Event in California*. Retrieved May 1, 2018, from https://www.fisheries.noaa.gov/national/marine-life-distress/2015-2018-guadalupe-fur-seal-unusual-mortality-event-california.

National Oceanic and Atmospheric Administration. (2018b, December 22, 2016). *White Abalone (Haliotis sorenseni)*. Retrieved 02/22/2017, 2017, from https://www.fisheries.noaa.gov/species/white-abalone.

National Oceanic and Atmospheric Administration. (2020, February 12). *2019-2020 Gray Whale Unusual Mortality Event along the West Coast*. Retrieved February 19, 2020, from https://www.fisheries.noaa.gov/national/marine-life-distress/2019-2020-gray-whale-unusual-mortality-event-along-west-coast.

National Oceanic and Atmospheric Administration. (2022). *A Cautionary Tale: The 2019 Coral Bleaching Event in Hawaii*.

National Oceanic and Atmospheric Administration. (2024a). *Hawksbill Turtle*. Retrieved March 12, 2024, from https://www.fisheries.noaa.gov/species/hawksbill-turtle.

National Oceanic and Atmospheric Administration. (2024b). *NOAA confirms 4th global coral bleaching Event*. Retrieved July 11, 2024, from https://www.noaa.gov/news-release/noaa-confirms-4th-global-coral-bleaching-event.

National Oceanic and Atmospheric Administration and National Marine Fisheries Service. (2008). *Habitat Connections: Deep Sea Corals*. Washington, DC: U.S. Department of Commerce.

National Oceanic and Atmospheric Administration and NOAA's Coral Reef Conservation Program. (2016, June 6, 2016). *Coral Facts*. Retrieved June 28, 2016, from http://coralreef.noaa.gov/education/coralfacts.html.

National Oceanic and Atmospheric Administration Fisheries. (2014). *Southern Resident Killer Whales: 10 Years of Research & Conservation*. Seattle, WA: Northwest Fisheries Science Center West Coast Region.

National Oceanic and Atmospheric Administration Fisheries. (2015, January 27, 2015). *Invertebrates and Plants. Office of Protected Resources*. Retrieved June 23, 2016, from http://www.nmfs.noaa.gov/pr/species/invertebrates/.

National Oceanic and Atmospheric Administration Fisheries. (2018). *Black Abalone (Haliotis cracherodii)*. Retrieved from https://www.st.nmfs.noaa.gov/ecosystems/lme/index.

National Oceanic and Atmospheric Administration Fisheries and Southwest Fisheries Science Center. (2010, October 15, 2010). *Deep sea coral communities and fisheries habitats off California*. Retrieved October 1, 2015, from https://swfsc.noaa.gov/textblock.aspx?Division=FED&ParentMenuId=54&id=16614.

National Oceanic and Atmospheric Administration Marine Debris Program. (2014). *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.

NatureServe. (2004, 23 November 2004). *Comprehensive report: Phoebastria albatrus—(Pallas, 1769): Short-tailed albatross*. Retrieved from http://www.natureserve.org.

Naval Facilities Engineering Command Pacific. (2018). *Final Record of Decision Pearl Harbor Sediment, Joint Base Pearl Harbor-Hickam Oahu Hawaii*. Honolulu, HI: Naval Facilities Engineering Command.

Neira, C., F. Delgadillo-Hinojosa, A. Zirino, G. Mendoza, L. A. Levin, M. Porrachia, and D. D. Deheyn. (2009). Spatial distribution of copper in relation to recreational boating in a California shallow-water basin. *Chemistry and Ecology 25* (6): 417–433.

Nelms, S. E., J. Alfaro-Shigueto, J. P. Y. Arnould, I. C. Avila, S. B. Nash, E. Campbell, M. I. D. Carter, T. Collins, R. J. C. Currey, C. Domit, V. Franco-Trecu, M. M. P. B. Fuentes, E. Gilman, R. G. Harcourt, E. M. Hines, A. R. Hoelzel, S. K. Hooker, D. W. Johnston, N. Kelkar, J. J. Kiszka, K. L. Laidre, J. C. Mangel, H. Marsh, S. M. Maxwell, A. B. Onoufriou, D. M. Palacios, G. J. Pierce, L. S. Ponnampalam, L. J. Porter, D. J. F. Russell, K. A. Stockin, D. Sutaria, N. Wambiji, C. R. Weir, B. Wilson, and B. J. Godley. (2021). Marine mammal conservation: over the horizon. *Endangered Species Research 44* 291–325.

Nelson, J. S., E. J. Crossman, H. Espinosa-Pérez, L. T. Findley, C. R. Gilbert, R. N. Lea, and J. D. Williams. (2004). Common and scientific names of fishes from the United States, Canada, and Mexico. In: *American Fisheries Society Special Publication 29* (6th ed.). Bethesda, MD: American Fisheries Society.

Nelson, J. S., T. C. Grande, and M. V. H. Wilson. (2016). *Fishes of the World* (5th ed.). Hoboken, NJ: John Wiley & Sons, Inc.

Nelson, S. K. (1997). *Marbled Murrelet (Brachyramphus marmoratus)*. *The Birds of North America Online.* Retrieved 276, from https://birdsna.org/Species-Account/bna/species/marmur/introduction.

Netburn, A. N. and J. A. Koslow. (2015). Dissolved oxygen as a constraint on daytime deep scattering layer depth in the Southern California current ecosystem. *Deep-Sea Research I 104* 149–158.

Neuman, M., B. Tissot, and G. VanBlaricom. (2010). Overall status and threats assessment of black abalone (*Haliotis cracherodii* Leach, 1814) populations in California. *Journal of Shellfish Research 29* (3): 577–586.

Newman, M. C. (1998). Uptake, biotransformation, detoxification, elimination, and accumulation. In M. C. Newman (Ed.), *Fundamentals of Ecotoxicology* (pp. 25). Chelsea, MI: Ann Arbor Press.

NOAA Fisheries Protected Resources Division. (2007). *Final Assessment of NOAA Fisheries' Critical Habitat Analytical Review Team (CHART) For the Oregon Coast Coho Salmon Evolutionarily Significant Unit*. Portland, OR: National Marine Fisheries Service.

Normandeau Associates, Inc. (2012). *Effects of Noise on Fish, Fisheries, and Invertebrates in the U.S. Atlantic and Arctic from Energy Industry Sound-Generating Activities*. Bedford, NH: US Department of the Interior, Bureau of Ocean Energy Management, Regulation and Enforcement, Pacific OCS Region.

Normandeau Associates, Inc. and APEM, Ltd. (2013a). *Aerial Survey of Seabirds and Marine Mammals at Ka`ula Island, Hawaii—August 2013* (GSA #GS-10F-0319M). Gainesville, FL: U.S. Navy Commander, U.S. Pacific Fleet Environmental Readiness.

Normandeau Associates, Inc. and APEM, Ltd. (2013b). *Aerial Survey of Seabirds and Marine Mammals at Ka`ula Island, Hawaii—Spring 2013* (GSA # GS-10F-0319M). Gainesville, FL: U.S. Navy Commander, U.S. Pacific Fleet Environmental Readiness.

Normandeau Associates, Inc. and APEM, Ltd. (2014). *Aerial Survey of Seabirds and Marine Mammals at Ka`ula Island, Hawaii—January 2014* (GSA # GS-10F-0319M). Gainesville, FL: U.S. Navy Commander, U.S. Pacific Fleet Environemtnal Readiness.

Normandeau Associates, Inc. and APEM, Ltd. (2015a). *Aerial Survey of Seabirds and Marine Mammals at Ka`ula Island, Hawaii—January 2015* (GSA #GS-10F-0319M). Gainesville, FL: U.S. Navy Commander, U.S. Pacific Fleet Environmental Readiness.

Normandeau Associates, Inc. and APEM, Ltd. (2015b). *Aerial Survey of Seabirds and Marine Mammals at Ka`ula Island, Hawaii—July 2015* (GSA #GS-10F-0319M). Gainesville, FL: U.S. Navy Commander, U.S. Pacific Fleet Environmental Readiness.

Normandeau Associates, Inc. and APEM, Ltd. (2016). *Aerial Survey of Seabirds and Marine Mammals at Ka`ula Island, Hawaii—March 2016* (GSA #GS-10F-0319M). Gainesville, FL: U.S. Navy Commander, U.S. Pacific Fleet Environmental Readiness.

Norris, K. S. and T. P. Dohl. (1980). Behavior of the Hawaiian spinner dolphin, *Stenella longirostris*. *Fishery Bulletin 77* (4): 821–849.

Norris, T. (2019). *Guadalupe Fur Seal Population Census and Tagging in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas in the Pacific Ocean*. Sausalito, CA: The Marine Mammal Center.

Norris, T. (2022, Junuary 20, 2022). Personal communication via email between Tenaya Norris (The Marine Mammal Center) and Michael Zickel (ManTech International) regarding updated abundance estimate for Guadalupe fur seals off California.

Norris, T. A. and F. R. Elorriaga-Verplancken. (2020). *Guadalupe Fur Seal Population Census and Tagging in Support of Marine Mammal Monitoring Across Multiple Navy Training Areas in the Pacific Ocean*. Sausalito, CA: The Marine Mammal Center.

Norris, T. F., M. McDonald, and J. Barlow. (1999). Acoustic detections of singing humpback whales (*Megaptera novaeangliae*) in the eastern North Pacific during their northbound migration. *The Journal of Acoustical Society of America 106* (1): 506–514.

North American Bird Conservation Initiative. (2022). *The State of the Birds, United States of America, 2022*. Washington, DC: North American Bird Conservation Initiative.

North American Bird Conservation Initiative and U.S. Committee. (2010). *The State of the Birds: 2010 Report on Climate Change, United States of America*. Washington, DC: U.S. Department of the Interior.

North American Bird Conservation Initiative U.S. Committee. (2009). *The State of the Birds, United States of America, 2009*. Washington, DC: U.S. Department of Interior. Retrieved from http://www.stateofthebirds.org/pdf_files/State_of_the_Birds_2009.pdf.

Noss, R. F., J. M. Cartwright, D. Estes, T. Witsell, G. Elliott, D. Adams, M. Albrecht, R. Boyles, P. Comer, C. Foffitt, D. Faber-Langendoen, J. V. Hill, W. C. Hunter, W. M. Knapp, M. E. Marshall, J. Singhurst, C. Tracey, J. Walck, and A. Weakley. (2020). Improving species status assessments under the U.S. Endangered Species Act and implications for multispecies conservation challenges worldwide. *Conservation Biology 2021* (35): 1715–1724. DOI:10.1111/cobi.13777

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'Farrell, M., M. Mohr, A. Grover, and W. Satterthwaite. (2012). *Sacramento River winter Chinook salmon cohort reconstruction: analysis of ocean fishery impacts.* Santa Cruz, CA: NOAA National Marine Fisheries Service, SWFSC Fisheries Ecology Division.

Ocean Conservancy. (2010a). *BP Oil Disaster: Relief, Restoration, and Reform*. *Our Work.* Retrieved from http://www.oceanconservancy.org/our-work/bp-oil-spill/.

Ocean Conservancy. (2010b). *Trash Travels: From Our Hands to the Sea, Around the Globe, and Through Time* (International Coastal Cleanup Report). Washington, DC: Ocean Conservancy.

Office of National Marine Sanctuaries. (2019). *Channel Islands National Marine Sanctuary 2016 Condition Report*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Ocean Service, National Marine Sanctuary Program.

Office of the Washington Governor. (2018). *Southern Resident Orca Task Force: Draft Report and Potential Recommendations*. Olympia, WA: State of Washington.

Ohizumi, H. (2002). Dietary studies of toothed whales: A review of technical issues and new topics. *Fisheries Science 68* (Supplement 1): 264–267.

Ohizumi, H., T. Matsuishi, and H. Kishino. (2002). Winter sightings of humpback and Bryde's whales in tropical waters of the western and central North Pacific. *Aquatic Mammals 28* (1): 73–77.

Oleson, E. M., R. W. Baird, K. K. Martien, and B. L. Taylor. (2013). *Island-associated stocks of odontocetes in the main Hawaiian Islands: A synthesis of available information to facilitate evaluation of stock structure* (PIFSC Working Paper WP-13-003). Pacific Islands Fisheries Science Center.

Oleson, E. M., S. Baumann-Pickering, A. Širović, K. P. Merkens, L. M. Munger, J. S. Trickey, and P. Fisher-Pool. (2015). *Analysis of long-term acoustic datasets for baleen whales and beaked whales within the Mariana Islands Range Complex (MIRC) for 2010 to 2013* (PIFSC Data Report DR-15-002).

Oleson, E. M., C. H. Boggs, K. A. Forney, M. B. Hanson, D. R. Kobayashi, B. L. Taylor, P. R. Wade, and G. M. Ylitalo. (2010a). *Status Review of Hawaiian Insular False Killer Whales (Pseudorca crassidens) under the Endangered Species Act*. U.S. Department of Commerce.

Oleson, E. M., C. H. Boggs, K. A. Forney, M. B. Hanson, D. R. Kobayashi, B. L. Taylor, P. R. Wade, and G. M. Ylitalo. (2010b). *Status Review of Hawaiian Insular False Killer Whales (Pseudorca crassidens) under the Endangered Species Act* (NOAA Technical Memorandum NMFS-PIFSC-22). Honolulu, HI: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Oleson, E. M., J. Calambokidis, E. Falcone, G. Schorr, and J. A. Hildebrand. (2009). *Acoustic and visual monitoring for cetaceans along the outer Washington coast*. San Diego, CA: University of California San Diego Scripps Institution of Oceanography and Cascadia Research Collective.

Olsen, Y. S., M. Sanchez-Camacho, N. Marba, and C. M. Duarte. (2012). Mediterranean seagrass growth and demography responses to experimental warming. *Estuaries and Coasts 35* (5): 1205–1213. DOI:10.1007/s12237-012-9521-z

Olson, P. A. (2009). Pilot whales, *Globicephala melas* and *G. macrorhynchus*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 898–903). Cambridge, MA: Academic Press.

Onley, D. and P. Scofield. (2007). *Albatrosses, Petrels and Shearwaters of the World*. Princeton, NJ: Princeton University Press.

Oregon State University. (2017). *Southern and Central California 2016 Whale Approach Summary from Bruce Mate regarding body condition of blue and fin whales off Southern and Central California*. Corvallis, OR: Oregon State University.

Ormerod, S. J. (2003). Current issues with fish and fisheries: Editor's overview and introduction. *Journal of Applied Ecology 40* (2): 204–213. DOI:10.1046/j.1365-2664.2003.00824

Oros, J., A. Torrent, P. Calabuig, and S. Deniz. (2005). Diseases and causes of mortality among sea turtles stranded in the Canary Islands, Spain (1998–2001). *Diseases of Aquatic Organisms 63* 13–24.

Ortmann, A. C., J. Anders, N. Shelton, L. Gong, A. G. Moss, and R. H. Condon. (2012). Dispersed oil disrupts microbial pathways in pelagic food webs. *PLoS ONE 7* (7): e42548. DOI:10.1371/journal.pone.0042548

Ospina-Alvarez, N. and F. Piferrer. (2008). Temperature-dependent sex determination in fish revisited: prevalence, a single sex ratio response pattern, and possible effects of climate change. *PLoS ONE 3* (7): 1–11. DOI:10.1371/journal.pone.0002837

Osterback, A. M. K., C. H. Kern, E. A. Kanawi, J. M. Perez, and J. D. Kiernan. (2018). The effects of early sandbar formation on the abundance and ecology of coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Oncorhynchus mykiss*) in a central California coastal lagoon. *Canadian Journal of Fisheries and Aquatic Sciences 75* (2): 2184-2197. DOI:10.1139/cjfas-2017-0455

Ostman-Lind, J., A. D. Driscoll-Lind, and S. H. Rickards. (2004). *Delphinid Abundance, Distribution and Habitat Use Off the Western Coast of the Island of Hawaii*. La Jolla, CA: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Oswald, J. N., J. Barlow, and T. F. Norris. (2003). Acoustic identification of nine delphinid species in the eastern tropical Pacific Ocean. *Marine Mammal Science 19* (1): 20–37.

Pablo-Rodriguez, N., D. Aurioles-Gamboa, and J. L. Montero-Munoz. (2015). Niche overlap and habitat use at distinct temporal scales among the California sea lions (*Zalophus californianus*) and Guadalupe fur seals (*Arctocephalus philippii townsendi*). *Marine Mammal Science* 24. DOI:10.1111/mms.12274

Pacific Fishery Management Council. (2011, March 22, 2011). *Habitat and Communities: Wave, Tidal, and Offshore Wind Energy*. Retrieved Accessed 3 July 2012, from http://www.pcouncil.org/habitat-and-communities/wave-tidal-and-offshore-wind-energy/.

Pacific Fishery Management Council. (2014). *Pacific Coast Groundfish Fishery Management Plan for the California, Oregon and Washington Groundfish Fishery*. Portland, OR: Pacific Fishery Managment Council.

Pacific Islands Fisheries Science Center. (2014). *Reefs for the future: Resilience of coral reefs in the main Hawaiian Islands*. Honolulu, HI: National Oceanic and Atmospheric Administration Fisheries Pacific Science Center, Pacific Islands Fisheries Science Center Special Publication.

Pacific Islands Fisheries Science Center. (2018). *2018 Marine Debris Removal and Assessment in the Northwestern Hawaiian Islands*. Honolulu, HI: National Oceanic Atmospheric Administration.

Packard, A., H. E. Karlsen, and O. Sand. (1990). Low frequency hearing in cephalopods. *Journal of Comparative Physiology A 166* 501–505.

Packer, D. B., M. S. Nizinski, M. S. Bachman, A. F. Drohan, M. Poti, and B. P. Kinlan. (2017). State of the Deep-Sea Coral and Sponge Ecosystems off the Northeast United States. In T. F. Hourigan, P. J.

Etnoyer, & S. D. Cairns (Eds.), *The State of Deep-Sea Coral and Sponge Ecosystems in the United States* (Vol. NOAA Technical Memorandum NMFS-OHC-X, pp. 9–1–9–65). Silver Spring, MD: National Oceanic and Atmospheric Administration.

Palacios, D. M., H. Bailey, E. A. Becker, S. J. Bograd, M. L. DeAngelis, K. A. Forney, E. L. Hazen, L. M. Irvine, and B. R. Mate. (2019). Ecological correlates of blue whale movement behavior and its predictability in the California Current Ecosystem during the summer-fall feeding season. *Movement Ecology 7* (1). DOI:10.1186/s40462-019-0164-6

Pandolfi, J. M., R. H. Bradbury, E. Sala, T. P. Hughes, K. A. Bjorndal, R. G. Cooke, D. McArdle, L. McClenachan, M. J. H. Newman, G. Paredes, R. R. Warner, and J. B. C. Jackson. (2003). Global trajectories of the long-term decline of coral reef ecosystems. *Science 301* 955–958.

Pandolfi, J. M., J. B. C. Jackson, N. Baron, R. H. Bradbury, H. M. Guzman, T. P. Hughes, C. V. Kappel, F. Micheli, J. C. Ogden, H. P. Possingham, and E. Sala. (2005). Are U.S. coral reefs on the slippery slope to slime? *Science 307* (5716): 1725–1726.

Panigada, S., M. Zanardelli, M. Mackenzie, C. Donovan, F. Melin, and P. S. Hammond. (2008). Modelling habitat preferences for fin whales and striped dolphins in the Pelagos Sanctuary (Western Mediterranean Sea) with physiographic and remote sensing variables. *Remote Sensing of Environment 112* (8): 3400–3412. DOI:10.1016/j.rse.2007.11.017

Pansch, C., G. S. I. Hattich, M. E. Heinrichs, A. Pansch, Z. Zagrodzka, and J. N. Havenhand. (2018). Long-term exposure to acidification disrupts reproduction in a marine invertebrate. *PLoS ONE 13* (2): e0192036. DOI:https://doi.org/10.1371/journal.pone.0192036

Parin, N. V. (1984). Oceanic ichthyologeography: An attempt to review the distribution and origin of pelagic and bottom fishes outside continental shelves and neritic zones. *Fourth Congress of European Ichthyologists 35* (1): 5–41.

Parker, D. M., P. H. Dutton, K. Kopitsky, and R. L. Pitman. (2003). *Movement and Dive Behavior Determined by Satellite Telemetry for Male and Female Olive Ridley Turtles in the Eastern Tropical Pacific* (Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation): U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service. Retrieved from http://www.nmfs.noaa.gov/pr/species/turtles/symposia.htm.

Parkinson, S. C., K. Dragoon, G. Reikard, G. García-Medina, H. T. Özkan-Haller, and T. K. A. Brekken. (2015). Integrating ocean wave energy at large-scales: A study of the U.S. Pacific Northwest. *Renewable Energy 76* 551–559.

Parrish, F. A. and A. R. Baco. (2007). State of Deep Coral Ecosystems in the U.S. Pacific Islands Region: Hawaii and the U.S. Pacific Territories. In S. E. Lumsden, T. F. Hourigan, A. Bruckner, & D. G. Wand (Eds.), *The State of Deep Coral Ecosystems of the United States* (Vol. NOAA Technical Memorandum CRCP-3, pp. 155–194). Silver Spring, MD: National Oceanic and Atmospheric Administration.

Parrish, F. A., A. R. Baco, C. Kelley, and H. Reiswig. (2015a). State of Deep-Sea Coral and Sponge Ecosystems of the U.S. Pacific Islands Region: 2015. In T. F. Hourigan, P. J. Etnoyer, & S. D. Cairns (Eds.), *The State of Deep-Sea Coral and Sponge Ecosystems of the United States: 2015* (pp. 38). Silver Spring, MD: National Oceanic and Atmospheric Administration.

Parrish, F. A., A. R. Baco, C. Kelley, and H. Reiswig. (2015b). State of Deep-Sea Coral and Sponge Ecosystems of the U.S. Pacific Islands Region: 2015. In T. F. Hourigan, P. J. Etnoyer, & S. D. Cairns

(Eds.), *The State of Deep-Sea Coral and Sponge Ecosystems of the United States: 2015*. Silver Spring, MD: National Oceanic and Atmospheric Administration.

Parrish, F. A., T. F. Hourigan, A. R. Baco, C. D. Kelley, and S. D. Cairns. (2022). *List of Deep-Sea Coral Taxa in the Hawaiian Archipelago and Johnston Atoll: Depth and Geographic Distribution (v. 2021)* (Online Supplement to The state of deep-sea coral and sponge ecosystems of the United States). Honolulu, HI: National Marine Fisheries Service.

Pastran, S. A. and D. B. Lank. (2024). Considerations for a threatened seabird: The impact of shoreline avian predators on at-sea marbled murrelets. *Wildlife Society Bulletin* e1514.

Patek, S. N. and R. L. Caldwell. (2006). The stomatopod rumble: Low frequency sound production in *Hemisquilla californiensis*. *Marine and Freshwater Behaviour and Physiology 39* (2): 99–111. DOI:10.1080/10236240600563289

Patek, S. N., L. E. Shipp, and E. R. Staaterman. (2009). The acoustics and acoustic behavior of the California spiny lobster (*Panulirus interruptus*). *The Journal of the Acoustical Society of America 125* (5): 3434–3443. DOI:10.1121/1.3097760

Patel, S. H., M. V. Winton, J. M. Hatch, H. L. Haas, V. S. Saba, G. Fay, and R. J. Smolowitz. (2021). Projected shifts in loggerhead sea turtle thermal habitat in the Northwest Atlantic Ocean due to climate change. *Scientific Reports 11* (1): 8850.

Patino-Martinez, J., A. Marco, L. Quinones, and B. Godley. (2008). Globally significant nesting of the leatherback turtle (*Dermochelys coriacea*) on the Caribbean coast of Colombia and Panama. *Biological Conservation 141* (8): 1982–1988. DOI:10.1016/j.biocon.2008.05.012.

Patrício, A. R., C. E. Diez, R. P. Van Dam, and B. J. Godley. (2016). Novel insights into the dynamics of green turtle fibropapillomatosis. *Marine Ecology Progress Series 547* 247–255.

Paull, C. K., W. R. Normark, W. Ussler, III, D. W. Caress, and R. Keaten. (2008). Association among active seafloor deformation, mound formation, and gas hydrate growth and accumulation within the seafloor of the Santa Monica Basin, offshore California. *Marine Geology 250* (3–4): 258–275. DOI:10.1016/j.margeo.2008.01.011

Pauly, D. and M. L. Palomares. (2005). Fishing down marine food web: It is far more pervasive than we thought. *Bulletin of Marine Science 76* (2): 197–211.

Pawlik, J. R. and S. McMurray. (2020). The Emerging Ecological and Biogeochemical Importance of Sponges on Coral Reefs. *Annual Review of Marine Science* (12): 315–337.

Paxton, J. R. and W. N. Eshmeyer. (1998). *Encyclopedia of Fishes* (2nd ed.). San Diego, CA: Academic Press.

Payne, J., D. L. Erickson, M. Donnellan, and S. T. Lindley. (2015). *Project to Assess Potential Impacts of the Reedsport Ocean Power Technologies Wave Energy Generation Facility on Migration and Habitat use of Green Sturgeon (Acipenser medirostris)*. Portland, OR: Oregon Wave Energy Trust.

Payne, P. M. and D. W. Heinemann. (1993). The distribution of pilot whales (*Globicephala* spp.) in shelf/shelf edge and slope waters of the northeastern United States, 1978–1988. *Reports of the International Whaling Commission 14* 51–68.

Pearcy, W. G. and J. P. Fisher. (1988). Migrations of coho salmon, *Oncorhynchus kisutch,* during their first summer in the ocean. *Fishery Bulletin 86* (2): 173.

Pearse, V., J. Pearse, M. Buchsbaum, and R. Buchsbaum. (1987). *Living Invertebrates*. Palo Alto, CA: Boxwood Press.

Perea-Blázquez, A., S. K. Davy, and J. J. Bell. (2012). Estimates of particulate organic carbon flowing from the pelagic environment to the benthos through sponge assemblages. *PLoS ONE 7* (1): e29569. DOI:10.1371/journal.pone.0029569

Perrin, W. F. (1976). First record of the melon-headed whale, *Peponocephala electra*, in the eastern Pacific, with a summary of world distribution. *Fishery Bulletin 74* (2): 457–458.

Perrin, W. F. (2001). *Stenella attenuata*. *Mammalian Species 683* 1–8.

Perrin, W. F. (2008a). Common dolphins, *Delphinus delphis* and *D. capensis*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 255–259). Academic Press.

Perrin, W. F. (2008b). Pantropical spotted dolphin, *Stenella attenuata*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 819–821). Academic Press.

Perrin, W. F. (2008c). Spinner dolphin, *Stenella longirostris*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 1100–1103). Academic Press.

Perrin, W. F., C. S. Baker, A. Berta, D. J. Boness, R. L. Brownell, Jr., M. L. Dalebout, D. P. Domning, R. M. Hamner, T. A. Jefferson, J. G. Mead, D. W. Rice, P. E. Rosel, J. Y. Wang, and T. Yamada. (2009a). *Marine Mammal Species and Subspecies*. Retrieved 2010, from http://www.marinemammalscience.org/index.php?option=com_content&view=article&id=420&Itemid=280.

Perrin, W. F., P. B. Best, W. H. Dawbin, K. C. Balcomb, R. Gambell, and G. J. B. Ross. (1973). Rediscovery of Fraser's dolphin, *Lagenodelphis hosei*. *Nature 241* 345–350.

Perrin, W. F., B. Würsig, and J. G. M. E. Thewissen. (2009b). *Encyclopedia of Marine Mammals* (2nd ed.). San Diego, CA: Academic Press.

Perry, A. L., P. J. Low, J. R. Ellis, and J. D. Reynolds. (2005). Climate change and distribution shifts in marine fishes. *Science 308* 1912–1914.

Perry, S. L., D. P. DeMaster, and G. K. Silber. (1999). The great whales: History and status of six species listed as endangered under the U.S. Endangered Species Act of 1973. *Marine Fisheries Review 61* (1): 1–74.

Perryman, W. L. (2008). Melon-headed whale, *Peponocephala electra*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 719–721). Academic Press.

Perryman, W. L., D. W. K. Au, S. Leatherwood, and T. A. Jefferson. (1994). Melon-headed whale, *Peponocephala electra* Gray, 1846. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 5: The first book of dolphins, pp. 363–386). San Diego, CA: Academic Press.

Peterson, C. H. (2001). The "Exxon Valdez" oil spill in Alaska: Acute, indirect and chronic effects on the ecosystem. In A. J. Southward, P. A. Tyler, C. M. Young, & L. A. Fuiman (Eds.), *Advances in Marine Biology* (Vol. 39, pp. 1–103). San Diego, CA: Academic Press.

Peterson, S. H., J. T. Ackerman, and D. P. Costa. (2015). Marine foraging ecology influences mercury bioaccumulation in deep-diving northern elephant seals. *Proceedings of the Royal Society B: Biological Sciences 282* (20150710): 10. DOI:10.1098/rspb.2015.0710

Peterson, W., J. Fisher, J. Peterson, C. Morgan, B. Burke, and K. Fresh. (2014a). Applied Fisheries Oceanography: Ecosystem Indicators of Ocean Conditions Inform Fisheries Management in the California Current. *Oceanography 27* (4): 80–89. DOI:10.5670/oceanog.2014.88

Peterson, W. T., R. Emmett, R. Goericke, E. Venrick, A. Mantyla, S. J. Bograd, F. B. Schwing, R. Hewitt, N. Lo, W. Watson, J. Barlow, M. Lowry, S. Talston, K. A. Forney, B. E. Lavaniegos, W. J. Sydeman, D. Hyrenbach, R. W. Bradley, P. Warzybok, F. Chavez, K. Hunter, S. Benson, M. Weise, and J. Harvey. (2006). The State of the California Current, 2005–2006: Warm in the North, Cool in the South. In S. M. Shoffler (Ed.), *California Cooperative Oceanic Fisheries Investigations* (Vol. 47, pp. 30–74). La Jolla, CA: California Department of Fish and Game, University of California, Scripps Institute of Oceanography, and the National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Peterson, W. T., J. L. Fisher, J. O. Peterson, C. A. Morgan, B. J. Burke, and K. L. Fresh. (2014b). Applied fisheries oceanography: Ecosystem indicators of ocean conditions inform fisheries management in the California Current. *Oceanography 27* (4): 80–89.

Peterson, W. T., C. A. Morgan, J. O. Peterson, J. L. Fisher, B. J. Burke, and K. Fresh. (2014c). *Ocean Ecosystem Indicators of Salmon Marine Survival in the Northern California Current*. Newport, OR: National Marine Fisheries Service, Northwest Fisheries Science Center.

Pew Oceans Commission. (2003). *America's Living Oceans: Charting a Course for Sea Change*. Arlington, VA: Pew Oceans Commission.

Phillips, R. C. and E. G. Meñez. (1988). Seagrasses. *Smithsonian Contributions to the Marine Sciences 34* 104.

Phillis, C. C., A. M. Sturrock, R. C. Johnson, and P. K. Weber. (2018). Endangered winter-run Chinook salmon rely on diverse rearing habitats in a highly altered landscape. *Biological Conservation 217* 358–362.

PHYSORG. (2017). *Biologist: Orca attacks on gray whales up in California bay*. Retrieved 16 Mat 2017, from https://phys.org/news/2017-04-biologist-orca-gray-whales-california.html.

Piatt, J. F. and N. L. Naslund. (1995). Abundance, distribution, and population status of marbled murrelets in Alaska. In C. J. Ralph, G. L. Hunt, Jr., M. G. Raphael, & J. F. Piatt (Eds.), *Ecology and Conservation of the Marbled Murrelet* (pp. 285–294). Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture.

Pierce, G. J., M. B. Santos, C. Smeenk, A. Saveliev, and A. F. Zuur. (2007). Historical trends in the incidence of strandings of sperm whales (*Physeter macrocephalus*) on North Sea coasts: An association with positive temperature anomalies. *Fisheries Research 87* (2–3): 219–228. DOI:10.1016/j.fishres.2007.06.001

Pike, D. A. (2014). Forecasting the viability of sea turtle eggs in a warming world. *Global Change Biology 20* (1): 7–15.

Pike, D. A., E. A. Roznik, and I. Bell. (2015). Nest inundation from sea-level rise threatens sea turtle population viability. *Royal Society Open Science 2* (7): 150127.

Pimentel, D., R. Zuniga, and D. Morrison. (2004). Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecological Economics 52* 273–288. DOI:10.1016/j.ecolecon.2004.10.002

Pine, M. K., A. G. Jeffs, D. Wang, and C. A. Radford. (2016). The potential for vessel noise to mask biologically important sounds within ecologically significant embayments. *Ocean & Coastal Management 127* 63–73. DOI:10.1016/j.ocecoaman.2016.04.007

Pitcher, T. J. (1995). The impact of pelagic fish behavior on fisheries. *Scientia Marina 59* (3–4): 295–306.

Pitman, R. (2008). Mesoplodont whales (*Mesoplodon spp.*). In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 721–726). Amsterdam, Netherlands: Academic Press.

Pitman, R. L., D. W. K. Au, M. D. Scott, and J. M. Cotton. (1988). *Observations of Beaked Whales (Ziphiidae) from the Eastern Tropical Pacific Ocean*. Cambridge, United Kingdom: International Whaling Commission.

Pitman, R. L. and L. T. Ballance. (1992). Parkinson's petrel distribution and foraging ecology in the eastern Pacific: Aspects of an exclusive feeding relationship with dolphins. *The Condor 94* 825–835.

Pitman, R. L. and M. S. Lynn. (2001). Biological observations of an unidentified mesoplodont whale in the eastern tropical Pacific and probable identity: *Mesoplodon peruvianus*. *Marine Mammal Science 17* (3): 648–657.

Pitman, R. L. and C. Stinchcomb. (2002). Rough-toothed dolphins (*Steno bredanensis*) as predators of mahimahi (*Coryphaena hippurus*). *Pacific Science 56* (4): 447–450.

Plot, V., B. De Thoisy, and J.-Y. Georges. (2015). Dispersal and dive patterns during the post-nesting migration of olive ridley turtles from French Guiana. *Endangered Species Research 26* (3): 221–234.

Plotkin, P. and A. F. Amos. (1998). *Entanglement and Ingestion of Marine Turtles Stranded Along the South Texas Coast* [Type]. Presented at the Eighth Annual Workshop on Sea Turtle Conservation and Biology. Fort Fisher, NC.

Poloczanska, E. S., M. T. Burrows, C. J. Brown, J. G. Molinos, B. S. Halpern, O. Hoegh-Guldberg, C. V. Kappel, P. J. Moore, A. J. Richardson, D. S. Schoeman, and W. J. Sydeman. (2016). Responses of marine organisms to climate change across oceans. *Frontiers in Marine Science 3* (62): 1–21. DOI:10.3389/fmars.2016.00062

Polovina, J., I. Uchida, G. Balazs, E. A. Howell, D. Parker, and P. Dutton. (2006). The Kuroshio Extension Bifurcation Region: A pelagic hotspot for juvenile loggerhead sea turtles. *Deep-Sea Research II 53* 326–339. DOI:10.1016/j.dsr2.2006.01.006

Polovina, J. J., G. H. Balazs, E. A. Howell, D. M. Parker, M. P. Seki, and P. H. Dutton. (2004). Forage and migration habitat of loggerhead (*Caretta caretta*) and olive ridley (*Lepidochelys olivacea*) sea turtles in the central North Pacific Ocean. *Fisheries Oceanography 13* (1): 36–51.

Polovina, J. J., E. Howell, D. M. Parker, and G. H. Balazs. (2002). Dive-depth distribution of loggerhead (*Caretta caretta*) and olive ridley (*Lepidochelys olivacea*) sea turtles in the central North Pacific: Might deep longline sets catch fewer turtles? *Fishery Bulletin 101* (1): 189–193.

Polovina, J. J., P. Kleiber, and D. R. Kobayashi. (1999). Application of TOPEX-POSEIDON satellite altimetry to simulate transport dynamics of larvae of spiny lobster, *Panulirus marginatus*, in the Northwestern Hawaiian Islands, 1993–1996. *Fishery Bulletin 97* 132–143.

Polovina, J. J., D. R. Kobayashi, D. M. Parker, M. P. Seki, and G. H. Balazs. (2000). Turtles on the edge: Movement of loggerhead turtles (*Caretta caretta*) along oceanic fronts, spanning longline fishing grounds in the central North Pacific, 1997–1998. *Fisheries Oceanography 9* (1): 71–82.

Pondella, D. J., J. T. Claisse, and J. Williams. (2015). *Characterization of Kelp Forest and Shallow Rock Ecosystems at San Clemente Island, California, 2012-2013* (Kelp Forest Ecosystems at San Clemente Island). Los Angeles, CA: Vantuna Research Group.

Ponganis, P. (2015). *Diving Physiology of Marine Mammals and Seabirds*. Cambridge, United Kingdom: Cambridge University Press.

Poole, A. F., R. O. Bierregaard, and M. S. Martell. (2002). *Osprey (Pandiol haliaetus)*. *The Birds of North America Online.* Retrieved 3, 2011, from http://bna.birds.cornell.edu/bna/species/563.

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. and M. C. Hastings. (2009). The effects of anthropogenic sources of sound on fishes. *Journal of Fish Biology 75* (3): 455–489. DOI:10.1111/j.1095-8649.2009.02319

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.

Porter, J. W., P. Dustan, W. C. Jaap, K. L. Patterson, V. Kosmynin, O. W. Meier, M. E. Patterson, and M. Parsons. (2001). Patterns of spread of coral disease in the Florida Keys. *Hydrobiologia 460* 1–24. DOI:doi: 10.1023/A:1013177617800

Powell, M. L., S. I. Kavanaugh, and S. A. Sower. (2005). Current knowledge of hagfish reproduction: Implications for fisheries management. *Integrative & Comparative Biology 45* 158–165.

Pratt, H., P. Bruner, and D. Berrett. (1987). *The Birds of Hawaii and the Tropical Pacific*. Princeton, NJ: Princeton University Press.

Prescott, R. (1982). Harbor seals: Mysterious lords of the winter beach. *Cape Cod Life 3* (4): 24–29.

Preskitt, L. (2001). *Halophila hawaiiana*. *Invasive Marine Algae of Hawai'i.* Retrieved from http://www.hawaii.edu/reefalgae/invasive_algae/seagrasses/halophia_hawaiiana.htm.

Preskitt, L. (2002a). *Edible Limu: Gifts from the Sea*. Retrieved from http://www.hawaii.edu/reefalgae/publications/ediblelimu/index.htm.

Preskitt, L. (2002b). *Halophila decipiens*. *Invasive Marine Algae of Hawai'i.* Retrieved from http://www.hawaii.edu/reefalgae/invasive_algae/seagrasses/halophila_decipiens.htm.

Preskitt, L. (2010). *Invasive Marine Algae of Hawaii*. Retrieved from http://www.hawaii.edu/reefalgae/invasive_algae/index.htm.

Price, M. and C. Antaky. (2020). HAWAIIAN BAND-RUMPED STORM PETREL PROJECT.

Price, S. A., S. T. Friedman, and P. C. Wainwright. (2015). How predation shaped fish: The impact of fin spines on body form evolution across teleosts. *Proceedings of the Royal Society B: Biological Sciences 282* (1819). DOI:10.1098/rspb.2015.1428

Putman, N. F. and K. L. Mansfield. (2015). Direct Evidence of Swimming Demonstrates Active Dispersal in the Sea Turtle "Lost Years". *Current Biology 25* (9): 1221–1227.

Putnam, H. M. and R. D. Gates. (2015). Preconditioning in the reef-building coral *Pocillopora damicornis* and the potential for trans-generational acclimatization in coral larvae under future climate change conditions. *The Journal of Experimental Biology 218* 2365–2372. DOI:10.1242/jeb.12301

Pyle, P., D. J. Long, J. Schonewald, R. E. Jones, and J. Roletto. (2001). Historical and recent colonization of the South Farallon Islands, California, by northern fur seals (*Callorhinus ursinus*). *Marine Mammal Science 17* (2): 397–402.

Pyle, R. L., R. K. Kosaki, H. T. Pinheiro, L. A. Rocha, R. K. Whitton, and J. M. Copus. (2019). Fishes: Biodiversity. In Bernhard M. Riegl & Richard E. Dodge (Eds.), *Mesophotic Coral Ecosystems* (pp. 749–777). Cham, Switzerland: Springer Nature.

Pyper, B., T. Garrison, S. Cramer, P. L. Brandes, D. P. Jacobson, and M. A. Banks. (2013). *Absolute abundance estimates of juvenile spring-run and winterrun Chinook salmon at Chipps Island*. Newport, OR: U.S. Fish and Wildlife Service.

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

Quinn, T. P. and K. W. Myers. (2005). Anadromy and the marine migrations of Pacific salmon and trout: Rounsefell revisited. *Reviews in Fish Biology and Fisheries 14* 421–442.

Rabalais, N. N., R. E. Turner, and W. J. Wiseman, Jr. (2002). Gulf of Mexico hypoxia, a.k.a. "The Dead Zone" *Annual Review of Ecology and Systematics 33* 235–263.

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.

Radford, C. A., J. A. Stanley, C. T. Tindle, J. C. Montgomery, and A. G. Jeffs. (2010). Localised coastal habitats have distinct underwater sound signatures. *Marine Ecology Progress Series 401* 21–29. DOI:10.3354/meps08451

Radjasa, O. K., Y. M. Vaske, G. Navarro, H. C. Vervoort, K. Tenney, R. G. Linington, and P. Crews. (2011). Highlights of marine invertebrate-derived biosynthetic products: Their biomedical potential and possible production by microbial associants. *Bioorganic and Medicinal Chemistry 19* (22): 6658 – 6674. DOI:10.1016/j.bmc.2011.07.017

Ragazzola, F., L. C. Foster, C. J. Jones, T. B. Scott, J. Fietzke, M. R. Kilburn, and D. N. Schmidt. (2016). Impact of high CO_2 on the geochemistry of the coralline algae *Lithothamnion glaciale*. *Scientific Reports 6* (20572): 1–9. DOI:10.1038/srep20572

Ragen, T. J., G. A. Antonelis, and M. Kiyota. (1995). Early migration of northern fur seal pups from St. Paul Island, Alaska. *Journal of Mammalogy 76* (4): 1137–1148.

Raimondi, P. (2016). *Assessment of abalone populations at Vandenberg Air Force Base with an emphasis on the endangered species Haliotis cracherodii – the black abalone and Haliotis kamtschatkana - the pinto abalone* (Project Number: XUMU448514 (Task 1 & 2)). Santa Cruz, CA: University of California Santa Cruz, Department of Ecology and Evolutionary Biology.

Raimondi, P., D. Orr, N. Fletcher, M. Redfield, D. Lohse, M. Miner, M. George, R. Gaddam, J. Engle, S. Lee, and S. Whitaker. (2012). *Estimation of population size of the black abalone, Haliotis* *cracherodii, on San Clemente Island*. Santa Cruz, CA: Department of Ecology and Evolutionary Biology; Long Marine Laboratory.

Raine, A. F., M. Boone, M. McKOWN, and N. Holmes. (2017). The breeding phenology and distribution of the Band-rumped Storm-petrel Oceanodroma castro on Kaua'i and Lehua Islet, Hawaiian Islands. *Marine Ornithology 45* 73-82.

Ralph, C. J. and S. L. Miller. (1995). Offshore population estimates of Marbled Murrelets in California. In C. J. Ralph, G. L. Hunt, Jr., M. G. Raphael, & J. F. Piatt (Eds.), *Ecology and Conservation of the Marbled Murrelet. USDA Forest Service General Technical Report PSW-152* (pp. 353–360). Albany, CA: Pacific Southwest Research Station.

Ralph, P. J. and F. T. Short. (2002). Impact of the wasting disease pathogen, *Labyrinthula zosterae*, on the photobiology of eelgrass *Zostera marina*. *Marine Ecology Progress Series 226* 265–271.

Ramos, J., L. Monteiro, E. Sola, and Z. Moniz. (1997). Characteristics and Competition for Nest Cavities in Burrowing Procellariiformes. *The Condor 99* 634–641.

Ramp, C., J. Delarue, P. J. Palsboll, R. Sears, and P. S. Hammond. (2015). Adapting to a warmer ocean—Seasonal shift of baleen whale movements over three decades. *PLoS ONE 10* (3): e0121374. DOI:10.1371/journal.pone.0121374

Rankin, S. and J. Barlow. (2005). Source of the North Pacific “boing” sound attributed to minke whales. *The Journal of Acoustical Society of America 118* (5): 3346. DOI:10.1121/1.2046747

Rankin, S. and J. Barlow. (2007). Sounds recorded in the presence of Blainville's beaked whales, *Mesoplodon densirostris*, near Hawaii. *The Journal of Acoustical Society of America 122* (1): 42–45. DOI:10.1121/1.2743159

Rankin, S., T. F. Norris, M. A. Smultea, C. Oedekoven, A. M. Zoidis, E. Silva, and J. Rivers. (2007). A visual sighting and acoustic detections of minke whales, *Balaenoptera acutorostrata* (Cetacea: Balaenopteridae), in nearshore Hawaiian waters. *Pacific Science 61* 395–398.

Raum-Suryan, K. L., M. J. Rehberg, G. W. Pendleton, K. W. Pitcher, and T. S. Gelatt. (2004). Development of Dispersal, Movement Patterns, and Haul-Out Use by Pup and Juvenile Steller Sea Lions (*Eumetopias jubatus*) in Alaska. *Marine Mammal Science 20* (4): 823–850.

Raven, J., K. Caldeira, H. Elderfield, O. Hoegh-Guldberg, P. Liss, U. Riebesell, J. Sheperd, C. Turley, A. Watson, R. Heap, R. Banes, and R. Quinn. (2005). *Ocean acidification due to increasing atmospheric carbon dioxide*. London, United Kingdom: The Royal Society.

Ream, R. R., J. T. Sterling, and T. R. Loughlin. (2005). Oceanographic features related to northern fur seal migratory movements. *Deep-Sea Research II 52* 823–843.

Redfern, J. V., M. F. McKenna, T. J. Moore, J. Calambokidis, M. L. Deangelis, E. A. Becker, J. Barlow, K. A. Forney, P. C. Fiedler, and S. J. Chivers. (2013). Assessing the risk of ships striking large whales in marine spatial planning. *Conservation Biology 27* (2): 292–302. DOI:10.1111/cobi.12029

Reed, D., S. Schroeter, and M. Page. (2010). *Annual Report of the Status of Condition C: Kelp Reef Mitigation* (San Onofre Nuclear Generating Station Mitigation Program). Submitted to the California Coastal Commission.

Rees, D. R., D. V. Jones, and B. A. Bartlett. (2016). *Haul-Out Counts and Photo-Identification of Pinnipeds in Chesapeake Bay, Virginia: 2015/16 Annual Progress Report. Final Report.* Norfolk, VA: Naval Facilities Engineering Command Atlantic.

Reeves, R. R., S. Leatherwood, and R. W. Baird. (2009). Evidence of a possible decline since 1989 in false killer whales (*Pseudorca crassidens*) around the main Hawaiian islands. *Pacific Science 63* (2): 253–261.

Reeves, R. R., B. D. Smith, E. A. Crespo, and G. Notarbartolo di Sciara. (2003). *Dolphins, Whales and Porpoises: 2002–2010 Conservation Action Plan for the World's Cetaceans*. Gland, Switzerland and Cambridge, UK: IUCN.

Reeves, R. R., B. S. Stewart, P. J. Clapham, and J. A. Powell. (2002). *National Audubon Society Guide to Marine Mammals of the World*. New York, NY: Alfred A. Knopf.

Reeves, R. R., B. S. Stewart, and S. Leatherwood. (1992). *The Sierra Club Handbook of Seals and Sirenians*. San Francisco, CA: Sierra Club Books.

Reilly, S. B. (1990). Seasonal changes in distribution and habitat differences among dolphins in the eastern tropical Pacific. *Marine Ecology Progress Series 66* 1–11.

Renaud, M. L., J. A. Carpenter, J. A. Williams, and A. M. Landry, Jr. (1996). Kemp's ridley sea turtle (*Lepidochelys kempii*) tracked by satellite telemetry from Louisiana to nesting beach at Rancho Nuevo, Tamaulipas, Mexico. *Chelonian Conservation and Biology 2* (1): 108–109.

Reshetiloff, K. (2004). *Chesapeake Bay: Introduction to an Ecosystem*. Washington, DC: Environmental Protection Agency.

Restore the Gulf. (2010). *America's Gulf Coast: A Long Term Recovery Plan After the Deepwater Horizon Oil Spill*. Washington, DC: U.S. Environmental Protection Agency.

Rex, M. A. and R. J. Etter. (1998). Bathymetric patterns of body size: Implications for deep-sea biodiversity. *Deep-Sea Research Part II-Topical Studies in Oceanography 45* 103–127.

Reyes, J. C., J. G. Mead, and K. Van Waerebeek. (1991). A new species of beaked whale, *Mesoplodon peruvianus* sp. n. (Cetacea: Ziphiidae), from Peru. *Marine Mammal Science 7* (1): 1–24.

Reynolds, J. D., N. K. Dulvy, N. B. Goodwin, and J. A. Hutchings. (2005). Biology of extinction risk in marine fishes. *Proceedings of the Royal Society B: Biological Sciences 272* (1579): 2337–2344. DOI:10.1098/rspb.2005.3281

Reynolds, M. and G. Ritchotte. (1997). Evidence of Newell's Shearwater Breeding in Puna District, Hawaii. *Journal of Field Ornithology 68* (1): 26–32.

Ribic, C. A., S. B. Sheavly, D. J. Rugg, and E. S. Erdmann. (2012). Trends in marine debris along the U.S. Pacific Coast and Hawaii 1998–2007. *Marine Pollution Bulletin 64* 994–1004.

Rice, D. W. (1989). Sperm whale, *Physeter macrocephalus* Linnaeus, 1758. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals, Volume 4: River dolphins and the larger toothed whales* (Vol. 4, pp. 177–234). San Diego, CA: Academic Press.

Rice, D. W. (1998). *Marine Mammals of the World: Systematics and Distribution* (Society for Marine Mammalogy Special Publication). Lawrence, KS: Society for Marine Mammalogy.

Rice, D. W., A. A. Wolman, and H. W. Braham. (1984). The Gray Whale, *Eschrichtius robustus*. *Marine Fisheries Review 46* (4): 7–14.

Rice, M. R. and G. H. Balazs. (2008). Diving behavior of the Hawaiian green turtle (*Chelonia mydas*) during oceanic migrations. *Journal of Experimental Marine Biology and Ecology 356* (1–2): 121–127. DOI:10.1016/j.jembe.2007.12.010

Richardson, J. I. and P. McGillivary. (1991). Post-hatchling loggerhead turtles eat insects in *Sargassum* community. *Marine Turtle Newsletter 55* 2–5.

Richie, M. W., R. K. Uyeyama, J. M. Aschettino, and M. A. Fagan. (2016). *Marine Species Surveys of Pearl Harbor, Nov 2013–Nov 2015, and Historical Occurrence of Marine Species in Pearl Harbor* (Submitted to: NAVFAC Hawaii for Commander, Navy Installation Command, Joint Base Pearl Harbor Hickam). Pearl Harbor, HI: NAVFAC Pacific.

Richie, M. W., R. K. Uyeyama, and J. Fujimoto. (2012). *Final Report Ka'ula Island ship-based marine mammal survey, July 6, 2012 Hawaii Range Complex Field Report*. Pearl Harbor, HI.

Rickel, S. and A. Genin. (2005). Twilight transitions in coral reef fish: The input of light-induced changes in foraging behaviour. *Animal Behaviour 70* (1): 133–144. DOI:10.1016/j.anbehav.2004.10.014

Rijke, A. M. (1970). Wettability and phylogenetic development of feather structure in water birds. *The Journal of Experimental Biology 52* (2): 469–479.

Ritchie, E. I. (2016, April 18, 2016). False killer whales spotted in feeding frenzy off Dana Point Coast–rare for this area.

Roberson, D. (2000, 26 May 2000). *California Short-Tailed Albatrosses. A Summary at the Turn of the 21st Century*. Retrieved from http://www.montereybay.com/creagrus/CA_STAL.html.

Roberts, L., S. Cheesman, M. Elliott, and T. Breithaupt. (2016). Sensitivity of *Pagurus bernhardus* (L.) to substrate-borne vibration and anthropogenic noise. *Journal of Experimental Marine Biology and Ecology 474* 185–194.

Roberts, S. and M. Hirshfield. (2003). Deep sea corals: Out of sight, but no longer out of mind. *Frontiers in Ecology & the Environment 2* (3): 123–130.

Robinson, N. J., S. E. Valentine, P. Santidrián Tomillo, V. S. Saba, J. R. Spotila, and F. V. Paladino. (2013). Multidecadal trends in the nesting phenology of Pacific and Atlantic leatherback turtles are associated with population demography. *Endangered Species Research 24* 197–206.

Robinson, P. W., D. P. Costa, D. E. Crocker, J. P. Gallo-Reynoso, C. D. Champagne, M. A. Fowler, C. Goetsch, K. T. Goetz, J. L. Hassrick, L. A. Huckstadt, C. E. Kuhn, J. L. Maresh, S. M. Maxwell, B. I. McDonald, S. H. Peterson, S. E. Simmons, N. M. Teutschel, S. Villegas-Amtmann, and K. Yoda. (2012). Foraging behavior and success of a mesopelagic predator in the northeast Pacific Ocean: Insights from a data-rich species, the northern elephant seal. *PLoS ONE 7* (5): e36728. DOI:10.1371/journal.pone.0036728

Robinson, S., M. Barbieri, and T. Johanos. (2022). The Hawaiian Monk Seal: Ethology Applied to Endangered Species Conservation and Recovery. In Daniel P. Costa & E. A. McHuron (Eds.), *Ethology and Behavioral Ecology of Phocids* (pp. 599-635). Cham, Switzerland: Springer Nature.

Robson, B. W., M. E. Goebel, J. D. Baker, R. R. Ream, T. R. Loughlin, R. C. Francis, G. A. Antonelis, and D. P. Costa. (2004). Separation of foraging habitat among breeding sites of a colonial marine predator, the northern fur seal (*Callorhinus ursinus*). *Canadian Journal of Zoology 82* (1): 20–29. DOI:10.1139/z03-208

Rocha, R. C., P. J. Clapham, Jr., and Y. V. Ivashchenko. (2014). Emptying the Oceans: A Summary of Industrial Whaling Catches in the 20th Century. *Marine Fisheries Review 76* (4): 37–48.

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.

Rockwood, R. C., J. D. Adams, S. Hastings, J. Morten, and J. Jahncke. (2021). Modeling whale deaths from vessel strikes to reduce the risk of fatality to endangered whales. *Frontiers in Marine Science 8*. DOI:10.3389/fmars.2021.649890

Rockwood, R. C., M. L. Elliott, B. Saenz, N. Nur, and J. Jahncke. (2020). Modeling predator and prey hotspots: Management implications of baleen whale co-occurrence with krill in Central California. *PLoS ONE 15* (7). DOI:10.1371/journal.pone.0235603

Roden, S. E., K. R. Stewart, M. C. James, K. L. Dodge, F. Dell'Amico, and P. H. Dutton. (2017). Genetic fingerprinting reveals natal origins of male leatherback turtles encountered in the Atlantic Ocean and Mediterranean Sea. *Marine Biology 164* (9): 181.

Rodgers, K. S., P. L. Jokiel, E. K. Brown, S. Hau, and R. Sparks. (2015). Over a Decade of Change in Spatial and Temporal Dynamics of Hawaiian Coral Reef Communities. *Pacific Science 69* (1): 1–13. DOI:10.2984/69.1.1

Rodríguez, A., N. Holmes, P. Ryan, K. Wilson, L. Faulquier, Y. Murillo, A. Raine, J. Penniman, V. Neves, B. Rodríguez, J. Negro, A. Chiaradia, P. Dann, T. Anderson, B. Metzger, M. Shirai, L. Deppe, J. Wheeler, P. Hodum, C. Gouveia, V. Carmo, G. Carreira, L. Delgado-Alburqueque, C. Guerra-Correa, F. Couzi, M. Travers, and M. Le Corre. (2017). A global review of seabird mortality caused by land-based artificial lights. *Conservation Biology* 1–40. DOI:10.1111/cobi.12900

Rodriguez, S., A. R. T. Santiago, and G. Shenker. (2001). *A Public-Access GIS-Based Model of Potential Species Habitat Distribution for the Santa Barbara Channel and the Channel Islands National Marine Sanctuary.* (Unpublished master's group project). University of California, Santa Barbara, Santa Barbara, CA.

Roe, P. and J. L. Norenburg. (1999). Observations on depth distribution, diversity and abundance of pelagic nemerteans from the Pacific Ocean off California and Hawaii. *Deep-Sea Research 46* (1): 1201–1220.

Roessig, J. M., C. M. Woodley, J. J. Cech, Jr., and L. J. Hansen. (2004). Effects of global climate change on marine and estuarine fishes and fisheries. *Reviews in Fish Biology and Fisheries 14* (2): 251–275.

Rogers-Bennett, L., P. L. Haaker, T. O. Huff, and P. K. Dayton. (2002). *Estimating Baseline Abundances of Abalone in California for Restoration* (California Cooperative Oceanic Fisheries Investigations [CalCOFI] Reports).

Rogers, K. S. (2016, 10-12-2016). Feral Cats are a Serious Threat to Hawaii's Endangered Birds, Monk Seals and Dolphins. *Honolulu Magazine*.

Roleda, M. Y., C. E. Cornwall, Y. Feng, C. M. McGraw, A. M. Smith, and C. L. Hurd. (2015). Effect of Ocean Acidification and pH Fluctuations on the Growth and Development of Coralline Algal Recruits, and an Associated Benthic Algal Assemblage. *PLoS ONE 10* (10): e0140394. DOI:10.1371/journal.pone.0140394

Rolland, R. M., S. E. Parks, K. E. Hunt, M. Castellote, P. J. Corkeron, D. P. Nowacek, S. K. Wasser, and S. D. Kraus. (2012). Evidence that ship noise increases stress in right whales. *Proceedings of the Royal Society B: Biological Sciences 279* (1737): 2363–2368. DOI:10.1098/rspb.2011.2429

Rooper, C. N., P. J. Etnoyer, K. L. Stierhoff, and J. V. Olson. (2016). Effects of Fishing Gear on Deep-Sea Corals and Sponges in U.S. Waters. In T. F. Hourigan, P. J. Etnoyer, & S. D. Cairns (Eds.), *The State of Deep-Sea Coral and Sponge Ecosystems in the United States* (Vol. NOAA Technical Memorandum NMFS-OHC-3, pp. 4–1–4–19). Silver Spring, MD: National Oceanic and Atmospheric Administration.

Rosel, P. E. and H. Watts. (2008). Hurricane impacts on bottlenose dolphins in the northern Gulf of Mexico. *Gulf of Mexico Science 25* (1): 88–94.

Roskov, Y., L. Abucay, T. Orrell, D. Nicolson, T. Kunze, A. Culham, N. Bailly, P. Kirk, T. Bourgoin, R. E. DeWalt, W. Decock, and A. De Weaver. (2015, July 6, 2015). *Species 2000 & ITIS Catalogue of Life, 2015 Annual Checklist*. Retrieved from http://www.catalogueoflife.org/annual-checklist/2015/.

Ross, S. W., A. W. J. Demopoulos, C. A. Kellogg, C. L. Morrison, M. S. Nizinski, C. L. Ames, T. L. Casazza, D. Gualtieri, K. Kovacs, J. P. McClain, A. M. Quattrini, A. Y. Roa-Varón, and A. D. Thaler. (2012). *Deepwater Program: Studies of Gulf of Mexico Lower Continental Slope Communities Related to Chemosynthetic and Hard Substrate Habitats*. Reston, VA: U.S. Geological Survey.

Rossi, T., S. D. Connell, and I. Nagelkerken. (2016). Silent oceans: Ocean acidification impoverishes natural soundscapes by altering sound production of the world's noisiest marine invertebrate. *Proceedings of the Royal Society B: Biological Sciences 283* (1826): 20153046. DOI:10.1098/rspb.2015.3046

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

Rubega, M. A., D. Schamel, D. M. Tracy, A. Poole, and F. Gill. (2000). Red-necked Phalarope (*Phalaropus lobatus*). *The Birds of North America Online* (538): 5. DOI:10.2173/bna.538

Ruggerone, G. T., T. P. Quinn, I. A. McGregor, and T. D. Wilkinson. (1990). Horizontal and vertical movements of adult steelhead trout, *Oncorhynchus mykiss*, in the Dean and Fisher channels, British Columbia. *Canadian Journal of Fisheries and Aquatic Science 47* 1963–1969.

Rugh, D., J. Breiwick, M. Muto, R. Hobbs, K. Shelden, C. D'Vincent, I. M. Laursen, S. Reif, S. Maher, and S. Nilson. (2008). *Report of the 2006–2007 Census of the Eastern North Pacific Stock of Gray Whales*. Seattle, WA: NOAA, NMFS, Alaska Fisheries Science Center.

Rugh, D. J., R. C. Hobbs, J. A. Lerczak, and J. M. Breiwick. (2005). Estimates of abundance of the eastern North Pacific stock of gray whales (*Eschrichtius robustus*) 1997–2002. *Journal of Cetacean Research and Management 7* (1): 1–12.

Russell, D. J. and G. H. Balazs. (2015). Increased use of non-native algae species in the diet of the green turtle (*Chelonia mydas*) in a primary pasture ecosystem in Hawaii. *Aquatic Ecosystem Health & Management 18* (3): 342–346.

Ruwa, R. K. (1996). Intertidal wetlands. In T. R. McClanahan & T. P. Young (Eds.), *East African Ecosystems and Their Conservation* (pp. 101–130). New York, NY: Oxford University Press.

Ruz-Moreno, P. M., P. Hidalgo, M. Cornejo-D'Ottone, P. Fierro-González, J. Babbonney, and K. W. Tang. (2023). Potential effects of shoaling of oxygen minimum zone on the population of *Acartia* (*Acanthacartia*) *tonsa* in the Humboldt Current Upwelling System (23 S). *Journal of Sea Research 193*. DOI:10.1016/j.seares.2023.102372

Ryland, J. S. and P. J. Hayward. (1991). *Marine Flora and Fauna of the Northeastern United States. Erect Bryozoa*. Washington, DC: National Oceanic and Atmospheric Administration.

Sade, J., Y. Handrich, J. Bernheim, and D. Cohen. (2008). Pressure equilibration in the penguin middle ear. *Acta Oto-Laryngologica 128* (1): 18–21. DOI:10.1080/00016480701299667

Sakamoto, W., K. Sato, H. Tanaka, and Y. Naito. (1993). Diving patterns and swimming environment of two loggerhead turtles during internesting. *Nippon Suisan Gakkaishi 59* (7): 1129–1137.

Sakashita, M. (2021). *Petition to List Sunflower Sea Star as Threatened or Endangered Under the U.S. Endangered Species Act*. Oakland, CA: Center for Biological Diversity.

Salmon, M., T. T. Jones, and K. W. Horch. (2004). Ontogeny of diving and feeding behavior in juvenile seaturtles: Leatherback seaturtles (*Dermochelys coriacea* L) and green seaturtles (*Chelonia mydas* L) in the Florida current. *Journal of Herpetology 38* (1): 36–43.

Salvadeo, C. J., A. Gomez-Gallardo U., M. Najera-Caballero, J. Urban-Ramirez, and D. Lluch-Belda. (2015). The effect of climate variability on gray whales (*Eschrichtius robustus*) within their wintering areas. *PLoS ONE 10* (8): 1–17. DOI:10.1371/journal.pone.0134655.g001

Salvadeo, C. J., D. Lluch-Belda, A. Gómez-Gallardo, J. Urbán-Ramírez, and C. D. MacLeod. (2010). Climate change and a poleward shift in the distribution of the Pacific white-sided dolphin in the northeastern Pacific. *Endangered Species Research 11* 13–19. DOI:10.3354/esr00252

Samuel, M., B. Woodworth, C. Atkinson, P. Hart, and D. LaPointe. (2015). Avian malaria in Hawaiian forest birds: Infection and population impacts across species and elevations. *Ecosphere 6* (6): 1–21.

Sanderson, C. E. and K. A. Alexander. (2020). Unchartered waters: Climate change likely to intensify infectious disease outbreaks causing mass mortality events in marine mammals. *Global Change Biology 26* (8): 4284–4301. DOI:10.1111/gcb.15163

Sanford, E., J. L. Sones, M. García-Reyes, J. H. R. Goddard, and J. L. Largier. (2019). Widespread shifts in the coastal biota of northern California during the 2014–2016 marine heatwaves. *Scientific Reports 9* (1): 1–14. DOI:10.1038/s41598-019-40784-3

Sanino, G. P., J. L. Yanez, and K. Van Waerebeek. (2007). A first confirmed specimen record in Chile, and sightings attributed to the lesser beaked whale, *Mesoplodon peruvianus* Reyes, Mead and Van Waerebeek, 1991. *Boletin del Museo Nacional de Historia Natural, Chile 56* 89–96.

Santora, J. A., N. J. Mantua, I. D. Schroeder, J. C. Field, E. L. Hazen, S. J. Bograd, W. J. Sydeman, B. K. Wells, J. Calambokidis, L. Saez, D. Lawson, and K. A. Forney. (2020). Habitat compression and ecosystem shifts as potential links between marine heatwave and record whale entanglements. *Nature Communications 11* (1): 536. DOI:10.1038/s41467-019-14215-w

São Miguel, R. A., R. Anastácio, and M. J. Pereira. (2022). Sea Turtle Nesting: What Is Known and What Are the Challenges under a Changing Climate Scenario. *Open Journal of Ecology 12* (1): 1-35.

Sarti-Martinez, L., S. A. Eckert, N. Garcia T., and A. R. Barragan. (1996). Decline of the world's largest nesting assemblage of leatherback turtles. *Marine Turtle Newsletter 74* 2–5.

Satterthwaite, W. H., S. M. Carlson, and A. Criss. (2017). Ocean Size and Corresponding Life History Diversity among the Four Run Timings of California Central Valley Chinook Salmon. *Transactions of the American Fisheries Society 146* (5): 594-610. DOI:10.1080/00028487.2017.1293562

Satterthwaite, W. H., J. Ciancio, E. D. Crandall, M. L. Palmer-Zwahlen, A. M. Grover, M. R. O'Farrell, E. C. Anderson, M. S. Mohr, and C. Garza. (2015). Stock composition and ocean spatial distribution inference from California recreational Chinook salmon fisheries using genetic stock identification. *Fisheries Research 170* 166-178. DOI:doi.org/10.1016/j.fishres.2015.06.001

Satterthwaite, W. H., M. S. Mohr, E. C. Anderson, M. A. Banks, S. J. Bates, M. R. Bellinger, L. A. Borgerson, E. D. Crandall, J. C. Garza, and B. J. Kormos. (2014). *Use of Genetic Stock Identification Data for Comparison of the Ocean Spatial Distribution, Size at Age, and Fishery Exposure of an Untagged Stock and Its Indicator: California Coastal versus Klamath River Chinook Salmon* (Transactions of the American Fisheries Society).

Satterthwaite, W. H., M. S. Mohr, M. R. O'Farrell, and B. K. Wells. (2013). A comparison of temporal patterns in the ocean spatial distribution of California's Central Valley Chinook salmon runs. *Canadian Journal of Fisheries and Aquatic Sciences 70* (4): 574–584.

Schiel, D. R., J. R. Steinbeck, and M. S. Foster. (2004). Ten years of induced ocean warming causes comprehensive changes in marine benthic communities. *Ecology 85* (7): 1833–1839.

Schlining, K., S. von Thun, L. Kuhnz, B. Sclining, L. Lundsten, N. J. Stout, L. Chaney, and J. Connor. (2013). Debris in the deep: Using a 22-year video annotation database to survey marine litter in Monterey Canyon, central California, USA. *Deep-Sea Research I 1* (79): 96–105. DOI:http://dx.doi.org/10.1016/j.dsr.2013.05.006

Schneider, D. C. and P. M. Payne. (1983). Factors affecting haul-out of harbor seals at a site in southeastern Massachusetts. *Journal of Mammalogy 64* (3): 518–520.

Schnetzer, A., P. E. Miller, R. A. Schaffner, B. A. Stauffer, B. H. Jones, S. B. Weisberg, P. M. DiGiacomo, W. M. Berelson, and D. A. Caron. (2007). Blooms of *Pseudo-nitzschia* and domoic acid in the San Pedro Channel and Los Angeles harbor areas of the Southern California Bight, 2003–2004. *Harmful Algae 6* 372–387. DOI:doi:10.1016/j.hal.2006.11.004

Schorr, G. S., E. A. Falcone, D. J. Moretti, and R. D. Andrews. (2014). First long-term behavioral records from Cuvier's beaked whales (*Ziphius cavirostris*) reveal record-breaking dives. *PLoS ONE 9* (3): e92633. DOI:10.1371/journal.pone.0092633

Schorr, G. S., E. A. Falcone, and B. K. Rone. (2017). *Distribution and demographics of Cuvier's beaked whales and fin whales in the Southern California Bight* (Annual report for on-water surveys conducted in conjunction with Marine Mammal Monitoring on Navy Ranges (M3R)). Seabeck, WA: Marine Ecology and Telemetry Research.

Schreiber, R. and J. Chovan. (1986). Roosting by pelagic seabirds: Energetic, populational, and social considerations. *The Condor 88* 487–492.

Schroeder, B. A., A. M. Foley, and D. A. Bagley. (2003). Nesting patterns, reproductive migrations, and adult foraging areas of loggerhead turtles. In A. B. Bolten & B. E. Witherington (Eds.), *Loggerhead Sea Turtles* (pp. 114–124). Washington, DC: Smithsonian Institution Press.

Schultz, J. K., J. D. Baker, R. J. Toonen, A. L. Harting, and B. W. Bowen. (2011). Range-wide genetic connectivity of the Hawaiian monk seal and implications for translocation. *Conserv Biol 25* (1): 124–132. DOI:10.1111/j.1523-1739.2010.01615.x

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.

Schwartz, F. J. (1989). *Zoogeography and Ecology of Fishes Inhabiting North Carolina's Marine Waters to Depths of 600 Meters*. Silver Spring, MD: National Oceanic and Atmospheric Administration.

Seagars, D. J. (1984). *The Guadalupe Fur Seal: A Status Review*. Terminal Island, CA: National Marine Fisheries Service, Southwest Region.

Seigel, J. A. (1985). The Scalloped Hammerhead, *Sphyrna lewini*, In Coastal Southern California Waters: Three Records Including the First Reported Juvenile. *California Fish and Game 71* (3).

Seitz, A. C. and M. B. Courtney. (2021). *Ocean Migration and Behavior of Steelhead Kelts in Alaskan OCS Oil and Gas Lease Areas, Examined with Satellite Telemetry*. Fairbanks, AK: Bureau of Ocean Energy and University of Alaska Fairbanks.

Seitz, A. C. and M. B. Courtney. (2022). *Telemetry and Genetic Identity of Chinook Salmon in Alaska: Preliminary Report of Satellite Tags Deployed in 2020-2021*. Fairbanks, AK: University of Alaska Fairbanks, College of Fisheries and Ocean Sciences.

Seitz, A. C. and M. B. Courtney. (2023). *Telemetry and Genetic Identity of Chinook Salmon in Alaska: Preliminary Report of Satellite Tags Deployed in 2020-2022*. Fairbanks, AK: University of Alaska Fairbanks, College of Fisheries and Ocean Sciences.

Seitz, A. C. and M. B. Courtney. (2024). *Telemetry and Genetic Identity of Chinook Salmon in Alaska: Final Report*. Fairbanks, AK: University of Alaska Fairbanks, College of Fisheries and Ocean Sciences.

Seminoff, J. A., C. D. Allen, G. H. Balazs, P. H. Dutton, T. Eguchi, H. L. Haas, S. A. Hargrove, M. P. Jensen, D. L. Klemm, A. M. Lauritsen, S. L. MacPherson, P. Opay, E. E. Possardt, S. L. Pultz, E. E. Seney, K. S. Van Houtan, and R. S. Waples. (2015). *Status Review of the Green Turtle (Chelonia mydas) Under the U.S. Endangered Species Act*. La Jolla, CA: Southwest Fisheries Science Center.

Seminoff, J. A., T. Eguchi, J. Carretta, C. D. Allen, D. Prosperi, R. Rangel, J. W. Gilpatrick, K. Forney, and S. H. Peckham. (2014). Loggerhead sea turtle abundance at a foraging hotspot in the eastern Pacific Ocean: Implications for at-sea conservation. *Endangered Species Research 24* (3): 207–220.

Seminoff, J. A., T. T. Jones, A. Resendiz, W. J. Nichols, and M. Y. Chaloupka. (2003a). Monitoring green turtles (*Chelonia mydas*) at a coastal foraging area in Baja California, Mexico: Multiple indices describe population status. *Journal of the Marine Biological Association of the United Kingdom 83* 1355–1362.

Seminoff, J. A., W. J. Nichols, A. Resendiz, and L. Brooks. (2003b). Occurrence of hawksbill turtles, *Eretmochelys imbricata* (Reptilia: Cheloniidae), near the Baja California peninsula, Mexico. *Pacific Science 57* (1): 9–16.

SEPM Strata. (2018). *Tide Dominated Deltas*. Retrieved from http://www.sepmstrata.org/page.aspx?pageid=313.

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.

Shah, A. A., F. Hasan, A. Hameed, and S. Ahmed. (2008). Biological degradation of plastics: A comprehensive review. *Biotechnology Advances 26* (3): 246–265. DOI:10.1016/j.biotechadv.2007.12.005

Shallenberger, E. W. (1981). *The Status of Hawaiian Cetaceans*. Kailua, HI: Manta Corporation.

Shamblin, B. M., D. A. Bagley, L. M. Ehrhart, N. A. Desjardin, R. E. Martin, K. M. Hart, E. Naro-Maciel, K. Rusenko, J. C. Stiner, and D. Sobel. (2015). Genetic structure of Florida green turtle rookeries as indicated by mitochondrial DNA control region sequences. *Conservation Genetics 16* (3): 673–685.

Shane, M. A. (2001). Records of Mexican Barracuda, *Sphyraena ensis*, and Scalloped Hammerhead, *Sphyrna lewini*, from Southern California Associated with Elevated Water Temperatures. *Southern California Academy of Sciences Bulletin* 7.

Shane, S. H. (1995). Relationship between pilot whales and Risso's dolphins at Santa Catalina Island, California, U.S.A. *Marine Ecology Progress Series 123* 5–11.

Shankar, K., J. Ramadevi, B. C. Choudhary, L. Singh, and R. K. Aggarwal. (2004). Phylogeography of olive ridley turtles (*Lepidochelys olivacea*) on the east coast of India: Implications for conservation theory. *Molecular Ecology 13* 1899–1909. DOI:10.1111/j.1365-294X.2004.02195

Shapovalov, L. and A. C. Taft. (1954). *The Life Histories of the Steelhead Rainbow Trout (Salmo gairdneri gairdneri) and Silver Salmon (Oncorhynchus kisutch) With Special Reference to Waddell Creek, California, and Recommendations Regarding Their Management*. San Diego, CA: University of California San Diego.

Sheavly, S. B. (2007). *National Marine Debris Monitoring Program: Final Program Report, Data Analysis, and Summary*. Washington, DC: Ocean Conservancy.

Shields, M., A. Poole, and F. Gill. (2002). Brown Pelican (Pelecanus occidentalis). *The Birds of North America Online* (609): 5. DOI:10.2173/bna.609

Shirasago-Germán, B., E. L. Pérez-Lezama, E. A. Chávez, and R. García-Morales. (2015). Influence of El Niño-Southern Oscillation on the population structure of a sea lion breeding colony in the Gulf of California. *Estuarine, Coastal and Shelf Science 154* 69–76. DOI:10.1016/j.ecss.2014.12.024

Shivji, M., D. Parker, M. Hartwick, J. Smith, and N. A. Slaoan. (1983). Feeding and Distribution Study of the Sunflower Sea Star *Pycnopodia helianthoides* (Brandt, 1835). *Pacific Science 37* (2): 133-140.

Sibley, D. (2014). *The Sibley Guide to Birds* (Second ed.). New York, NY: Alfred A. Knopf.

Sidle, J. G., D. E. Carlson, E. M. Kirsch, and J. J. Dinan. (1992). Flooding: Mortality and habitat renewal for least terns and piping plovers. *Colonial Waterbirds 15* (1): 132–136.

Siegel-Causey, D. and S. Kharitonov. (1990). The evolution of coloniality. *Current Ornithology 7* 285–330.

Sigler, M. F., S. M. Gende, and D. J. Csepp. (2017). Association of foraging Steller sea lions with persistent prey hot spots in southeast Alaska. *Marine Ecological Progress Series 571* 233–243. DOI:10.3354/meps12145

Silber, G. K., M. D. Lettrich, P. O. Thomas, J. D. Baker, M. Baumgartner, E. A. Becker, P. Boveng, D. M. Dick, J. Fiechter, J. Forcada, K. A. Forney, R. B. Griffis, J. A. Hare, A. J. Hobday, D. Howell, K. L. Laidre, N. Mantua, L. Quakenbush, J. A. Santora, K. M. Stafford, P. Spencer, C. Stock, W. Sydeman, K. Van Houtan, and R. S. Waples. (2017). Projecting Marine Mammal Distribution in a Changing Climate. *Frontiers in Marine Science 4* 14. DOI:10.3389/fmars.2017.00413

Simeone, C. A., F. M. Gulland, T. Norris, and T. K. Rowles. (2015). A systematic review of changes in marine mammal health in North America, 1972–2012: The need for a novel integrated approach. *PLoS ONE 10* (11): e0142105. DOI:10.1371/journal.pone.0142105

Simmonds, M. P. and W. J. Eliott. (2009). Climate change and cetaceans: Concerns and recent developments. *Journal of the Marine Biological Association of the United Kingdom 89* (1): 203–210. DOI:10.1017/s0025315408003196

Simmons, S. E., D. E. Crocker, J. L. Hassrick, C. E. Kuhn, P. W. Robinson, Y. Tremblay, and D. P. Costa. (2010). Climate-scale hydrographic features related to foraging success in a capital breeder, the northern elephant seal, *Mirounga angustirostris*. *Endangered Species Research 10* 233–243. DOI:10.3354/esr00254

Simmons, S. E., D. E. Crocker, R. M. Kudela, and D. P. Costa. (2007). Linking foraging behaviour of the northern elephant seal with oceanography and bathymetry at mesoscales. *Marine Ecological Progress Series 346* 265–275.

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

Sin, H., C. Clatterbuck, and K. Rice. (2024). *California Least Tern Breeding Survey: 2018-19 Season Final Report*. San Diego, CA: Wildlife Diversity Program.

Sinaei, M., R. Zare, M. Talebi Matin, and J. Ghasemzadeh. (2021). Marine debris and trace metal (Cu, Cd, Pb, and Zn) pollution in the stranded Green Sea Turtles (*Chelonia mydas*). *Archives of Environmental Contamination and Toxicology 80* 634–644.

Širović, A., S. Baumann-Pickering, J. A. Hildebrand, A. J. Debich, S. T. Herbert, A. Meyer-Löbbecke, A. Rice, B. Thayre, J. S. Trickey, S. M. Wiggins, and M. A. Roch. (2016). *Passive Acoustic Monitoring for Marine Mammals in the SOCAL Range Complex July 2014–May 2015* (MPL Technical Memorandum #607). La Jolla, CA: Marine Physical Laboratory, Scripps Institution of Oceanography, University of California; Department of Computer Science, San Diego State University.

Širović, A., J. A. Hildebrand, S. M. Wiggins, M. A. McDonald, S. E. Moore, and D. Thiele. (2004). Seasonality of blue and fin whale calls and the influence of sea ice in the Western Antarctic Peninsula. *Deep Sea Research II 51* (17–19): 2327–2344. DOI:10.1016/j.dsr2.2004.08.005

Širović, A., A. Rice, E. Chou, J. Hildebrand, S. Wiggins, and M. A. Roch. (2015a). Seven years of blue and fin whale call abundance in the Southern California Bight. *Endangered Species Research 28* 61–76. DOI:10.3354/esr00676

Širović, A., A. Rice, E. Chou, J. A. Hildebrand, S. M. Wiggins, and M. A. Roch. (2015b). Seven years of blue and fin whale call abundance in the Southern California Bight. *Endangered Species Research 28* 61–76. DOI:10.3354/esr00676

Skillman, R. A. and G. H. Balazs. (1992). Leatherback turtle captured by ingestion of squid bait on swordfish longline. *Fishery Bulletin 90* (4): 807–808.

Skillman, R. A. and P. Kleiber. (1998). *Estimation of Sea Turtle Take and Mortality in the Hawaii-based Longline Fishery, 1994–1996* (NOAA Technical Memorandum NMFS-SWFSC-257). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Small, D. P., P. Calosi, D. Boothroyd, S. Widdicombe, and J. I. Spicer. (2016). The sensitivity of the early benthic juvenile stage of the European lobster *Homarus gammarus* (L.) to elevated $p\text{CO}_2$ and temperature. *Marine Biology 163* (53). DOI:10.1007/s00227-016-2834-x

Smith, A., T. Stehly, and W. Musial. (2015). *2014 U.S. Offshore Wind Market Report: Industry Trends, Technology Advancement, and Cost Reduction*. Baltimore, MD: National Renewable Energy Laboratory.

Smith, A. B., M. Kissling, A. M. Capuano, S. B. Lewis, and T. A. Mooney. (2023). Aerial hearing thresholds and ecoacoustics of a threatened pursuit-diving seabird, the marbled murrelet *Brachyramphus marmoratus*. *Endangered Species Research 50* 167-179. DOI:10.3354/esr01234

Smith, B. D., G. Braulik, S. Strindberg, R. Mansur, M. A. A. Diyan, and B. Ahmed. (2009). Habitat selection of freshwater-dependent cetaceans and the potential effects of declining freshwater flows and sea-level rise in waterways of the Sundarbans mangrove forest, Bangladesh. *Aquatic Conservation: Marine and Freshwater Ecosystems 19* (2): 209–225. DOI:10.1002/aqc.987

Smith, C. E., B. J. Hurley, C. N. Toms, A. D. Mackey, M. Solangi, and S. A. Kuczaj Ii. (2013). Hurricane impacts on the foraging patterns of bottlenose dolphins *Tursiops truncatus* in Mississippi Sound. *Marine Ecology Progress Series 487* 231–244. DOI:10.3354/meps10372

Smith, G., C. Stamm, and F. Petrovic. (2003). *Haliotis cracherodii*. *IUCN Red List of Threatened Species. Version 2010.1.* Retrieved from www.iucnredlist.org.

Smith, J. E., C. L. Hunter, and C. M. Smith. (2002). Distribution and reproductive characteristics of nonindigenous and invasive marine algae in the Hawaiian Islands. *Pacific Science 56* (3): 299–315.

Smith, R. C., P. Dustan, D. Au, K. S. Baker, and E. A. Dunlap. (1986). Distribution of cetaceans and sea-surface chlorophyll concentrations in the California current. *Marine Biology 91* 385–402.

Smultea, M. (2012). Short note: Bryde’s Whale (*Balaenoptera brydei/edeni*) sightings in the Southern California Bight. *Aquatic Mammals 38* (1): 92–97. DOI:10.1578/am.38.1.2012.92

Smultea, M. A., C. E. Bacon, and J. S. D. Black. (2011). *Aerial Survey Marine Mammal Monitoring off Southern California in Conjunction with US Navy Major Training Events (MTE), July 27- August 3 and September 23–28, 2010—Final Report, June 2011*. Prepared for Commander, U.S. Pacific Fleet, Pearl Harbor, HI. Submitted to Naval Facilities Engineering Command Pacific (NAVFAC), EV2 Environmental Planning, Pearl Harbor, HI, 96860 3134, under Contract No. N00244-10-C-0021 issued to University of California, San Diego, 7835 Trade St., San Diego, CA 92121. Submitted by Smultea Environmental Sciences (SES), Issaquah, WA, 98027, www.smultea.com, under Purchase Order No. 10309963.

Smultea, M. A., K. Bacon, B. Wursig, and K. Lomac-McNair. (2014, May 2-3, 2014). *Behaviors of Southern California Cetaceans: Observations from a Small Aircraft 2008–2013*. Presented at the Southern California Academy of Sciences 107th Annual Meeting. California State University, Santa Monica, CA.

Smultea, M. A., A. B. Douglas, C. E. Bacon, T. A. Jefferson, and L. Mazzuca. (2012). Bryde's whale (*Balaenoptera brydei/edeni*) sightings in the southern California bight. *Aquatic Mammals 38* (1): 92–97. DOI:10.1578/AM.38.1.2012.92

Smultea, M. A. and T. Jefferson. (2014). Changes in Relative Occurrence of Cetaceans in the Southern California Bight: A Comparison of Recent Aerial Survey Results with Historical Data Sources. *Aquatic Mammals 40* (1): 32–43. DOI:10.1578/AM.40.1.2014.32

Smultea, M. A., T. A. Jefferson, and A. M. Zoidis. (2010). Rare sightings of a Bryde's whale (*Balaenoptera edeni*) and Sei whales (*B. borealis*) (Cetacea: Balaenopteridae) northeast of Oahu, Hawaii. *Pacific Science 64* (3): 449–457. DOI:10.2984/64.3.449

Snieszko, S. F. (1978). Control of Fish Diseases. *Marine Fisheries Review 40* (3): 65–68.

Soldevilla, M. S. (2008). *Risso's and Pacific white-sided dolphins in the Southern California Bight: Using echolocation clicks to study dolphin ecology.* (Published doctoral dissertation). University of California, San Diego, San Diego, CA.

Soldevilla, M. S., E. E. Henderson, G. S. Campbell, S. M. Wiggins, J. A. Hildebrand, and M. A. Roch. (2008). Classification of Risso's and Pacific white-sided dolphins using spectral properties of echolocation clicks. *The Journal of Acoustical Society of America 124* (1): 609–624. DOI:10.1121/1.2932059

Solé, M., K. Kaifu, T. A. Mooney, S. L. Nedelec, F. Olivier, A. N. Radford, M. Vazzana, M. A. Wale, J. M. Semmens, S. D. Simpson, G. Buscaino, A. Hawkins, N. Aguilar de Soto, T. Akamatsu, L. Chauvaud, R. D. Day, Q. Fitzgibbon, R. D. McCauley, and M. André. (2023). Marine invertebrates and noise. *Frontiers in Marine Science 10*. DOI:10.3389/fmars.2023.1129057

Southern California Marine Institute. (2016). *Southern California Offshore Banks, National Marine Sanctuary Proposal*. Los Angeles, CA: Marine Institute and the Vantuna Research Group and Moore Laboratory of Zoology, Occidental College.

Southwood, A. L., R. D. Andrews, M. E. Lutcavage, F. V. Paladino, N. H. West, R. H. George, and D. R. Jones. (1999). Heart rates and diving behavior of leatherback sea turtles in the eastern Pacific Ocean. *The Journal of Experimental Biology 202* 1115–1125.

Spalding, H. L., J. M. Copus, B. W. Bowen, R. K. Kosaki, K. Longenecker, A. D. Montgomery, J. L. Padilla-Gamiño, F. A. Parrish, M. S. Roth, S. J. Rowley, R. J. Toonen, and R. L. Pyle. (2019). The Hawaiian Archipeligo. In Y. Loya, K. A. Puglise, & T. C. I. Bridge (Eds.), *Mesophotic Coral Ecosystems* (pp. 445-464). Cham, Switzerland: Springer Nature Switzerland.

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.

Spalding, M. D., H. E. Fox, G. R. Allen, N. Davidson, Z. A. Ferdaña, M. Finlayson, B. S. Halpern, M. A. Jorge, A. Lombana, S. A. Lourie, K. D. Martin, E. McManus, J. Molnar, C. A. Recchia, and J. Roberston. (2007). Marine ecoregions of the world: A bioregionalisation of coastal and shelf areas. *Bioscience 57* (7): 573–583.

Spalding, M. D., C. Ravilious, and E. P. Green. (2001). *World Atlas of Coral Reefs*. Berkeley, CA: University of California Press.

Spatz, D. R., K. M. Newton, R. Heinz, B. Tershy, N. D. Holmes, S. H. Butchart, and D. A. Croll. (2014). The biogeography of globally threatened seabirds and island conservation opportunities. *Conservation Biology 28* (5): 1282–1290. DOI:10.1111/cobi.12279

Spear, L. B., D. G. Ainley, N. Nur, and S. N. G. Howell. (1995). Population size and factors affecting at-sea distributions of four endangered procellariids in the tropical Pacific. *The Condor 97* (3): 613–638.

Spence, B. C., E. P. Bjorkstedt, J. C. Garza, J. J. Smith, D. G. Hankin, D. Fuller, W. E. Jones, R. Macedo, T. H. Williams, and E. Mora. (2008). *A Framework for Assessing the Viability of Threatened and*

Endangered Salmon and Steelhead in the North-Central California Coast Recovery Domain. La Jolla, CA: National Marine Fisheries Service, Southwest Fisheries Science Center.

Spence, B. C. and T. H. Williams. (2011). *Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Central California Coast coho salmon ESU*. Santa Cruz, CA: National Marine Fisheries Service, Southwest Fisheries Science Center.

Speybroeck, J., D. Bonte, W. Courtens, T. Gheskiere, P. Grootaert, J. P. Maelfait, S. Provoost, K. Sabbe, E. W. M. Stienen, V. Van Lancker, W. Van Landuyt, M. Vincx, and S. Degraer. (2008). The Belgian sandy beach ecosystem: A review. *Marine Ecology–an Evolutionary Perspective 29* (Supplement 1): 171–185.

Spina, A. P., M. A. Allen, and M. Clarke. (2005). Downstream migration, rearing abundance, and pool habitat associations of juvenile steelhead in the lower main stem of a south-central California stream. *North American Journal of Fisheries Management 25* (3): 919–930.

Spotila, J. R. (2004). *Sea Turtles: A Complete Guide to Their Biology, Behavior, and Conservation*. Baltimore, MD: John Hopkins University Press.

Stafford, K. M., D. R. Bohnenstiehl, M. Tolstoy, E. Chapp, D. K. Mellinger, and S. E. Moore. (2004). Antarctic-type blue whale calls recorded at low latitudes in the Indian and eastern Pacific Oceans. *Deep Sea Research Part I: Oceanographic Research Papers 51* (10): 1337–1346. DOI:10.1016/j.dsr.2004.05.007

Stafford, K. M., S. L. Nieukirk, and G. G. Fox. (2001). Geographic and seasonal variation of blue whale calls in the North Pacific. *Journal of Cetacean Research Management 3* (1): 65–76.

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

Starbird, C. H., A. Baldridge, and J. T. Harvey. (1993). Seasonal occurrence of leatherback sea turtles (*Dermochelys coriacea*) in the Monterey Bay region, with notes on other sea turtles, 1986–1991. *California Fish and Game 79* (2): 54–62.

State of California. (2009). *California Ocean Plan: Water Quality Control Plan for the Ocean Water of California*. Sacramento, CA: California Environmental Protection Agency.

State of Hawaii Department of Health. (2014). *2014 State of Hawaii Water Quality Monitoring and Assessment Report* (Integrated Report to the U.S. Environmental Protection Agency and the U.S. Congress Pursuant to §303(d) and §305(b), Clean Water Act [P.L. 97-117]). Honolulu, HI: Clean Water Branch.

Steneck, R. S., M. H. Graham, B. J. Bourque, D. Corbett, J. M. Erlandson, J. A. Estes, and M. J. Tegner. (2002). Kelp forest ecosystems: Biodiversity, stability, resilience and future. *Environmental Conservation 29* (4): 436–459. DOI:10.1017/S0376892902000322

Sterling, J. T. and R. R. Ream. (2004). At-sea behavior of juvenile male northern fur seals (*Callorhinus ursinus*). *Canadian Journal of Zoology 82* (10): 1621–1637. DOI:10.1139/z04-136

Sterling, J. T., A. M. Springer, S. J. Iverson, S. P. Johnson, N. A. Pelland, D. S. Johnson, M. A. Lea, and N. A. Bond. (2014). The sun, moon, wind, and biological imperative-shaping contrasting wintertime migration and foraging strategies of adult male and female northern fur seals (*Callorhinus ursinus*). *PLoS ONE 9* (4): e93068. DOI:10.1371/journal.pone.0093068

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

Stevens, P. W., D. A. Blewett, and J. P. Casey. (2006). Short-term effects of a low dissolved oxygen event on estuarine fish assemblages following passage of Hurricane Charley. *Estuaries and Coasts 29* (6A): 997–1003.

Stewart, B. (1981). The Guadalupe fur seal (*Arctocephalus townsendi*) on San Nicolas Island, California. *Bulletin of the Southern California Academy of Sciences 80* (3): 134–136.

Stewart, B. S. (1997). Ontogeny of differential migration and sexual segregation in northern elephant seals. *Journal of Mammalogy 78* (4): 1101–1116.

Stewart, B. S., G. A. Antonelis, J. D. Baker, and P. K. Yochem. (2006). Foraging biogeography of Hawaiian monk seals in the Northwestern Hawaiian Islands. *Atoll Research Bulletin 543* 131–146.

Stewart, B. S. and R. L. DeLong. (1994). Postbreeding foraging migrations of northern elephant seals. In B. J. Le Boeuf & R. M. Laws (Eds.), *Elephant Seals: Population Ecology, Behavior, and Physiology* (pp. 290–309). Berkeley, CA: University of California Press.

Stewart, B. S. and R. L. DeLong. (1995). Double migrations of the northern elephant seal, *Mirounga angustirostris*. *Journal of Mammalogy 76* (1): 196–205.

Stewart, B. S. and H. R. Huber. (1993). *Mirounga angustirostris*. *Mammalian Species 449* 1–10.

Stewart, B. S., P. K. Yochem, R. L. DeLong, and G. A. Antonelis. (1993a). Trends in abundance and status of pinnipeds on the Southern California Channel Islands. In F. G. Hochberg (Ed.), *Third California Islands Symposium: Recent Advances in Research on the California Islands* (pp. 501–516). Santa Barbara, CA: Santa Barbara Museum of Natural History.

Stewart, B. S., P. K. Yochem, R. L. DeLong, and G. A. Antonelis. (1993b). Trends in abundance and status of pinnipeds on the southern California Channel Islands. In F. G. Hochberg (Ed.), *Third California Islands Symposium: Recent Advances in Research on the California Islands* (pp. 501–516). Santa Barbara, CA: Santa Barbara Museum of Natural History.

Stewart, B. S., P. K. Yochem, H. R. Huber, R. L. DeLong, R. J. Jameson, W. J. Sydeman, S. G. Allen, and B. J. Le Boeuf. (1994). History and present status of the northern elephant seal population. In B. J. Le Boeuf & R. M. Laws (Eds.), *Elephant Seals: Population Ecology, Behavior, and Physiology* (pp. 29–48). Berkeley, CA: University of California Press.

Stewart, J. D., C. S. Beale, D. Fernando, A. B. Sianipar, R. S. Burton, B. X. Semmens, and O. Aburto-Oropeza. (2016a). Spatial ecology and conservation of *Manta birostris* in the Indo-Pacific. *Biological Conservation 200* 178-183.

Stewart, J. G. and B. Myers. (1980). Assemblages of Algae and Invertebrates in Southern California *Phyllospadix*-Dominated Intertidal Habitats. *Aquatic Botany 9* 73–94.

Stewart, K. R., E. L. LaCasella, S. E. Roden, M. P. Jensen, L. W. Stokes, S. P. Epperly, and P. H. Dutton. (2016b). Nesting population origins of leatherback turtles caught as bycatch in the US pelagic longline fishery. *Ecosphere 7* (3): 1–18.

Stierhoff, K. L., M. Neuman, and J. L. Butler. (2012). On the road to extinction? Population declines of the endangered white abalone, *Haliotis sorenseni*. *Biological Conservation 152* 46–52. DOI:10.1016/j.biocon.2012.03.013

Stierhoff, K. L., M. Neumann, S. A. Mau, and D. W. Murfin. (2014a). *White Abalone at San Clemente Island: Population Estimates and Management Recommendations*. Long Beach, CA: National Marine Fisheries Service.

Stierhoff, K. L., M. Neumann, S. A. Mau, and D. W. Murfin. (2014b). *White Abalone at San Clemente Island: Population Estimates and Management Recommendations*. Long Beach, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Stillman, J. H. and A. W. Paganini. (2015). Biochemical adaptation to ocean acidification. *The Journal of Experimental Biology 218* (Pt 12): 1946–1955. DOI:10.1242/jeb.115584

Stillwater Sciences. (2024). *Rescue, Reintroduction, and Genetic Conservation for Southern California Steelhead – Evaluation and Guidance*. Ventura, CA: Stillwater Sciences.

Stinson, M. L. (1984). *Biology of Sea Turtles in San Diego Bay, California, and in the Northeastern Pacific Ocean.* (Unpublished master's thesis). San Diego State University, San Diego, CA.

Stokes, H. J., N. Esteban, and G. C. Hays. (2024). Predation of sea turtle eggs by rats and crabs. *Marine Biology 171* (1): 17.

Stout, H. A., P. W. Lawson, D. L. Bottom, T. D. Cooney, M. J. Ford, C. E. Jordan, R. G. Kope, L. M. Kruzic, G. R. Pess, G. H. Reeves, M. D. Scheuerell, T. C. Wainwright, R. S. Waples, E. Ward, L. A. Weitkamp, J. G. Williams, and T. H. Williams. (2012). *Scientific conclusions of the status review for Oregon coast coho salmon (Oncorhynchus kisutch)*. Seattle, WA: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center.

Strachan, G., M. McAllister, and C. J. Ralph. (1995). *Marbled Murrelet at-sea and foraging behavior. Ecology and Conservation of the Marbled Murrelet* (General Technical Report PSW-152). Washington, DC: U.S. Department of Agriculture, Forest Service.

Strong, C. (2020). *Marbled Murrelet Population Monitoring At Sea in Conservation Zone 4 During 2019: Results From Southern Oregon and Northern California*. Crescent City, CA: U.S. Fish and Wildlife Service.

Sumich, J. L. and I. T. Show. (2011). Offshore migratory corridors and aerial photogrammetric body length comparisons of southbound gray whales, *Eschrichtius robustus*, in the Southern California Bight, 1988–1990. *Marine Fisheries Review 73* (1): 28–34.

Suzuki, D., T. Miyamoto, T. Kikawada, M. Watanabe, and T. Suzuki. (2014). A leech capable of surviving exposure to extremely low temperatures. *PLoS ONE 9* (1): e86807.

Svejkovsky, J., N. P. Nezlin, N. M. Mustain, and J. B. Kum. (2010). Tracking stormwater discharge plumes and water quality of the Tijana River with multispectral aerial imagery. *Estuarine, Coastal and Shelf Science 87* (3): 387–398.

Swartz, S. L., B. L. Taylor, and D. J. Rugh. (2006). Gray whale, *Eschrichtius robustus,* population and stock identity. *Mammal Review 36* (1): 66–84.

Sydeman, W. J. and S. G. Allen. (1999). Pinniped population dynamics in central California: Correlations with sea surface temperature and upwelling indices. *Marine Mammal Science 15* (2): 446–461.

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*.

Szesciorka, A. R., L. T. Ballance, A. Širović, A. Rice, M. D. Ohman, J. A. Hildebrand, and P. J. S. Franks. (2020). Timing is everything: Drivers of interannual variability in blue whale migration. *Scientific Reports 10* (1). DOI:10.1038/s41598-020-64855-y

Talley, T. S., P. K. Dayton, and S. E. Ibarra-Obando. (2000). Tidal flat macrofaunal communities and their associated environments in estuaries of Southern California and Northern Baja California, Mexico. *Estuaries 23* (1): 97–114.

Teo, S. L., P. T. Sandstrom, E. D. Chapman, R. E. Null, K. Brown, A. P. Klimley, and B. A. Block. (2013). Archival and acoustic tags reveal the post-spawning migrations, diving behavior, and thermal habitat of hatchery-origin Sacramento River steelhead kelts (*Oncorhynchus mykiss*). *Environmental Biology of Fishes 96* 175–187.

Teresa, A. R. M. (2021). *Assessing temporal and spatial patterns in habitat use of green sea turtles (Chelonia mydas) in Pearl Harbor, Hawaii.* (M.S.). University of Lisbon, Lisbon, Portugal. Retrieved from http://hdl.handle.net/10451/48470.

Teuten, E. L., S. J. Rowland, T. S. Galloway, and R. C. Thompson. (2007). Potential for plastics to transport hydrophobic contaminants. *Environmental Science and Technology 41* (22): 7759–7764. DOI:10.1021/es071737s

The Nature Conservancy. (2015, August 25, 2015). *Reef Resilience: Changes in Storm Patterns*. Retrieved June 23, 2016, from http://www.reefresilience.org/coral-reefs/stressors/climate-and-ocean-change/changes-in-storm-patterns/.

The Nature Conservancy. (2016, August 30, 2016). *Reef Resilience: Sea-level Rise*. Retrieved 02/15/2017, 2017, from http://www.reefresilience.org/coral-reefs/stressors/climate-and-ocean-change/sea-level-rise/.

Theisen, T. C., B. W. Bowen, W. Lanier, and J. D. Baldwin. (2008). High connectivity on a global scale in the pelagic wahoo, *Acanthocybium solandri* (tuna family Scombridae). *Molecular Ecology 17* (19): 4233–4247.

Thiel, M. and I. Kruse. (2001). Status of the Nemertea as predators in marine ecosystems. *Hydrobiologia 456* 21–32.

Thiessen, G. J. (1958). Threshold of hearing of a ring-billed gull. *The Journal of the Acoustical Society of America 30* (11): 1047.

Thomas, K., J. T. Harvey, T. Goldstein, J. Barakos, and F. Gulland. (2009). Movement, dive behavior, and survival of California sea lions (*Zalophus californianus*) posttreatment for domoic acid toxicosis. *Marine Mammal Science 26* (1): 36–52. DOI:10.1111/j.1748-7692.2009.00314.x

Thometz, N. M., T. L. Kendall, B. P. Richter, and T. M. Williams. (2016). The high cost of reproduction in sea otters necessitates unique physiological adaptations. *The Journal of Experimental Biology 219* (Pt 15): 2260–2264. DOI:10.1242/jeb.138891

Thompson, B., J. Dixon, S. Schroeter, and D. J. Reish. (1993). Benthic Invertebrates. In M. D. Dailey, D. J. Reish, & J. W. Anderson (Eds.), *Ecology of the Southern California Bight: A Synthesis and Interpretation* (pp. 369–458). Berkeley, CA: University of California Press.

Thompson, B., A. R. Melwani, and J. A. Hunt. (2009). *Estimated Sediment Contaminant Concentrations Associated with Biological Impacts at San Diego Bay Clean-up Sites*. Oakland, CA: Aquatic Science Center.

Thompson, B. C., J. A. Jackson, J. Burger, L. A. Hill, E. M. Kirsch, and J. L. Atwood. (1997). Least Tern (*Sterna antillarum*). *The Birds of North America Online* (290): 1–32. DOI:10.2173/bna.290

Thompson, S. A., W. J. Sydeman, J. A. Santora, B. A. Black, R. M. Suryan, J. Calambokidis, W. T. Peterson, and S. J. Bograd. (2012). Linking predators to seasonality of upwelling: Using food web indicators and path analysis to infer trophic connections. *Progress in Oceanography 101* (1): 106–120. DOI:10.1016/j.pocean.2012.02.001

Tierra Data. (2008). *Naval Auxiliary Landing Field, San Clemente Island Black Abalone (Haliotis cracherodii) Survey.* San Diego, CA: Naval Facilities Engineering Command Southwest.

Tierra Data Inc. (2018). *San Diego Bay Avian Species Surveys 2016-2017*. San Diego, CA: Naval Facilities Engineering Command Southwest.

Tinker, M. T., D. P. Costa, J. A. Estes, and N. Wieringa. (2007). Individual dietary specialization and dive behaviour in the California sea otter: Using archival time–depth data to detect alternative foraging strategies. *Deep Sea Research Part II: Topical Studies in Oceanography 54* (3-4): 330–342. DOI:10.1016/j.dsr2.2006.11.012

Tinker, M. T., J. A. Estes, K. Ralls, T. M. Williams, D. Jessup, and D. P. Costa. (2006). *Population Dynamics and Biology of the California Sea Otter (Enhydra lutris nereis) at the Southern End of its Range, Final Technical Summary, Final Study Report*. Santa Barbara, CA: Minerals Management Service.

Tinker, M. T., J. Tomoleoni, N. LaRoche, L. Bowen, A. K. Miles, M. Murray, M. Staedler, and Z. Randell. (2017). *Southern Sea Otter Range Expansion and Habitat Use in the Santa Barbara Channel, California* (Open-File Report 2017–1001). Reston, VA: U.S. Geological Survey.

Tinker, M. T., J. L. Yee, K. L. Laidre, B. B. Hatfield, M. D. Harris, J. A. Tomoleoni, T. W. Bell, E. Saarman, L. P. Carswell, and A. K. Miles. (2021). Habitat features predict carrying capacity of a recovering marine carnivore. *Journal of Wildlife Management 85* (3): 302–323.

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.

Tittensor, D. P., A. R. Baco, P. E. Brewin, M. R. Clark, M. Consalvey, J. Hall-Spencer, A. A. Rowden, T. Schlacher, K. I. Stocks, and A. D. Rogers. (2009). Predicting global habitat suitability for stony corals on seamounts. *Journal of Biogeography 36* 1111–1128. DOI:10.1111/j.1365-2699.2008.02062

Tomaszewicz, T., C. N, L. Avens, E. L. LaCasella, T. Eguchi, P. H. Dutton, R. A. LeRoux, and J. A. Seminoff. (2022). Mixed-stock aging analysis reveals variable sea turtle maturity rates in a recovering population. *The Journal of Wildlife Management 86* (4): e22217.

Tomich, P. Q. (1986). *Mammals in Hawaii* (2nd ed.). Honolulu, HI: Bishop Museum Press.

Torres, M. E., J. McManus, and C. Huh. (2002). Fluid seepage along the San Clemente fault scarp: Basin-wide impact on barium cycling. *Earth and Planetary Science Letters 203* 181–194.

Towle, E. K., A. C. Baker, and C. Langdon. (2016). Preconditioning to high CO_2 exacerbates the response of the Caribbean branching coral *Porites porites* to high temperature stress. *Marine Ecology Progress Series 546* 75–84. DOI:10.3354/meps11655

Treible, L. M., K. A. Pitt, S. G. Klein, and R. H. Condon. (2018). Exposure to elevated pCO_2 does not exacerbate reproductive suppression of *Aurelia aurita* jellyfish polyps in low oxygen environments. *Marine Ecology Progress Series 591* 129–139. DOI:10.3354/meps12298

Tsao, F. and L. E. Morgan. (2005). Corals That Live On Mountaintops. *The Journal of Marine Education 21* (4): 9–11.

Tulloch, V. J. D., E. E. Plaganyi, C. Brown, A. J. Richardson, and R. Matear. (2018). Future recovery of baleen whales is imperiled by climate change. *Global Change Biology* 1–19. DOI:10.1111/gcb.14573

Turner-Tomaszewicz, C. and J. A. Seminoff. (2012). Turning off the heat: Impacts of power plant decommissioning on green turtle research in San Diego Bay. *Coastal Management 40* (1): 73–87.

Turner, R. E. and N. N. Rabalais. (2003). Linking landscape and water quality in the Mississippi River basin for 200 years. *BioScience 53* (6): 563–572.

Twiss, J. R., Jr. and R. R. Reeves. (1999). *Conservation and Managment of Marine Mammals*. Washington, DC: Smithsonian Institution Press.

Tyack, P., W. Zimmer, D. Moretti, B. Southall, D. Claridge, J. Durban, C. Clark, A. D'Amico, N. DiMarzio, S. Jarvis, E. McCarthy, R. Morrissey, J. Ward, and I. Boyd. (2011). Beaked Whales Respond to Simulated and Actual Navy Sonar. *PLoS ONE 6* (3): 15. DOI:10.1371/journal.pone.0017009.

Tyler, H. R., Jr. and W. G. Pearcy. (1975). The Feeding Habits of Three Species of Lanternfishes (Family Myctophidae) off Oregon, USA. *Marine Biology 32* 7–11.

Tyne, J. A., D. W. Johnston, F. Christiansen, and L. Bejder. (2017). Temporally and spatially partitioned behaviours of spinner dolphins: Implications for resilience to human disturbance. *Royal Society Open Science 4* (1): 160626. DOI:10.1098/rsos.160626

Tyne, J. A., D. W. Johnston, R. Rankin, N. R. Loneragan, L. Bejder, and A. Punt. (2015). The importance of spinner dolphin (*Stenella longirostris*) resting habitat: Implications for management. *Journal of Applied Ecology 52* (3): 621–630. DOI:10.1111/1365-2664.12434

U.S. Army Corps of Engineers. (2002). *Silver Strand Shoreline Final General Reevaluation Report*. Imperial Beach, CA: U.S. Army Corps of Engineers.

U.S. Army Corps of Engineers. (2012). *U.S. Waterway Data*. Retrieved from http://www.ndc.iwr.usace.army.mil/data/datawcus.htm.

U.S. Coast Guard. (2015). *2014 Recreational Boating Statistics*. Washington, DC: U.S. Department of Homeland Security.

U.S. Commission on Ocean Policy. (2004). *An Ocean Blueprint for the 21st Century (Final Report)*. Washington, DC: U.S. Commission on Ocean Policy.

U.S. Department of Agriculture. (1993). Soil Survey Manual. In Soil Conservation Service (Ed.), *U.S. Department of Agriculture Handbook 18* (3rd ed.). Washington, DC: U.S. Department of Agriculture.

U.S. Department of Commerce, National Oceanic and Atmospheric Administration, and National Marine Sanctuary Program. (2008). *Channel Islands National Marine Sanctuary final management plan/final environmental impact statement*. Silver Spring, MD.

U.S. Department of the Navy. (2000). *Quantifying in situ Metal Contaminant Mobility in Marine Sediments*. San Diego, CA: Space and Naval Warfare Systems Command.

U.S. Department of the Navy. (2006a). *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. (2006b). *Rim of the Pacific Exercise After Action Report: Analysis of Effectiveness of Mitigation and Monitoring Measures as Required Under the Marine Mammals Protection Act (MMPA) Incidental Harassment Authorization and the National Defense Exemption from the Requirements of the MMPA for Mid-Frequency Active Sonar Mitigation Measures*.

U.S. Department of the Navy. (2010a). *Remedial Investigation Addendum, Pearl Harbor Sediment (Draft Final)*. Pearl Harbor, HI: Naval Facilities Engineering Command, Pacific.

U.S. Department of the Navy. (2010b). *San Clemente Island Range Assessment (Poster)* [Type]. Presented at the 2010 Strategic Environmental Research and Development Program (SERDP) conference.

U.S. Department of the Navy. (2011). *Marine Species Monitoring for the U.S. Navy's Hawaii Range Complex and the Southern California Range Complex*: U.S. Navy Pacific Fleet.

U.S. Department of the Navy. (2013a). *Comprehensive Exercise and Marine Species Monitoring Report for the U.S. Navy's Southern California Range Complex*. San Diego, CA: U.S. Navy Pacific Fleet.

U.S. Department of the Navy. (2013b). *Integrated Natural Resources Management Plan, Naval Auxiliary Landing Field, San Clemente Island, California*. San Diego, CA: Naval Base Coronado, Natural Resources Office.

U.S. Department of the Navy. (2013c). *Integrated Natural Resources Management Plan, Naval Base Coronado, California*. Washington, DC: U.S. Department of the Navy.

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. (2014). *Marine Species Monitoring For the U.S. Navy's Hawaii Range Complex 2013 Annual Report*. Pearl Harbor, HI: U.S. Navy Pacific Fleet.

U.S. Department of the Navy. (2015). *Monitoring Report for Fuel Pier Replacement Project (P-151) Naval Base Point Loma, San Diego, CA 8 October 2014 to 30 April 2015*. San Diego, CA: Tierra Data Inc.

U.S. Department of the Navy. (2017a). *2017 Report to Congress on the Southern Sea Otter Military Readiness Area Monitoring and Research Plan*. Washington, DC: U.S. Department of the Navy.

U.S. Department of the Navy. (2017b). *Dive Distribution and Group Size Parameters for Marine Species Occurring in the U.S. Navy's Atlantic and Hawaii-Southern California Training and Testing Study Areas*. Newport, RI: Naval Undersea Warfare Center Division.

U.S. Department of the Navy. (2018). *2017 U.S. Navy Annual Marine Species Monitoring Report for the Pacific: A Multi-Range-Complex Monitoring Report For Hawaii-Southern California Training and Testing (HSTT), Mariana Islands Training and Testing (MITT), Northwest Training and Testing (NWTT), and the Gulf of Alaska Temporary Maritime Activities Area (GOA TMAA)*. Silver Spring, MD: National Marine Fisheries Service, Office of Protected Resources.

U.S. Department of the Navy. (2022). *Progress Report: Rocky Intertidal Community Monitoring at Naval Base Point Loma, Spring and Fall 2021*. San Diego, CA: Naval Facilities Engineering Command Southwest.

U.S. Department of the Navy. (2024). *U.S. Navy Marine Species Density Database Phase IV for the Hawaii-California Training and Testing Study Area. Technical Report.* Pearl Harbor, HI: U.S. Pacific Fleet, Environmental Readiness Division,.

U.S. Department of the Navy and National Oceanic and Atmospheric Administration. (2016). *Memorandum of Agreement Between the Department of the Navy and the National Oceanic and Atmospheric Administration, National Marine Fisheries Service Regarding the Establishment of a Seven-Year Plan to Support the Recovery of the Endangered White Abalone at Tanner and Cortes Banks*: Department of the Navy and National Marine Fisheries Service.

U.S. Department of the Navy, U.S. Fish and Wildlife Service, and U.S. Geological Survey. (2016). *Monitoring and Research Plan for Southern Sea Otter Military Readiness Areas*. Washington, DC: U.S. Department of the Navy.

U.S. Environmental Protection Agency. (2004). *Regional Analysis Document for Cooling Water Intake Structues-CWA 316(b), Phase II-Large existing electric generating plants*.

U.S. Environmental Protection Agency. (2008a). *Interim Drinking Water Health Advisory For Perchlorate*. Washington, DC: Health and Ecological Criteria Division.

U.S. Environmental Protection Agency. (2008b). *Toxicity Characteristic Leaching Procedure (TCLP) for VOCs, SVOCs, Chlorinated Pesticides and Herbicides, and Metals by SW-846 Method 1311 and Analysis*. Washington, DC: U.S. Environmental Protection Agency.

U.S. Environmental Protection Agency. (2009). *National Recommended Water Quality Criteria*. Washington, DC: Office of Water and the Office of Science and Technology. Retrieved from http://www.epa.gov/ost/criteria/wqctable/.

U.S. Environmental Protection Agency. (2010). *National Marine Debris Monitoring Program: Lessons Learned*. Virginia Beach, VA: U.S. Environmental Protection Agency.

U.S. Environmental Protection Agency. (2012). *National Coastal Condition Report IV*. Washington, DC: Office of Research and Development/Office of Water. Retrieved from http://water.epa.gov/type/oceb/assessmonitor/nccr/index.cfm.

U.S. Environmental Protection Agency. (2016). *National Coastal Condition Assessment 2010*. Washington, DC: Office of Water and Office of Research and Development. Retrieved from http://www.epa.gov/national-aquatic-resource-surveys/ncca.

U.S. Environmental Protection Agency. (2024). *Southern California Ocean Disposal Site #2 Investigation*. Retrieved September 9, 2024, from https://www.epa.gov/ocean-dumping/southern-california-ocean-disposal-site-2-investigation.

U.S. Fish & Wildlife Service. (2016). *5-Year Review Short Form Summary: Newell's Shearwater (Puffinus auricularis newelli)*. Honolulu, HI: Pacific Islands Fish and Wildlife Office.

U.S. Fish & Wildlife Service. (2024a). *5-Year Review: Summary and Evaluation, Marbled Murrelet (Brachyramphus marmoratus) CA, OR, WA DPS*. Lacey, WA: Washington Fish and Wildlife Office.

U.S. Fish & Wildlife Service. (2024b). *Avian Influenza*. Retrieved September 17, 2024, from https://www.fws.gov/avian-influenza.

U.S. Fish and Wildlife Service. (1983). *Hawaiian Dark-rumped Petrel and Newell's Manx Shearwater Recovery Plan*. Portland, OR: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (1985). *Recovery Plan for the California Least Tern Sterna Antillarum brownii*. Portland, OR: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (1990). *Recovery Plan for the Interior Population of the Least Tern (Sterna antillarum)*. Twin Cities, MN: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (1996). *Endangered and Threatened Wildlife and Plants; Final Designation of Critical Habitat for the Marbled Murrelet*. Washington, DC: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (1999). *Multi-Species Recovery Plan for South Florida: Coastal Salt Marsh*. Washington, DC: U.S. Department of the Interior, U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2000). Endangered and Threatened Wildlife and Plants; Final Rule To List the Short-Tailed Albatross as Endangered in the United States. *Federal Register 65* (147): 46643–46654.

U.S. Fish and Wildlife Service. (2001). *Short-tailed Albatross (Phoebastria albatrus) Threatened and Endangered Species*. Retrieved from https://www.fws.gov/alaska/fisheries/endangered/pdf/STALfactsheet.pdf.

U.S. Fish and Wildlife Service. (2003). *Hawaiian Islands Birds and Draft Scores (May 2003)*. Portland, OR: U.S. Fish and Wildlife Service, Migratory Birds and Habitat Programs, Pacific Region.

U.S. Fish and Wildlife Service. (2004). *Species Assessment and Listing Priority Assignment Form: Oceanodroma castro*. Washington, DC: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2005a). *Regional Seabird Conservation Plan, Pacific Region*. Portland, OR: U.S. Fish and Wildlife Service, Migratory Birds and Habitat Programs, Pacific Region.

U.S. Fish and Wildlife Service. (2005b). *Short-Tailed Albatross Draft Recovery Plan*. Anchorage, AK: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2006). *California Least Tern (Sternula antillarum browni) 5-year Review. Summary and Evaluation*. Carlsbad, CA: U.S. Fish and Wildlife Service Carlsbad Fish and Wildlife Office.

U.S. Fish and Wildlife Service. (2008a). *Birds of Conservation Concern 2008*. Arlington, VA: U.S. Department of the Interior, Fish and Wildlife Service, Division of Migratory Bird Management.

U.S. Fish and Wildlife Service. (2008b). *Short-tailed Albatross Recovery Plan*. Anchorage, AK: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2008c). *Short-Tailed Albatross Recovery Plan*. Anchorage, AK: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2009). *Marbled Murrelet (Brachyramphus marmoratus) 5-Year Review*. Lacey, WA: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2010a). *Species Account: California Least Tern (Sternula antillarum browni)*.

U.S. Fish and Wildlife Service. (2010b). *Species Account: California Least Tern (Sternula antillarum browni)*. Retrieved from http://fws.gov/sacramento/es/animal_spp_acct/acctbird.htm.

U.S. Fish and Wildlife Service. (2011a). *Hawaiian Dark-rumped Petrel (Pterodroma phaeopygia sandwichensis) 5-Year Review: Summary and Evaluation*. Honolulu, HI: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2011b). *Newell's Shearwater (Puffinus auricularis newelli) 5-year Review: Summary and Evaluation*. Honolulu, HI: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2011c). *Recovery Plan for Hawaiian Waterbirds, Second Revision*. Portland, OR: U.S. Fish and Wildlife Service, Region 1.

U.S. Fish and Wildlife Service. (2012a). Endangered and Threatened Wildlife and Plants; Termination of the Southern Sea Otter Translocation Program; Final Rule. *Federal Register 77* (244): 75266–75297.

U.S. Fish and Wildlife Service. (2012b, September 20). *Endangered Species in the Pacific Islands, Hawaiian Petrel (Pterodroma sandwichensis)*. Retrieved June 1, 2017, from https://www.fws.gov/pacificislands/fauna/uau.html.

U.S. Fish and Wildlife Service. (2012c, September 20). *Endangered Species in the Pacific Islands: Newell's Shearwater (Puffinus auricularis newelli)*. Retrieved June 1, 2017, from https://www.fws.gov/pacificislands/fauna/newellsshearwater.html.

U.S. Fish and Wildlife Service. (2012d, September 20). *Short-Tailed Albatross (Phoebastria albatrus)*. Retrieved June 1, 2017, from https://www.fws.gov/pacificislands/fauna/stalbatross.html.

U.S. Fish and Wildlife Service. (2013). *General Provisions; Revised List of Migratory Birds*. Washington, DC: U.S. Department of the Interior, U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2014). *Short-tailed Albatross (Phoebastria albatrus) 5-Year Review: Summary and Evaluation*. Anchorage, AK: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2015). *Information for Planning and Conservation Lists of Threatened and Endangered Species for the Study Area*. Retrieved December 9, 2015, from https://ecos.fws.gov/ipac/.

U.S. Fish and Wildlife Service. (2017a). *Newell's Townsend's shearwater (Puffinus auricularis newelli) 5-Year Review*. U.S. Fish and Wildlife Service,.

U.S. Fish and Wildlife Service. (2017b). *Species Profile for Band-Rumped storm-petrel (Oceanodroma castro)*. Retrieved June 1, 2017, from https://ecos.fws.gov/ecp0/profile/speciesProfile?spcode=B08V.

U.S. Fish and Wildlife Service. (2020a). *5-year Review Short-tailed Albatross (Phoebastria albatrus)*. Anchorage, AK: Anchorage Fish and Wildlife Conservation Office.

U.S. Fish and Wildlife Service. (2020b). *California Least Tern (Sternula antillarum browni) 5-Year Review: Summary and Evaluation*. Carlsbad, CA: U.S. Fish and Wildlife Service, Carlsbad Fish and Wildlife Office.

U.S. Fish and Wildlife Service. (2021a). *Birds of Conservation Concern 2021*. Washington, DC: U.S. Fish and Wildlife Service, Migratory Birds Program.

U.S. Fish and Wildlife Service. (2021b). *Southern Sea Otter (Enhydra lutris nereis): Stock Assessment Report*. Ventura, CA: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2021c). *Species Report for the Band-rumped storm-petrel (Oceanodroma castro), Version 1.0*. Pacific Islands Fish and Wildlife Office, Honolulu, HI.

U.S. Fish and Wildlife Service. (2022). *5-Year Review Short Form Summary Hawaiian petrel (Pterodroma sandwichensis)*. Honolulu, HI: Pacific Islands Fish and Wildlife Office.

U.S. Geological Survey. (2016, April 19, 2016). *Sea-Level Rise and Climate Change Impacts to Reefs*. *Pacific Coastal and Marine Science Center.* Retrieved June 23, 2016, from http://coralreefs.wr.usgs.gov/climatechg.html.

U.S. National Response Team. (2010). *What are the Effects of Oil on Seagrass?* Washington, DC: U.S. Environmental Protection Agency. Retrieved from http://www.nrt.org/production/NRT/RRTHome.nsf/resources/RRTIV-Pamphlets/$File/27_RRT4_Seagrass_Pamphlet.pdf.

Udyawer, V., M. A. Read, M. Hamann, C. A. Simpfendorfer, and M. R. Heupel. (2013). First record of sea snake (*Hydrophis elegans*, Hydrophiinae) entrapped in marine debris. *Marine pollution bulletin 73* (1): 336–338.

United Nations Environmental Program. (2005). *Marine Litter: An analytical overview*. Nairobi, Kenya: United Nations Environment Programme's Regional Seas Programme.

University of California Berkeley. (2010a). *Introduction to Porifera*. Retrieved from http://www.ucmp.berkeley.edu/porifera/porifera.html.

University of California Berkeley. (2010b). *Introduction to the Bryozoa: "Moss animals"*. Retrieved from http://www.ucmp.berkeley.edu/bryozoa/bryozoa.html.

University of California Berkeley. (2010c). *Introduction to the Foraminifera*. Retrieved from http://www.ucmp.berkeley.edu/foram/foramintro.html.

University of California Berkeley. (2010d). *Introduction to the Platyhelminthes*. Retrieved September 15, 2010, 2010, from http://www.ucmp.berkeley.edu/platyhelminthes/platyhelminthes.html.

University of California Berkeley. (2010e). *Radiolaria: Life history and ecology*. Retrieved from http://www.ucmp.berkeley.edu/protista/radiolaria/rads.html.

University of California Davis. (2017, February 16, 2017). *Spawning success for white abalone*. Retrieved 02/22/2107, 2017, from http://blogs.ucdavis.edu/egghead/2015/01/29/spawning-success-for-white-abalone/.

Urban-Ramirez, J. and D. Aurioles-Gamboa. (1992). First record of the pygmy beaked whale, *Mesoplodon peruvianus* in the North Pacific. *Marine Mammal Science 8* (4): 420–425.

Urbán, J. R., E. Jiménez-López, H. M. Guzmán, and L. Viloria-Gómora. (2021). Migratory Behavior of an Eastern North Pacific Gray Whale From Baja California Sur to Chirikov Basin, Alaska. *Frontiers in Marine Science 8*. DOI:10.3389/fmars.2021.619290

Urrutia, Y. S. and G. H. Dziendzielewski. (2012). *Diagnóstico de la vulnerabilidad de las cuatro especies de pinnípedos (lobo marino, lobo fino, foca de Puerto y elefante marino) en México, frente al cambio climático global* (Translation: Diagnosis of the vulnerability of the four species of pinnipeds (sea lion, fur seal, harbor seal and elephant seal) in Mexico, addressing global climate change.): Documento de el Fondo Sectorial SEMARNAT-CONACYT (FONSEC-SEMARNAT).

Valette-Silver, N. J. (1993). The use of sediment cores to reconstruct historical trends in contamination of estuarine and coastal sediments. *Estuaries 16* (3, Part B: Dedicated Issue: Historical Trends in Contamination of Estuarine and Coastal Sediments: Symposium Papers from the Eleventh Biennial International Estuarine Research Conference): 577–588.

Valverde, R. A., C. M. Orrego, M. T. Tordoir, F. M. Gomex, D. S. Solis, R. A. Hernandez, G. B. Gomez, L. S. Brenes, J. P. Baltodano, L. G. Fonseca, and J. R. Spotila. (2012). Olive Ridley Mass Nesting Ecology and Egg Harvest at Ostional Beach, Costa Rica. *Chelonian Conservation and Biology 11* (1): 1–11.

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, T. V. N. Cole, A. G. Henry, L. Hall, B. Mase-Guthrie, T. Wimmer, and M. J. Moore. (2015). Vessel strikes to large whales before and after the 2008 ship strike rule. *Conservation Letters 8* (1): 24–32. DOI:10.1111/conl.12105

Van Doornik, D. M., D. J. Teel, D. R. Kuligowski, C. A. Morgan, and E. Casillas. (2007). Genetic analyses provide insight into the early ocean stock distribution and survival of juvenile coho salmon off the coasts of Washington and Oregon. *North American Journal of Fisheries Management 27* (1): 220–237.

Van Houtan, K. S., D. L. Francke, S. Alessi, T. T. Jones, S. L. Martin, L. Kurpita, C. S. King, and R. W. Baird. (2016). The developmental biogeography of hawksbill sea turtles in the North Pacific. *Ecology and Evolution 6* (8): 2378–2389. DOI:10.1002/ece3.2034

Van Parijs, S. M. (2015). Letter of introduction to the Biologically Important Areas Issue. *Aquatic Mammals 41* (1): 1–128. DOI:10.1578/AM.41.1.2015.1

Van Soest, R. W., N. Boury-Esnault, J. Vacelet, M. Dohrmann, D. Erpenbeck, N. J. de Voogd, N. Santodomingo, B. Vanhoorne, M. Kelly, and J. N. Hooper. (2012). Global diversity of sponges (Porifera). *PLoS ONE 7* (4): e35105. DOI:10.1371/journal.pone.0035105

Van Waerebeek, K., F. Felix, B. Haase, D. M. Palacios, D. M. Mora-Pinto, and M. Munoz-Hincapie. (1998). Inshore records of the striped dolphin, *Stenella coeruleoalba*, from the Pacific coast of South America. *Reports of the International Whaling Commission 48* 525–532.

Vandenbosch, R. (2000). Effects of ENSO and PDO events on seabird populations as revealed by Christmas bird count data. *Waterbirds 23* (3): 416–422.

Veirs, S., V. Veirs, and J. Wood. (2015). Ship noise in an urban estuary extends to frequencies used for echolocation by endangered killer whales. *PeerJ 4* e1657. DOI:10.7287/peerj.preprints.955v2

Venrick, E. L., T. W. Backman, W. C. Bartram, C. J. Platt, M. S. Thornhill, and R. E. Yates. (1973). Man-made objects on the surface of the central North Pacific Ocean. *Nature 241* (5387): 271–271. DOI:10.1038/241271a0

Verkaik, K., J. Hamel, and A. Mercier. (2016). Carry-over effects of ocean acidification in a cold-water lecithotrophic holothuroid. *Marine Ecology Progress Series 557* 189–206. DOI:doi: 10.3354/meps11868

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

Veron, J. (2000). *Corals of the World*. Cape Ferguson, Australia: Australian Institute of Marine Sciences and CRR Qld Pty Ltd.

Veron, J. (2013). Overview of the taxonomy of zooxanthellate Sceractinia. *Zoological Journal of the Linnean Society 169* 485–508.

Voiland, A. (2013, March 5, 2013). *In a Warming World, Storms May Be Fewer but Stronger*. Retrieved June 23, 2016, 2016, from http://earthobservatory.nasa.gov/Features/ClimateStorms/page1.php.

Von Moos, N., P. Burkhardt-Holm, and A. Kohler. (2012). Uptake and Effects of Microplastics on Cells and Tissue of the Blue Mussel *Mytilus edulis L.* after an Experimental Exposure. *Environmental Science and Technology 46* 11327–11335. DOI:dx.doi.org/10.1021/es302332w

Wade, P. R. (2021). *Estimates of abundance and migratory destination for North Pacific humpback whales in both summer feeding areas and winter mating and calving areas* (SC/68C/IA/03). Seattle, WA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Alaska Fisheries Science Center.

Wade, P. R. and T. Gerrodette. (1993). Estimates of cetacean abundance and distribution in the eastern tropical Pacific. *Reports of the International Whaling Commission 43* 477–493.

Wade, P. R., T. J. Quinn II, J. Barlow, C. S. Baker, A. M. Burdin, J. Calambokidis, P. J. Clapham, E. A. Falcone, J. K. B. Ford, C. M. Gabriele, D. K. Mattila, L. Rojas-Bracho, J. M. Straley, and B. Taylor. (2016). *Estimates of abundance and migratory destination for North Pacific humpback whales in both summer feeding areas and winter mating and calving areas*.

Waggoner, B. and B. R. Speer. (1998, 08 August 1998). *Introduction to the Dinoflagellata*. Retrieved from http://www.ucmp.berkeley.edu/protista/dinoflagellata.html.

Waikiki Aquarium. (2009a, September 2009). *Marine Life Profile: Ghost Crab*. Retrieved from http://www.waquarium.org/marinelifeprofiles_ed.html.

Waikiki Aquarium. (2009b, September 2009). *Marine Life Profile: Hawaiian Slipper Lobsters*. Retrieved from http://www.waquarium.org/marinelifeprofiles_ed.html.

Waikiki Aquarium. (2009c, September 2009). *Marine Life Profile: Hawaiian Spiny Lobster*. Retrieved from http://www.waquarium.org/marinelifeprofiles_ed.html.

Wainwright, P. C. and B. A. Richard. (1995). Predicting patterns of prey use from morphology of fishes. *Environmental Biology of Fishes 44* 97–113. DOI:10.1007/BF00005909

Wallace, B. P., R. L. Lewison, S. L. McDonald, R. K. McDonald, C. Y. Kot, S. Kelez, R. K. Bjorkland, E. M. Finkbeiner, S. Helmbrecht, and L. B. Crowder. (2010). Global patterns of marine turtle bycatch. *Conservation Letters 3* (3): 131–142. DOI:10.1111/j.1755-236x.2010.00105

Wallace, G. T., Jr., G. L. Hoffman, and R. A. Duce. (1977). The influence of organic matter and atmospheric deposition on the particulate trace metal concentration of northwest Atlantic surface seawater. *Marine Chemistry 5* 143–170.

Walter, T. C. and G. Boxshall. (2017, February 22, 2017). *World of Copepods Database*. Retrieved 02/22/2017, 2017, from http://www.marinespecies.org/copepoda/.

Wang, J. Y. and S. C. Yang. (2006). Unusual cetacean stranding events of Taiwan in 2004 and 2005. *Journal of Cetacean Research and Management 8* (3): 283–292.

Wang, J. Y., S. C. Yang, and H. C. Liao. (2001). Species composition, distribution and relative abundance of cetaceans in the waters of southern Taiwan: Implications for conservation and eco-tourism. *Journal of the National Parks of Taiwan 11* (2): 136–158.

Wang, P., K. Richer, L. Rivera-Duarte, B. Davidson, B. Wild, R. Baru, Q. Liao, J. Germano, K. Markille, and J. Gailari. (2014a). *Evaluation of Resuspension from Propeller Wash in Pearl Harbor and San Diego Bay*. San Diego, CA: Space and Naval Warfare Systems Center Pacific.

Wang, P. F., B. Chadwick, W. Choi, C. Jones, W. Wen, and M. Yoshioka. (2009). *Resuspension and Transport of Sediments by Propeller Wash in Pearl Harbor* [Type]. Presented at the Fifth International Conference on Remediation of Contaminated Sediments February 2–5. Jacksonville, FL. Retrieved from www.battelle.org/sedimentscon.

Wang, P. F., D. B. Chadwick, C. Johnson, and J. Grovhoug. (2006). *Modeling Copper and Biocide Concentrations from Hull Paint Leachate in San Diego Bay* (Technical Report 1935). San Diego, CA: SPAWAR Systems Center San Diego.

Wang, P. F., D. Sutton, K. Richter, and B. Chadwick. (2000). *Modeling Migration of Sediment and Sorbed Contaminants Resuspended by Ship Docking in San Diego Bay* [Type]. Presented at the Proceedings of the 4th International Conference on Hydroscience & Engineering.

Wang, Z., Z. Liu, K. Xu, L. M. Mayer, Z. Zhang, A. S. Kolker, and W. Wu. (2014b). Concentrations and sources of polycyclic aromatic hydrocarbons in surface coastal sediments of the northern Gulf of Mexico. *Geochemical Transactions 15* (1): 1.

Ward-Paige, C. A., D. M. Keith, B. Worm, and H. K. Lotze. (2012). Recovery potential and conservation options for elasmobranchs. *Journal of Fish Biology 80* (5): 1844–1869. DOI:10.1111/j.1095-8649.2012.03246

Waring, G. T., T. Hamazaki, D. Sheehan, G. Wood, and S. Baker. (2001). Characterization of beaked whale (Ziphiidae) and sperm whale (*Physeter macrocephalus*) summer habitat in shelf-edge and deeper waters off the northeast U.S. *Marine Mammal Science 17* (4): 703–717.

Warrant, E. J. and N. A. Locket. (2004). Vision in the deep sea. *Biological Reviews 79* 671–712. DOI:10.1017/s1464793103006420

Watson, T. K., K. Hoomanawanui, R. Thurman, B. Thao, and K. Boyne. (2017). *Na Ikena I Kai (Seaward Viewsheds): Inventory of Terrestrial Properties for Assessment of Marine Viewsheds on the Eight MAin HAwaiian Islands*. Camarillo, CA.

Watters, D. L., M. M. Yoklavich, M. S. Love, and D. M. Schroeder. (2010). Assessing marine debris in deep seafloor habitats off California. *Marine Pollution Bulletin 60* 131–138. DOI:10.1016/j.marpolbul.2009.08.019

Watts, A. J. R., C. Lewis, R. M. Goodhead, S. J. Beckett, J. Moger, C. R. Tyler, and T. Galloway. (2014). Uptake and retention of microplastics by the shore crab *Carcinus maenas*. *Environmental Science and Technology 48* 8823–8830. DOI:10.1021/es501090e

Webster, D. L., R. W. Baird, B. K. Rone, and D. B. Anderson. (2015, December 14–18, 2015). *Rough-toothed dolphins on a Navy range in Hawaii: using LIMPET satellite-tag data to assess movements, habitat use, and overlap with Navy activities.* Presented at the Poster presented at the 21st Biennial Conference on the Biology of Marine Mammals. San Francisco, CA. Retrieved from http://www.cascadiaresearch.org/Hawaii/Websteretal_2015_SMM.pdf.

Weise, M. J., D. P. Costa, and R. M. Kudela. (2006). Movement and diving behavior of male California sea lion (*Zalophus californianus*) during anomalous oceanographic conditions of 2005 compared to those of 2004. *Geophysical Research Letters 33* (22).

Weise, M. J., J. T. Harvey, and D. P. Costa. (2010). The role of body size in individual-based foraging strategies of a top marine predator. *Ecology 91* (4): 1004–1015.

Weitkamp, L. and K. Neely. (2002). Coho salmon (*Oncorhynchus kisutch*) ocean migration patterns: Insight from marine coded-wire tag recoveries. *Canadian Journal of Fisheries and Aquatic Science 59* 1100–1115.

Weitkamp, L., T. C. Wainwright, G. J. Bryant, D. J. Teel, and R. G. Kope. (2020). Review of the status of coho salmon from Washington, Oregon, and California *Sustainable Fisheries Management* (pp. 111-118). Boca Raton, FL: CRC Press.

Weitkamp, L. A., T. C. Wainwright, G. J. Bryant, G. B. Milner, D. J. Teel, R. G. Kope, and R. S. Waples. (1995). *Status Review of Coho Salmon from Washington, Oregon, and California*. Seattle, WA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center, Coastal Zone and Estuarine Studies Division.

Welch, J. (2017). *Mouthline pigmentation loss and fisheries associated injuries of rough-toothed dolphins (Steno bredanensis) in Hawaii.* (Master's in environmental studies). Evergreen State College, Olympia, WA.

Weller, D. W., S. Bettridge, R. L. Brownell, J. L. Laake, M. J. Moore, P. E. Rosel, B. L. Taylor, and P. R. Wade. (2013). *Report of the National Marine Fisheries Service Gray Whale Stock Identification Workshop* (NOAA Technical Memorandum NMFS-SWFSC-507). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Weller, D. W., A. M. Burdin, B. Würsig, B. L. Taylor, and R. L. Brownell, Jr. (2002). The western gray whale: A review of past exploitation, current status and potential threats. *Journal of Cetacean Research and Management 4* (1): 7–12.

Weller, D. W., G. S. Campbell, A. Debich, A. G. Kesaris, and R. H. Defran. (2016). *Mark-Recapture Abundance Estimate of California Coastal Stock Bottlenose Dolphins: November 2009 to April 2011* (NOAA Technical Memorandum NMFS-SWFSC-563). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Weller, D. W., A. Klimek, A. L. Bradford, J. Calambokidis, A. R. Lang, B. Gisborne, A. M. Burdin, W. Szaniszlo, J. Urban, A. Gomex-Gallardo Unzueta, S. L. Swartz, and R. L. Brownell Jr. (2012a). Movements of gray whales between the western and eastern North Pacific. *Endangered Species Research 18* 193–199. DOI:10.3354/esr00447

Weller, D. W., A. Klimek, A. L. Bradford, J. Calambokidis, A. R. Lang, B. Gisborne, A. M. Burdin, W. Szaniszlo, J. Urbán, A. Gomez-Gallardo Unzueta, S. Swartz, and R. L. Brownell. (2012b). Movements of gray whales between the western and eastern North Pacific. *Endangered Species Research 18* (3): 193–199. DOI:10.3354/esr00447

Wells, M. L., V. L. Trainer, T. J. Smayda, B. S. O. Karlson, C. G. Trick, R. M. Kudela, A. Ishikawa, S. Bernard, A. Wulffi, D. M. Anderson, and W. P. Cochlan. (2015). Harmful algal blooms and climate change: Learning from the past and present to forecast the future. *Harmful algae 49* (68–93).

Wells, R. S., C. A. Manire, L. Byrd, D. R. Smith, J. G. Gannon, D. Fauqiuer, and K. D. Mullin. (2009). Movements and dive patterns of a rehabilitated Risso's dolphin, *Grampus griseus*, in the Gulf of Mexico and Atlantic Ocean. *Marine Mammal Science 25* (2): 420–429. DOI:10.1111/j.1748-7692.2008.00251

West, K. (2016). *Necropsy Findings, Stomach Contents and Hook Ingestion in IFKW's*. Honolulu, HI: University of Hawaii, Hawaii Pacific University.

West, K. L., S. Sanchez, D. Rotstein, K. M. Robertson, S. Dennison, G. Levine, N. Davis, D. Schofield, C. W. Potter, and B. Jensen. (2012). A Longman's beaked whale (*Indopacetus pacificus*) strands in Maui, Hawaii, with first case of morbillivirus in the central Pacific. *Marine Mammal Science*. DOI:10.1111/j.1748-7692.2012.00616.x

Western Pacific Regional Fishery Management Council. (2001). *Final Fishery Management Plan for Coral Reef Ecosystems of the Western Pacific Region*. Honolulu, HI: Western Pacific Regional Fishery Management Council.

Western Pacific Regional Fishery Management Council. (2009). *Fishery Ecosystem Plan for the Hawaii Archipelago*. Honolulu, HI: Western Pacific Regional Fishery Management Council.

Wever, E. G., P. N. Herman, J. A. Simmons, and D. R. Hertzler. (1969). Hearing in the blackfooted penguin (*Spheniscus demersus*), as represented by the cochlear potentials. *Proceedings of the National Academy of Sciences 63* 676–680.

Wheat, C. G., H. W. Jannasch, J. N. Plant, C. L. Moyer, F. J. Sansone, and G. M. McMurtry. (2000). Continuous sampling of hydrothermal fluids from Loihi Seamount after the 1996 event. *Journal of Geophysical Research 105* (B8): 19353–19367.

Whipple, A., R. M. Grossinger, D. Rankini, B. Stanford, and R. A. Askevold. (2012). *Sacramento San Joaquin Delta Historical Ecology Investigation: Exploring Pattern and Process*. Richmond, CA: San Francisco Estuary Institute.

White, C., J. A. Sayer, and G. M. Gadd. (1997). Microbial solubilization and immobilization of toxic metals: Key biogeochemical processes for treatment of contamination. *FEMS Microbiology Reviews 20* 503–516.

White, H. K., P. Y. Hsing, W. Cho, T. M. Shank, E. E. Cordes, A. M. Quattrini, R. K. Nelson, R. Camilli, A. W. Demopoulos, C. R. German, J. M. Brooks, H. H. Roberts, W. Shedd, C. M. Reddy, and C. R. Fisher. (2012). Impact of the Deepwater Horizon oil spill on a deep-water coral community in the Gulf of Mexico. *Proceedings of the National Academy of Sciences of the United States of America 109* (50): 20303–20308. DOI:10.1073/pnas.1118029109

Whitehead, H. (2003). *Sperm Whales Social Evolution in the Ocean*. Chicago, IL: University of Chicago Press.

Whitehead, H. (2009). Sperm whale, *Physeter macrocephalus*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 1091–1097). Cambridge, MA: Academic Press.

Whitehead, H., A. Coakes, N. Jaquet, and S. Lusseau. (2008). Movements of sperm whales in the tropical Pacific. *Marine Ecology Progress Series 361* 291–300. DOI:10.3354/meps07412

Whitehead, H. and L. Weilgart. (2000). The sperm whale; Social females and roving males. In J. Mann, R. C. Connor, P. L. Tyack, & H. Whitehead (Eds.), *Cetacean Societies; Field Studies of Dolphins and Whales* (pp. 154–172). Chicago, IL: University of Chicago Press.

Whitmire, C. E. and M. E. Clarke. (2007). *State of Deep Coral Ecosystems of the U.S. Pacific Coast: California to Washington* (The State of Deep Coral Ecosystems of the United States). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Whitney, J. L., R. R. Coleman, and M. H. Deakos. (2023). Genomic evidence indicates small island-resident populations and sex-biased behaviors of Hawaiian reef Manta Rays. BMC ecology and evolution. *BMC Ecology and Evolution 23* (31). DOI:10.1186/s12862-023-02130-0

Whitworth, D., J. Koepke, H. Carter, F. Gress, and S. Fangman. (2005). *Nest Monitoring of Xantus's Murrelets (Synthliboramphus hypoleucus) at Anacapa Island, California: 2005 Annual Report*. Davis, CA: California Institute of Environmental Studies.

Whitworth, D. L., S. K. Nelson, S. H. Newman, G. B. Van Vliet, and W. P. Smith. (2000). Foraging distances of radio-marked marbled murrelets from inland areas in southeast Alaska. *The Condor 102* (2). DOI:10.1650/0010-5422(2000)102[0452:FDORMM]2.0.CO;2

Wilcove, D. S. and L. Y. Chen. (1998). Management costs for endangered species. *Conservation Biology 12* (6): 1405–1407.

Wilcox, C., N. J. Mallos, G. H. Leonard, A. Rodriguez, and B. D. Hardesty. (2016). Using expert elicitation to estimate the impacts of plastic pollution on marine wildlife. *Marine Policy 65* 107–114.

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.

Wild, L. A., H. E. Riley, H. C. Pearson, C. M. Gabriele, J. L. Neilson, A. Szabo, J. Moran, J. M. Straley, and S. DeLand. (2023). Biologically Important Areas II for cetaceans within U.S. and adjacent waters – Gulf of Alaska Region. *Frontiers in Marine Science 10*. DOI:10.3389/fmars.2023.1134085

Wilk, R. J., M. G. Raphael, and J. Boloxton, T. D. (2016). Nesting habitat characteristics of Marbled Murrelets occurring in near-shore waters of the Olympic Peninsula, Washington. *Journal of Field Ornithology 87* (2): 162–175. DOI:10.1111/jofo.12150

Wilkens, S. L., J. A. Stanley, and A. G. Jeffs. (2012). Induction of settlement in mussel (*Perna canaliculus*) larvae by vessel noise. *Biofouling 28* (1): 65–72. DOI:10.1080/08927014.2011.651717

Wilkinson, C. (2002). Executive Summary. In C. Wilkinson (Ed.), *Status of Coral Reefs of the World: 2002* (pp. 7–31). Townsville, Australia: Global Coral Reef Monitoring Network.

Williams, R., E. Ashe, L. Blight, M. Jasny, and L. Nowlan. (2014a). Marine mammals and ocean noise: Future directions and information needs with respect to science, policy and law in Canada. *Marine Pollution Bulletin 86* (1-2): 29–38. DOI:10.1016/j.marpolbul.2014.05.056

Williams, R., C. Erbe, E. Ashe, A. Beerman, and J. Smith. (2014b). Severity of killer whale behavioral responses to ship noise: A dose-response study. *Marine Pollution Bulletin 79* (1–2): 254–260. DOI:10.1016/j.marpolbul.2013.12.004

Williams, R. and L. Thomas. (2007). Distribution and abundance of marine mamals in the coastal waters of British Columbia, Canada. *Journal of Cetacean Research and Management 9* (1): 15–28.

Williams, R., S. Veirs, V. Veirs, E. Ashe, and N. Mastick. (2019). Approaches to reduce noise from ships operating in important killer whale habitats. *Marine Pollution Bulletin 139* 459–469. DOI:10.1016/j.marpolbul.2018.05.015

Williams, S. L. (1988). *Thalassia testudinum* productivity and grazing by green turtles in a highly disturbed seagrass bed. *Marine Biology 98* 447–455.

Williams, T. H., S. T. Lindley, B. C. Spence, and D. A. Boughton. (2011). *Status Review Update for Pacific Salmon and Steelhead Listed under the Endangered Species Act: Southwest*. Santa Cruz, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, Fisheries Ecology Division.

Williams, T. H., B. C. Spence, D. A. Boughton, R. C. Johnson, L. Crozier, N. Mantua, M. O'Farrell, and S. T. Lindley. (2016a). *5-Year Review: Summary and Evaluation for California Central Valley Steelhead Distinct Population Segment*. Santa Cruz, CA: National Marine Fisheries Service, West Coast Region.

Williams, T. H., B. C. Spence, D. A. Boughton, R. C. Johnson, L. G. Crozier, N. J. Mantua, M. R. O'Farrell, and S. T. Lindley. (2016b). *Viability Assessment for Pacific Salmon and Steelhead Listed under the Endangered Species Act: Southwest*. Santa Cruz, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Williams, T. H., B. C. Spence, D. A. Boughton, R. C. Johnson, L. G. Crozier, N. J. Mantua, M. R. O'Farrell, and S. T. Lindley. (2016c). *Viability Assessment for Pacific Salmon and Steelhead Listed under the Endangered Species Act: Southwest*. Santa Cruz, CA: National Marine Fisheries Service, Southwest Fisheries Science Center.

Wilson, C. (2002, 23 September 2002). *Giant Kelp (Macrocystis pyrifera)*. Retrieved from http://www.dfg.ca.gov/mlpa/response/kelp.pdf.

Wilson, D., R. Billings, R. Oommen, B. Lange, J. Marik, S. McClutchey, and H. Perez. (2010). *Year 2008 Gulfwide Emission Inventory Study: Draft Report*. Gulf of Mexico OCS Region: Bureau of Ocean Energy Management, Regulation and Enforcement.

Wilson, K., A. D'Amico, and C. Littnan. (2012). *Habitat Use and Behavioral Monitoring of Hawaiian Monk Seals in Proximity to the Navy Hawaii Range Complex*: U.S. Pacific Fleet.

Wilson, M., R. T. Hanlon, P. L. Tyack, and P. T. Madsen. (2007). Intense ultrasonic clicks from echolocating toothed whales do not elicit anti-predator responses or debilitate the squid, *Loligo pealeii*. *Biology Letters 3* 225–227. DOI:10.1098/rsbl.2007.0005

Wilson, M. F., R. H. Armstrong, M. C. Hermans, and K. Koshi. (2006). *Eulachon: A Review of Biology and an Annotated Bibleography*. Juneau, AK: National Marine Fisheries Service, Alaska Fisheries Science Center.

Wilson, S. C. (1978). *Social Organization and Behavior of Harbor Seals, Phoca vitulina concolor, in Maine*. Washington, DC: Smithsonian Institution Press.

Winkler, C., S. Panigada, S. Murphy, and F. Ritter. (2020). *Global Numbers of Ship Strikes: An Assessment of Collisions Between Vessels and Cetaceans Using Available Data in the IWC Ship Strike Database*. Cambridge, United Kingdom: International Whaling Commission.

Wiseman, W. J., Jr. and R. W. Garvine. (1995). Plumes and coastal currents near large river mouths. *Estuaries 18* (3): 509–517.

Witzell, W. N. (1983). *Synopsis of biological data on the hawksbill turtle, Eretmochelys imbricata (Linnaeus, 1766)*. Rome, Italy: United Nations Environment Programme, Food and Agriculture Organization of the United Nations.

Wood, A. C. L., P. K. Probert, A. A. Rowden, and A. M. Smith. (2012). Complex habitat generated by marine bryozoans: A review of its distribution, structure, diversity, threats and conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems 22* (4): 547–563. DOI:10.1002/aqc.2236

Wood, J. B. and C. Day. (2005, 16 June 2006). *CephBase*. Retrieved June 3, 2005, from http://www.cephbase.utmb.edu/.

Wood, J. R. and F. E. Wood. (1980). Reproductive biology of captive green sea turtles *Chelonia mydas*. *American Zoologist 20* (3): 499-505.

Woods Hole Oceanographic Institution. (2015, 2015). *Life at Vents & Seeps*. Retrieved June 13, 2016, from https://www.whoi.edu/main/topic/life-at-vents-seeps.

Wootton, E. C., A. P. Woolmer, C. L. Vogan, E. C. Pope, K. M. Hamilton, and A. F. Rowley. (2012). Increased disease calls for a cost-benefits review of marine reserves. *PLoS ONE 7* (12): e51615. DOI:10.1371/journal.pone.0051615

Work, T. M. and G. H. Balazs. (2013). Tumors in sea turtles: The insidious menace of fibropapillomatosis. *The Wildlife Professional* 44–47.

World Register of Marine Species Editorial Board. (2015). *Towards a World Register of Marine Species*. Retrieved 07/06/2015, 2016, from http://www.marinespecies.org/about.php.

Worthy, G. A. J., P. A. Morris, D. P. Costa, and B. J. Le Boeuf. (1992). Molt energetics of the northern elephant seal. *Journal of Zoology 227* 257–265.

Wren, P. A. and L. A. Leonard. (2005). Sediment transport on the mid-continental shelf in Onslow Bay, North Carolina during Hurricane Isabel. *Estuarine, Coastal and Shelf Science 63* (1–2): 43–56. DOI:10.1016/j.ecss.2004.10.018

Wright, A. J., N. A. Soto, A. L. Baldwin, M. Bateson, C. M. Beale, C. Clark, T. Deak, E. F. Edwards, A. Fernandez, A. Godinho, L. T. Hatch, A. Kakuschke, D. Lusseau, D. Martineau, M. L. Romero, L. S. Weilgart, B. A. Wintle, G. Notarbartolo-di-Sciara, and V. Martin. (2007). Anthropogenic noise as a stressor in animals: A multidisciplinary perspective. *International Journal of Comparative Psychology 20* 250–273.

Wright, S. L., R. C. Thompson, and T. S. Galloway. (2013). The physical impacts of microplastics on marine organisms: A review. *Environmental Pollution 178* 483–492. DOI:http://dx.doi.org/10.1016/j.envpol.2013.02.031

Wu, J. and E. Boyle. (1997). Lead in the western North Atlantic Ocean: Completed response to leaded gasoline phaseout. *Geochimica et Cosmochimica Acta 61* (15): 3279–3283. DOI:0016-7037/97

Würsig, B., T. A. Jefferson, and D. J. Schmidly. (2000). *The Marine Mammals of the Gulf of Mexico*. College Station, TX: Texas A&M University Press.

Wyllie-Echeverria, S. and J. D. Ackerman. (2003). The seagrasses of the Pacific coast of North America. In E. P. Green & F. T. Short (Eds.), *World Atlas of Seagrasses* (pp. 199–206). Berkeley, CA: University of California Press.

Yasugi, M. and M. Hori. (2016). Predominance of parallel- and cross-predation in anglerfish. *Marine Ecology 37* (3): 576–587.

Yee, J. L., J. A. Tomoleoni, M. C. Kenner, J. Fujii, G. B. Bentall, M. T. Tinker, and B. B. Hatfield. (2020). *Southern (California) Sea Otter Population Status and Trends at San Nicolas Island, 2017-2020*. Reston, VA: U.S. Geological Survey.

Yee, J. L., J. A. Tomoleoni, M. C. Kenner, J. A. Fujii, G. B. Bentall, M. M. Staedler, and B. B. Hatfield. (2023). *Southern (California) sea otter population status and trends at San Nicolas Island, 2020–2023* (Open-File Report). Reston, VA.

Yeh, J. (2008). *Depth Zonation of Deep-Sea Megafaunal Scavengers of the Hawaiian Islands.* (Thesis for a Masters of Science in Oceanography). University of Hawaii, Honolulu, HI.

Yender, R. A., J. Michel, and R. Hoff. (2010). *Oil Spills in Coral Reefs: Planning & Response Considerations*. Silver Spring, MD: National Oceanic and Atmospheric Administration.

Yoklavich, M., T. Laidig, A. Taylor, D. Watters, L. Krigsman, and M. Love. (2013). *A Characterization of the Christmas Tree Black Coral (Antipathes dendrochristos) Community on Three Seamounts in the*

Southern California Bight from a Survey using a Manned Submersible. Santa Barbara, CA: National Oceanic and Atmospheric Administration Deep-Sea Coral Research and Technology Program.

Yoklavich, M. M., T. E. Laidig, K. Graiff, M. E. Clarke, and C. E. Whitmire. (2017). Incidence of disturbance and damage to deep-sea corals and sponges in areas of high trawl bycatch near the California and Oregon border. *Deep-Sea Research Part II*.

Yoshiyama, R. M., F. W. Fisher, and P. B. Moyle. (1998). Historical abundance and decline of Chinook salmon in the Central Valley region of California. *North American Journal of Fisheries Management 18* (3): 487–521.

Young, C. N., J. Carlson, C. Hutt, D. Kobayashi, C. T. McCandless, and J. Wraith. (2016a). *Status review report: Oceanic whitetip shark (Carcharhinius longimanus)* (Final Report to the National Marine Fisheries Service, Office of Protected Resources). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Young, C. N. and J. K. Carlson. (2020). The biology and conservation status of the oceanic whitetip shark (*Carcharhinus longimanus*) and future directions for recovery. *Reviews in Fish Biology and Fisheries 30* (2): 293–312. DOI:10.1007/s11160-020-09601-3

Young, L. C., C. R. Kohley, E. A. VanderWerf, L. Fowlke, D. Casillas, M. Dalton, M. Knight, A. Pesque, E. M. Dittmar, and A. F. Raine. (2023). Successful translocation of Newell's Shearwaters and Hawaiian Petrels to create a new, predator free breeding colony. *Frontiers in Conservation Science 4* 1177789. DOI:10.3389/fcosc.2023.1177789

Young, M., K. Cavanaugh, T. Bell, P. Raimondi, C. A. Edwards, P. T. Drake, L. Erikson, and C. Storlazzi. (2016b). Environmental controls on spatial patterns in the long-term persistence of giant kelp in central California. *Ecological Monographs 86* (1): 45–60.

Young, N. C., Brower, A. A., Muto, M. M., Freed, J. C., Angliss, R. P., Friday, N. A., Boveng, P. L., Brost, B. M., Cameron, M. F., Crance, J. L., Dahle, S. P., Fadely, B. S., Ferguson, M. C., Goetz, K. T., London, J. M., Oleson, E. M., Ream, R. R., Richmond, E. L., Shelden, K. E. W., Sweeney, K. L., Towell, R. G., Wade, P. R., Waite, J. M., and Zerbini, A. N. (2023). *Alaska Marine Mammal Stock Assessments: 2022* (NOAA Technical Memorandum NMFS-AFSC-474). Washington, DC: U.S. Department of Commerce.

Zabin, C. J., E. M. Danner, E. P. Baumgartner, D. Spafford, K. A. Miller, and J. S. Pearse. (2013). A comparison of intertidal species richness and composiiton between Central California and Oahu, Hawaii. *Marine Ecology 34* 131–156. DOI:10.1111/maec.12007

Zagzebski, K. A., F. M. D. Gulland, M. Haulena, M. E. Lander, D. J. Greig, L. J. Gage, B. M. Hanson, P. K. Yochem, and B. S. Stewart. (2006). Twenty-five years of rehabilitation of odontocetes stranded in central and northern California, 1977 to 2002. *Aquatic Mammals 32* (3): 334–345. DOI:10.1578/am.32.3.2006.334

Zavala-Gonzalez, A. and E. Mellink. (2000). Historical exploitation of the California sea lion, *Zalophus californianus*, in Mexico. *Marine Fisheries Review 62* (1): 35–40.

Zeyl, J. N., E. P. Snelling, M. Connan, M. Basille, T. A. Clay, R. Joo, S. C. Patrick, R. A. Phillips, P. A. Pistorius, P. G. Ryan, A. Snyman, and S. Clusella-Trullas. (2022). Aquatic birds have middle ears adapted to amphibious lifestyles. *Scientific Reports 12* (1): 5251. DOI:10.1038/s41598-022-09090-3

Zydelis, R., C. Small, and G. French. (2013). The incidental catch of seabirds in gillnet fisheries: A global review. *Biological Conservation 162* 76–88.