

Endangered Species Act (ESA) Section 7(a)(2) Biological Opinion and Magnuson–Stevens Fishery Conservation and Management Act Essential Fish Habitat Response

Reissuance of National Pollutant Discharge Elimination System (NPDES) permit (#CA0107409) for the Point Loma Wastewater Treatment Plant and Ocean Outfall

NMFS Consultation Number: WCRO-2021-03010

Action Agency: U.S. Environmental Protection Agency Region 9

Affected Species and NMFS’ Determinations:


ESA-Listed Species	Status	Is Action Likely to Adversely Affect Species?	Is Action Likely to Jeopardize the Species?	Is Action Likely to Adversely Affect Critical Habitat?	Is Action Likely to Destroy or Adversely Modify Critical Habitat?
Green turtle, East Pacific DPS (<i>Chelonia mydas</i>)	Threatened	Yes	No	NA	
Leatherback turtle (<i>Dermochelys coriacea</i>)	Endangered	Yes	No	NA	
Loggerhead turtle, North Pacific Ocean DPS (<i>Caretta caretta</i>)	Endangered	Yes	No	NA	
Olive ridley turtle (<i>Lepidochelys olivacea</i>)	Threatened	Yes	No	NA	
Guadalupe fur seal (<i>Arctocephalus townsendii</i>)	Threatened	Yes	No	NA	
Blue whale (<i>Balaenoptera musculus</i>)	Endangered	Yes	No	NA	
Fin whale (<i>Balaenoptera physalus</i>)	Endangered	Yes	No	NA	

ESA-Listed Species	Status	Is Action Likely to Adversely Affect Species?	Is Action Likely to Jeopardize the Species?	Is Action Likely to Adversely Affect Critical Habitat?	Is Action Likely to Destroy or Adversely Modify Critical Habitat?
Gray whale, Western North Pacific DPS (<i>Eschrichtius robustus</i>)	Endangered	Yes	No	NA	
Humpback whale, Central American DPS (<i>Megaptera novaeangliae</i>)	Endangered	Yes	No	NA	
Humpback whale, Mexico DPS (<i>Megaptera novaeangliae</i>)	Endangered	Yes	No	NA	
White abalone (<i>Haliotis sorenseni</i>)	Endangered	Yes	No	NA	
Giant manta ray (<i>Mobula birostris</i>)	Threatened	No		NA	
Green sturgeon, Southern DPS (<i>Acipenser medirostris</i>)	Threatened	No		NA	
Oceanic whitetip shark (<i>Carcharhinus longimanus</i>)	Threatened	No		NA	
Scalloped hammerhead shark, Eastern Pacific DPS (<i>Sphyrna lewini</i>)	Endangered	No		NA	
Steelhead, Southern California DPS (<i>Oncorhynchus mykiss</i>)	Endangered	No		NA	

ESA-Listed Species	Status	Is Action Likely to Adversely Affect Species?	Is Action Likely to Jeopardize the Species?	Is Action Likely to Adversely Affect Critical Habitat?	Is Action Likely to Destroy or Adversely Modify Critical Habitat?
North Pacific right whale (<i>Eubalaena japonica</i>)	Endangered	No		NA	
Sei whale (<i>Balaenoptera borealis</i>)	Endangered	No		NA	
Sperm whale (<i>Physeter macrocephalus</i>)	Endangered	No		NA	
Black abalone (<i>Haliotis cracherodii</i>)	Endangered	No		NA	

Fishery Management Plan That Identifies EFH in the Project Area	Does Action Have an Adverse Effect on EFH?	Are EFH Conservation Recommendations Provided?
Coastal Pelagic Species	Yes	No
Highly Migratory Species	Yes	No
Pacific Coast Groundfish	Yes	No

Consultation Conducted By: National Marine Fisheries Service, West Coast Region

Issued By: 
 Chris Yates
 Assistant Regional Administrator for Protected Resources
 West Coast Region
 National Marine Fisheries Service

Date: March 4, 2022

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LIST OF ACRONYMS

BE	Biological evaluation
BOD	Biological oxygen demand
CA/OR/WA	California-Oregon-Washington
CCAP	Climate Change Action Plan
CCE	California Current Ecosystem
CEC	Contaminants of Emerging Concern
CFR	Code of Federal Register
CHL- <i>a</i>	Chlorophyll- <i>a</i>
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
City	City of San Diego
CO ₂	Carbon dioxide
CPS	Coastal pelagic species
CWA	Clean Water Act
DCP	Diphenylcresyl phosphate
DDD	Dichlorodiphenyldichloroethane
DDE	Dichlorodiphenyldichloroethylene
DDT	Dichlorodiphenyltrichloroethane
DPS	Distinct Population Segment
DQA	Data Quality Act
EDC	Endocrine-disrupting chemical
EEZ	Exclusive Economic Zone
EFH	Essential Fish Habitat
EFHA	Essential Fish Habitat Assessment
ENP	Eastern North Pacific
EPA	Environmental Protection Agency
ERL	Effects Range-Low
ERM	Effects Ranged Median
ESA	Endangered Species Act
FMP	Fishery Management Plan
FR	Federal Register
HAB	Harmful algal bloom
HAPC	Habitat area of particular concern
HMS	Highly migratory species
IPPP	Isopropylated triphenyl phosphate
ITS	Incidental Take Statement
IUCN	International Union for Conservation of Nature
LACSD	Los Angeles County Sanitation Districts
LASAN	City of Los Angeles Sanitation and Environment
MGD	Million gallons per day
MMPA	Marine Mammal Protection Act
MSA	Magnuson-Stevens Fishery Conservation and Management Act
mtDNA	Mitochondrial DNA
NA	Not Applicable
ND	Not Detected

NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOEC	No observed effect concentration
NPDES	National Pollutant Discharge Elimination System
OEHHA	Office of Environmental Health Hazard Assessment
PAH	Polycyclic aromatic hydrocarbon
PBDE	Polybrominated diphenyl ether
PBR	Potential biological removal
PCB	Polychlorinated biphenyl
PDO	Pacific Decadal Oscillation
PFMC	Pacific Fishery Management Council
POP	Persistent organic pollutant
PPCP	Pharmaceutical and personal care products
PSP	Paralytic Shellfish Poisoning
PSRG	Pacific Scientific Review Group
RFMO	Regional Fishery Management Organization
ROMS	Regional Oceanic Modeling Systems
PTMP	Plume Tracking Monitoring Plan
RPM	Reasonable and prudent measure
SAR	Stock assessment report
SBOO	South Bay Ocean Outfall
SCB	Southern California Bight
SCCOOS	Southern California Coastal Ocean Observing System
SCCWRP	Southern California Coastal Water Research Project
SDRWQCB	San Diego Regional Water Quality Control Board
SPLASH	Structure of Populations, Levels of Abundance and Status of Humpbacks
SWRCB	State Water Resources Control Board
T&Cs	Terms and conditions
T2IPPP	Tris(2-isopropylphenyl) phosphate
TBT	Tributyltin
TBOEP	Tris(2-butoxyethyl) phosphate
TCDD	Tetrachlorodibenzo-p-dioxin
TCEP	Tris(chloroethyl) phosphate
TCIPP	Tris(chloroisopropyl) phosphate
TCPP	Tris (chloropropyl)phosphate
TDCPP	Tris (1,3-dichloro-2-propyl) phosphate
TDS	Total dissolved solids (a measure of salinity)
TEP	triethyl phosphate
TIWRP	Terminal Island Water Reclamation Plant
TNBP	Tri-n-butyl phosphate
TPP or TPhP	Triphenyl phosphate
TPPO	Triphenylphosphine oxide
TSS	Total Suspended Solids
UME	Unusual mortality event
USFWS	U.S. Fish and Wildlife Service
WCR	West Coast Region
WNP	Western North Pacific

WQBEL	Water-quality based effluent limitation
WWF	World Wildlife Fund
WWTP	Wastewater treatment plant
ZID	Zone of initial dilution

1. INTRODUCTION

This Introduction section provides information relevant to the other sections of this document and is incorporated by reference into Sections 2 and 3 below.

1.1. Background

The National Marine Fisheries Service (NMFS) prepared the biological opinion (opinion) and incidental take statement (ITS) portions of this document in accordance with section 7(b) of the Endangered Species Act (ESA) of 1973 (16 U.S.C. 1531 et seq.), as amended, and implementing regulations at 50 CFR part 402.

We also completed an essential fish habitat (EFH) consultation on the proposed action, in accordance with section 305(b)(2) of the Magnuson–Stevens Fishery Conservation and Management Act (MSA) (16 U.S.C. 1801 et seq.) and implementing regulations at 50 CFR part 600.

We completed pre-dissemination review of this document using standards for utility, integrity, and objectivity in compliance with applicable guidelines issued under the Data Quality Act (DQA) (section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001, Public Law 106-554). The document will be available within 2 weeks at the NOAA Library Institutional Repository [<https://repository.library.noaa.gov/welcome>]. A complete record of this consultation is on file at the NMFS West Coast Region (WCR) Long Beach Office.

1.2. Consultation History

On May 5, 2016, NMFS's WCR received two letters transmitted via email from the City of San Diego (City) regarding the City's application to the U.S. Environmental Protection Agency (EPA) and San Diego Regional Water Quality Control Board (SDRWQCB) requesting to renew its National Pollutant Discharge Elimination System (NPDES) permit to discharge treated wastewater from the Point Loma Wastewater Treatment Plant (WWTP) to the Pacific Ocean via the Point Loma Ocean Outfall (PLOO). The City's application seeks to renew modified secondary treatment requirements for the PLOO discharge in accordance with provisions of Section 301(h) and Section 301(j)(5) of the Clean Water Act (CWA).

The first letter transmitted the City's Endangered Species (ES) Assessment, and requested NMFS's comments on it as well as asking for our concurrence with the City's determination that the continued discharge of treated wastewater from the PLOO was not likely to adversely affect endangered and threatened species or their critical habitat. In addition, the City requested updates on any new species listings, critical habitat designations, or other pertinent information not reflected in the ES Assessment or in previous correspondence noted in the letter.

The second letter transmitted the City's Essential Fish Habitat (EFH) Assessment requesting NMFS's comments on it, as well as asking for our concurrence with the City's determination that the continued discharge of treated wastewater from the PLOO would not have an adverse effect

on EFH consistent with the MSA. The letter also requested that NMFS provide any new information not reflected in the EFH Assessment.

Since receiving the City's letters in 2016, NMFS, the City, and the EPA have exchanged a number of communications. On June 16, 2016, NMFS provided pre-consultation technical assistance related to both the ESA and EFH consultations, including clarifying information on regulatory requirements associated with the role of a non-Federal representative, additional ESA-listed species to consider in the City's assessment, the listing status for black abalone under the ESA, and EFH effects determinations. On November 3, 2016, NMFS WCR received a letter from EPA appointing the City as the non-Federal representative, stating that it would prepare a biological assessment, and instructing it to conduct the ESA and EFH consultations with NMFS, referencing regulations at 50 CFR sections 402.08 (ESA) and 600.920(c) (EFH).

On May 2, 2017, NMFS notified the EPA via email that, to manage workload priorities and streamline the overall Point Loma WWTP analysis, NMFS planned to complete the ESA and EFH consultations for the City of Los Angeles's Hyperion Treatment Plant (Hyperion) before initiating ESA and EFH consultations for the PLOO project. The email also provided additional information regarding the role of a designated non-Federal representative under a formal ESA section 7 consultation. In a separate email on May 2, 2017, NMFS also included a letter we sent to the EPA for the Hyperion NPDES permit renewal that captured additional information needs that would likely pertain to the Point Loma WWTP consultations as well.

On August 4, 2017, the EPA and the SDRWQCB jointly issued a NPDES permit to the Point Loma WWTP for discharge of treated wastewater to the Pacific Ocean through the PLOO.

On September 26, 2018, NMFS WCR sent a letter to the EPA to request additional information needed to initiate the ESA and EFH consultations. NMFS WCR provided specific comments on the ESA-listed resources present in the action area and requested additional information about the outfalls, discharge plume, dilution factors, and monitoring program. In addition, based on preliminary assessment of the project and the information provided, NMFS notified the EPA that NMFS likely would not be able to concur with the City's determination that continued discharge of treated wastewater from the PLOO would not adversely affect EFH and was not likely to adversely affect ESA-listed species and their critical habitat.

On March 15, 2019, NMFS WCR received a letter from the City responding to our request for additional information (City of San Diego 2019). On June 6, 2019, representatives from NMFS, the EPA, the City, and the SDRWQCB held a technical meeting via conference call to discuss NMFS' comments, questions, and request for additional information.

On August 12, 2019, NMFS WCR sent a letter to the EPA with additional questions regarding the consultations and requesting additional information that was not adequately addressed by the EPA and the City in their written and verbal responses to NMFS's previous request.

On March 25, 2021, NMFS WCR received a letter from the City responding to NMFS's questions and August 2019 request for additional information (City of San Diego 2021c). The City proposed resolutions to the issues raised by NMFS and volunteered to complete the

monitoring studies and actions described in the proposed resolutions. The City also requested NMFS' concurrence with the City's determination that the issuance of the NPDES permit and continued discharge of treated wastewater into the Pacific Ocean via the PLOO would not adversely affect EFH and is not likely to adversely affect ESA-listed species and their critical habitat.

In response to the City's letter, NMFS and the EPA had several communications, including conference calls on July 28, August 10, and August 12, 2021, to discuss NMFS's comments regarding the City's ESA and EFH assessments and determinations. Through these conversations, NMFS and the EPA agreed that a formal consultation was needed for ESA-listed resources.

On October 6, 2021, NMFS WCR received a letter from the EPA requesting formal consultation under Section 7 of the ESA for the EPA's issuance of a NPDES permit for the Point Loma WWTP and PLOO. The EPA also requested to consult on EFH under Section 305(b)(2) of the MSA. As part of the consultation request, the EPA referenced the ES and EFH assessments previously transmitted to NMFS in 2015, along with the additional information provided in response to NMFS's subsequent requests. In the letter, the EPA acknowledged they would directly engage in the formal consultation with NMFS and would not be represented by the City, as a designated non-Federal representative cannot represent the Federal action agency in a formal consultation (50 CFR 402.08). However, the EPA mentioned that the City would continue to participate in the special monitoring studies.

We reviewed the 2017 NPDES permit issued jointly by the EPA and the SDRWQCB, the City's ES and EFH assessments, and the additional information provided by the EPA and the City in response to NMFS's requests. After evaluating all the information provided, we agreed that EPA has satisfied the requirements for initiating formal consultation under 50 CFR §402.14(c) and initiated formal consultation on October 6, 2021. Subsequently, we exchanged emails with EPA staff (11/16/21 and 12/13/21) regarding questions associated with the consultation.

In the October 6, 2021 letter requesting consultation, the EPA also requested that a draft biological opinion and EFH response be made available for their review and discussion prior to finalizing. On February 8, 2022, we transmitted a copy of a portion of the draft biological opinion to review and discuss any Reasonable and Prudent Measures and associated Terms and Conditions, as provided by 50 CFR § 402.14(g), and we mutually agreed with the EPA (on February 17, 2022) to extend the timeline of the formal consultation to accommodate their review and any subsequent deliberation. On February 17 and 28 as well as March 2, 2022, the EPA submitted to NMFS (via email) their suggested revisions to language in the Incidental Take Statement and in the Terms and Conditions. We considered and incorporated the EPA's comments in preparing this final biological opinion and EFH response on the proposed action.

1.3. Proposed Federal Action

Under the ESA, "action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by Federal agencies (see 50 CFR 402.02). Under the MSA,

“Federal action” means any action authorized, funded, or undertaken, or proposed to be authorized, funded, or undertaken by a Federal agency (see 50 CFR 600.910).]

The proposed action is EPA’s reissuance of an NPDES permit for the Point Loma Wastewater Treatment Plant and Ocean Outfall. The purpose of the NPDES permit is to authorize the discharge of modified secondary (i.e., advanced primary) treated wastewater to the Pacific Ocean from the 23,472 foot long, 310-foot deep Point Loma Ocean Outfall. The NPDES permit would be valid for a period of five years. In the following sections, we describe Point Loma’s wastewater treatment plant discharge operations, effluent limits, monitoring program, and the special studies that would be included in the NPDES permit.

We considered whether or not the proposed action would cause any other activities, and determined that it would not.

1.3.1. Point Loma Wastewater Treatment Plant Operations

The San Diego Metropolitan Sewerage System (Metro System) provides wastewater service for the City and 12 participating agencies. The E.W. Blom Point Loma Wastewater Treatment Plant (Point Loma WWTP) serves as the terminal Metro System treatment facility. The discharge of treated wastewater from the Point Loma WWTP to the Pacific Ocean via the PLOO is currently regulated by a joint permit issued by SDRWQCB and the EPA. The EPA (Regional Board Order No. R9-2017-0007) establishes modified secondary treatment requirements for the PLOO discharge in accordance with Sections 301(h) and 301(j)(5) of the CWA.

The PLOO, located 4.5 miles off the coast of Point Loma, California (see Figure 1), discharges modified secondary treated (i.e., equivalent to secondary or chemically enhanced primary effluent) waste water. Section 301(h) of the CWA allows an alternative approach to the national technology-based secondary treatment requirements, provided that the discharger can demonstrate compliance with a variety of requirements that ensure protection of ocean water quality in the absence of secondary treatment. Among these requirements is the implementation of enhanced source control, the implementation of a comprehensive ocean monitoring program, and comprehensive scientific assessments of the monitoring data to demonstrate: (1) compliance with water quality, sediment, and benthic protection standards; and (2) maintenance of a balanced indigenous population of fish and wildlife in receiving waters. The EPA has determined that the City demonstrated compliance with all state and Federal receiving water standards and all CWA Section 301(h) and 301(j)(5) requirements for the protection of the ocean environment. The City documented in their multi-volume permit application that the current PLOO discharge continues to comply with:

- Receiving water standards applicable to all ocean dischargers, and
- All applicable criteria for issuance of 301(h) and 301(j)(5) modified requirements for Biological oxygen demand (BOD) and Total Suspended Solids (TSS), including maintaining a balanced indigenous population of fish and wildlife in receiving waters.

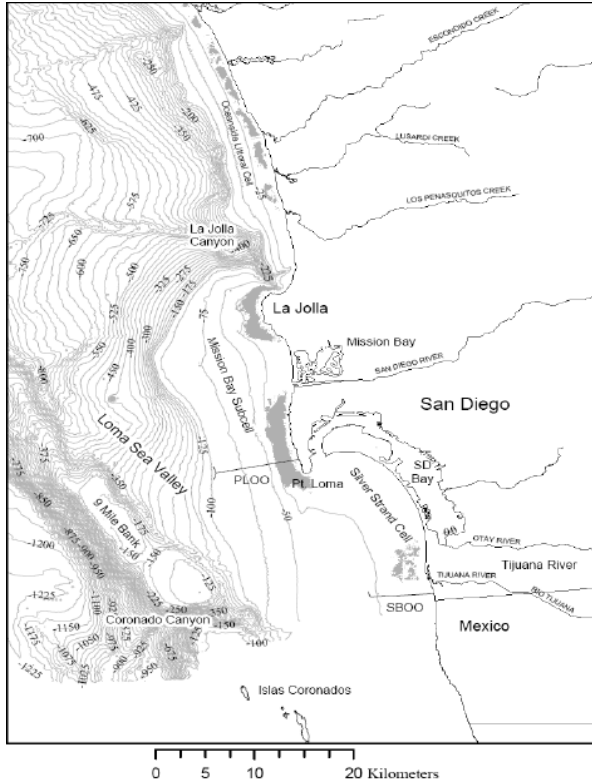


Figure 1. Location of the Point Loma Ocean Outfall (PLOO).

The PLOO discharges through diffuser ports ranging in depths from 306 ft (93 m) to 310 ft (98 m). The PLOO discharges approximately 140 million gallons per day (MGD) of treated wastewater, generated by more than 2.2 million residents and industries (with source controls) in a 450 square mile (mi²) (1,165 square kilometers (km²)) area. The Point Loma Wastewater Treatment Plant has an overall capacity of 240 MGD. Mathematical models of outfall operation indicate a median (50th percentile) initial dilution of 338:1 at a discharge flow of 240 MGD (the maximum design flow). The minimum initial dilution value (the initial dilution as determined assuming zero ocean currents and using the worst case density conditions from over 13,000 density data profiles) is computed at 204:1 which is 204 parts seawater per part wastewater.

The NPDES permit plans to target reductions in TSS mass emissions over the next 20 years. Table 1 illustrates that the program goal is to cap TSS mass emissions from the PLOO at 9,942 metric tons per year by year 2028 and beyond.

Table 1. NPDES permitted PLOO TSS mass emissions.

Year	TSS Mass Emission Rate Limit (metric tons per year)
2015 through 2025	12,000
2026 through 2027	11,500
2028 forward	9,942

In addition, the City is committing to implementing a 20-year comprehensive water and wastewater facilities plan called Pure Water San Diego (City of San Diego 2021c). The Pure Water San Diego program has a goal of reducing future Point Loma WWTP discharge flows by producing 30 MGD of potable reuse water by 2025, and a cumulative 83 MGD of potable reuse water by 2036.

1.3.2. Permitted Effluent Limits

The proposed permitted effluent limits and wastewater discharge requirements for the Point Loma WWTP and the Outfall are described in the EPA NPDES permit (EPA 2017a) unless otherwise noted. These effluent limitations and discharge specifications apply to an extensive list of constituents or parameters that represent markers of potential harm to marine life and human health, as well as to the overall impact on the local marine environment. The NPDES permit also includes monitoring of pollutants with concentration and mass-based effluent limits and monitoring of pollutants with performance goals. Some performance goals are not considered enforceable effluent limits, but are instead based on the last five years of actual performance data for the Point Loma WWTP and used as an indication of treatment efficiency. Permitted effluent limits, performance goals, and mass emission benchmarks may be measured over varying time scales, such as average weekly, monthly, six-month median, or annual values, as well as daily or instantaneous minimum and maximum values. The permitted effluent limits, performance goals, and mass emission benchmarks are largely derived from or reflect the objectives that are laid out in the Water Quality Control Plan, Ocean Waters of California, California Ocean Plan (Ocean Plan), along with site-specific considerations and the performance of Point Loma's discharge under previous NPDES permits.

The NPDES permit establishes effluent limits for the following nine pollutants/parameters: TSS, biological oxygen demand (BOD), chlorine, oil and grease, settleable solids, chronic toxicity, pH, turbidity, and aldrin. Additional pollutants are monitored in the effluent (e.g., metals such as arsenic, mercury, lead, copper, chromium, and cadmium; ammonia; endosulfan; hexachlorocyclohexane; phthalate; ethylbenzene; toluene; benzene; polycyclic aromatic hydrocarbons (PAHs); polychlorinated biphenyls (PCBs); and vinyl chloride); however, the permit states that these additional pollutants do not have water quality-based effluent limitations (WQBELs). Instead, the monitoring of these additional pollutants is used only for informational purposes and compliance do not apply.

Minimum dilution ratios are used to calculate the WQBELs or permitted levels for pollutants in the discharge. These dilution ratios assume that some minimum level of dilution (initial dilution) occurs immediately upon discharge and buffers the receiving waters outside of this zone from exposure to the effluent at full concentration. The WQBELs reflect the permitted levels of pollutants in the receiving waters immediately surrounding the discharge, prior to undergoing this minimum level of initial dilution. For the PLOO, the minimum dilution ratio used to calculate effluent limitations is 204:1 (204 parts seawater to one part effluent).

1.3.3. Monitoring Program

The EPA NPDES permit prescribes a detailed monitoring program to demonstrate compliance with the effluent limitations, discharge specifications, and other requirements in the permit (detailed in Appendix E of the permit). The core monitoring area off Point Loma extends from stations along the shore seaward to a depth of about 116m, encompassing an area of approximately 184km² (Figure 2). Under the permit, City of San Diego Point Loma is required to collect and analyze samples of influent, effluent, and receiving water, conduct whole effluent toxicity testing, and participate in regional monitoring to assess the effects of discharge on the receiving waters.



Figure 2. Monitoring stations sampled around the Point Loma Ocean Outfall as part of the City of San Diego's Ocean Monitoring Program.

Point Loma's WWTP monitoring program consists of three interconnected components:

- (1) Core monitoring;
- (2) Bight Regional Monitoring; and
- (3) Special Studies

1.3.3.1 Core Monitoring

The core monitoring provides for specific sampling locations where designated constituents are measured and monitored over time. It is directed toward assessing compliance with Federal standards established by EPA and state-wide standards established in the California Ocean Plan. Moreover, the core monitoring is critical to assessing long-term temporal changes in local marine

environmental conditions and to determining whether any such changes may be due to either anthropogenic or natural influences. The core monitoring program consists of the following elements: influent and effluent monitoring stations; receiving water monitoring stations (inshore and offshore); plume monitoring, benthic and sediment chemistry sampling; and fish and invertebrate monitoring.

The permit requires monitoring of influent and effluent at varying intervals (daily, weekly, monthly) depending on the constituent being monitored. In 2008, Point Loma WWTP began partial chlorination of the effluent, which made using bacteria as a plume tracer ineffective. As a replacement plume tracer, receiving water monitoring for ammonia was added to Order No. R9-2009-0001. However, monitoring for ammonia produced limited useful data (see results in City of San Diego (2016) as an example) because ammonia rapidly converts to nitrate in the receiving water near the PLOO. Given this, receiving water monitoring for ammonia has been removed in the Permit.

Monitoring of receiving water is conducted in the waste field in the vicinity of the zone of initial dilution (ZID) boundary, at reference stations, and at areas beyond the ZID where discharge impacts might be reasonably expected. Locations include both the shoreline and offshore areas. Shoreline water quality monitoring is conducted once per week to monitor for temperature as well as total and fecal coliforms and enterococcus. Offshore monitoring is conducted every quarter and parameters monitored include temperature, pH, salinity, dissolved oxygen (DO), light transmittance, chlorophyll a, and enterococcus. Kelp stations in the offshore environment are also monitored on a weekly basis for temperature, pH, salinity, DO, light transmittance, chlorophyll a, total and fecal coliforms and enterococcus.

In the Core Monitoring area, the plume is also monitored. Under the Plume Tracking Monitoring Plan (PTMP), monitoring is conducted to map the dispersion and fate of the wastewater plume discharged from the PLOO. To implement this plan, the following are required:

- Installation and operation by the discharger of a permanent, real-time oceanographic mooring system located near the terminal Y-shaped diffuser structure of the PLOO. The mooring system is designed to measure, at minimum, direction and velocity of subsurface currents and ocean stratification.
- Development of a work plan or pilot study (special study) for implementation of the PLOO real-time mooring system, including data acquisition and processing.
- Networking the PLOO system to be compatible with a similar system being deployed by the Discharger near the South Bay Ocean Outfall discharge site, as well as a third system operated by the University of California San Diego, Scripps Institution of Oceanography in the coastal waters off the City of Del Mar.
- Development of a work plan or pilot study (special study) for utilizing advanced oceanographic sampling technologies such as an autonomous underwater vehicle (AUV) or remotely operated towed vehicle (ROTV) in conjunction with the PLOO real-time mooring system to enhance collection of water quality data in real-time and provide higher resolution maps of plume location and movement.

Benthic communities, sediment toxicity and sediment chemistry samplings are conducted twice a year (winter and summer) at 12 primary and 10 secondary offshore benthic stations where the

secondary stations may be exchanged to monitor for the Bight Regional Monitoring Program. These samplings are conducted because benthic organisms are strongly affected by sediment contaminant exposure. Benthic organisms often live in continual direct contact with sediment/pore water, and many species ingest significant quantities of sediment as a source of nutrition. Since the benthos are dependent on their surroundings, they serve as biological indicators that reflect the overall conditions of the aquatic environment.

Fish monitoring data is also required because health risks of individuals that consume fish need to be assessed. Many pollutants discharged into receiving waters have the potential to bioaccumulate and persist in tissues of aquatic organisms, including marine fishes. Chemical pollutants that bioaccumulate tend to magnify in concentration as they pass through the aquatic food chain. Therefore, fish are assessed for trends of contaminants levels in the receiving waters over time. In addition, aquatic benthic invertebrates are monitored because they are excellent indicators of ecosystem health. They are ubiquitous, abundant, diverse, and typically sedentary. The growth, survival, and reproduction of many species of aquatic invertebrates are all sensitive to changes in environmental health, making analysis of assemblage structure a good ecosystem monitoring tool.

Single community trawls for fish and invertebrates are conducted semi-annually in the winter (e.g., January) and summer (e.g., July) at six trawl stations. These stations represent two areas near the Discharge Point, two areas up coast of the Discharge Point, and two areas down coast of the Discharge Point. Chemical analyses of fish tissues (e.g., metals, PCBs, Dichlorodiphenyltrichloroethane (DDTs), endosulfan, fluorine etc.) are performed annually on target species (i.e., flatfish including Pacific sanddab (*Citharichthys sordidus*), longfin sanddab (*Citharichthys xanthostigma*), bigmouth sole (*Hippoglossina stomata*), and hornyhead turbot (*Pleuronichthys verticalis*)) collected at or near the trawl stations. If sufficient numbers of trawl zone target species cannot be, or are unlikely to be, captured by trawling, fish for tissue analysis from these areas may be collected using alternative methods (e.g., hook and line, baited lines). Rig fishing is also conducted annually and muscle tissues of fish collected in each of the two rig fishing zones are analyzed for uptake of pollutants. The fish are collected by hook and line or by setting baited lines within zones surrounding rig fishing stations. The species targeted for muscle tissue analysis in the rig fishing stations are representative of those caught by recreational and/or commercial fishery activities in the region. The primary species targeted for muscle tissue analysis are rockfish including the vermilion rockfish (*Sebastes miniatus*) and the copper rockfish (*Sebastes caurinus*).

1.3.3.2 Bight Regional Monitoring

Regional monitoring provides information necessary to make assessments over large areas and serves to evaluate cumulative effects of anthropogenic compounds. Regional monitoring data also assists in the interpretation of core monitoring studies by providing a more accurate and complete characterization of reference conditions and natural variability. Regional monitoring also leads to methods standardization and improved quality control through inter-calibration exercise. The coalitions implementing regional monitoring enable sharing of technical resources, trained personnel, and associated costs. Focusing these resources on regional issues and developing a broader understanding of pollutants effects on ocean waters enables the

development of more rapid and effective response strategies. Under the permit, the City of San Diego Point Loma is required to participate in two regional monitoring activities conducted in the Southern California Bight (SCB): the SCB Regional Monitoring Program and the San Diego Kelp Forest Ecosystem Monitoring Project. We describe these programs in more detail below.

The SCB Regional Monitoring Program is part of a collaborative effort to provide a large-scale, integrated assessment of the SCB. Program studies include water quality, benthic infauna, sediment chemistry, fish communities, fish predator risk, and ocean acidification. The Southern California Coastal Water Research Project (SCCWRP) coordinates the Bight Regional Monitoring Program. The Program occurs on five-year cycles, with the most recent program occurring in 2018. The 2018 program covered five topics: sediment quality, microbiology, ocean acidification, harmful algal blooms, and trash. The next Bight Regional Monitoring program will occur in 2023.

The City of San Diego Point Loma also participates with other Southern California ocean dischargers in an ongoing regional survey of coastal kelp beds in the Southern California Bight. The intent of these surveys is to provide an indication of the health of these kelp beds. Kelp beds are monitored by means of vertical aerial infrared photography to determine the maximum areal extent of the canopies of coastal kelp beds each year. Surveys are conducted as close as possible to when kelp bed canopies are at their greatest extent during the year. Each year, the maximum kelp bed canopy areas are compared to those observed in previous years. Any significant losses that persist for more than one year are then investigated by divers to document benthic and understory conditions.

In addition to the two programs described above, the City also conducts annual region-wide benthic surveys at 40 randomly selected sites off the coast of San Diego. This required monitoring program is associated with the South Bay Ocean Outfall (SBOO) discharge and has been conducted since 1995. This region-wide monitoring is designed to augment the 5-year SCB surveys and span both the SBOO and PLOO regions.

1.3.3.3 Ammonia and Nitrogen Monitoring

Recent studies show that wastewater discharges and upwelling are the two most important contributors of nutrients in the SCB (Howard et al. 2014). Considering this finding, the EPA NPDES permit for Point Loma includes monitoring of ammonia and/or nitrogen from effluent, sediment, and biosolid samples. Ammonia nitrogen (as total nitrogen) is monitored in effluent via the PLOO on a weekly basis, but as stated in section 1.3.1 above, levels of nitrogen are used only for informational purposes and are not compliance determinations or water quality-based effluent limitations (WQBELs). Total nitrogen is also monitored twice a year from sediments collected at offshore benthic monitoring stations. Moreover, both nitrogen and ammonia are monitored within biosolid/sewage sludge samples collected twice a year and are reported annually; however, no biosolids are permitted to enter wetlands or other water of the U.S.

The City of San Diego (2021b) plans to implement a special one-year supplemental receiving water study in which nitrogen, phosphorus and chlorophyll-*a* data are collected monthly at

selected offshore stations near the PLOO in conjunction with regular NPDES monitoring of stations near and surrounding the PLOO. Any information generated by the special studies would be made available to the scientific community involved in studying HABs and nutrient inputs.

1.3.3.4 Whole Effluent Toxicity (WET) Testing

The whole effluent toxicity (WET) testing refers to the overall aggregate toxic effect on aquatic organisms from all pollutants contained in a facility's wastewater effluent. WET tests evaluate: (1) the aggregate toxic effects of all chemicals in the effluent including additive, synergistic, or antagonistic effects; (2) the effects of unmeasured chemicals in the effluent; and (3) variability in bioavailability of the chemicals in the effluent. Toxicity is determined relative to a control or reference sample using a given test method, and two types of tests are conducted: acute toxicity testing, conducted semiannually, measures mortality over a short period of time; chronic testing, conducted monthly, measures mortality, reproduction, and/or growth effects over a short or longer exposure period of time.

WET testing employs the use of standardized, surrogate freshwater or marine (depending upon the mixture of effluent and receiving water) plants, invertebrates, and vertebrates. Different test species can exhibit different sensitivities to toxicants. However, the EPA considers standard test species to represent the sensitive range of all ecosystems analyzed. Standard test species used for chronic toxicity tests at Point Loma WWTP are topsmelt (*Atherinops affinis*), red abalone (*Haliotis rufescens*), purple sea urchin (*Strongylocentrotus purpuratus*), sand dollar (*Dendraster excentricus*), and giant kelp (*Macrocystis pyrifera*).

The EPA has noted that the results of whole effluent tests in the past using five marine/estuarine short-term chronic test methods indicate that no species or test method is always the most sensitive. Therefore, the proposed permit requires re-screening of the standard test species every two years to ensure the most sensitive test species are used in evaluating the toxicity of the effluent. Results for both acute and chronic toxicity tests are reported as either “Pass” or “Fail” following the Test of Significant Toxicity statistical approach described in the NPDES Test of Significant Toxicity Implementation Document (EPA 2010).

1.3.4. Special Studies

Special studies are directed monitoring efforts designed in response to specific management or research questions identified through either core or regional monitoring programs. They are used to help understand core or regional monitoring results, where a specific environmental process is not well understood, or to address unique issues of local importance.

The EPA NPDES permit requires Point Loma WWTP to prepare and submit a climate change action plan (CCAP) within three years of the effective date of this permit. Changing climate conditions may fundamentally alter the way publicly-owned treatment plants are designed and operated. Climate change research indicates the overarching driver of change is increased atmospheric carbon dioxide (CO₂) from human activity. The increased CO₂ emissions trigger changes in climatic patterns, which increase the intensity of sea level rise and coastal storm

surges (Δ Sea Level), lead to more erratic rainfall and local weather patterns (Δ Weather Patterns), trigger a gradual warming of freshwater and ocean temperatures (Δ Water Temperature), and trigger changes to ocean water chemistry (Δ Water pH). The CCAP shall identify projected regional impacts on Metro System facilities and operations due to climate change if current trends continue. The CCAP shall also identify steps being taken or planned to address greenhouse gas emissions attributable to wastewater treatment plants, solids handling, and effluent discharge processes. The CCAP shall also identify steps being taken or planned to address:

- Flooding and sea level rise risks;
- Volatile rain period impacts (both dry and wet weather);
- Challenges in accommodating high and low wastewater flows;
- Impacts on process design parameters due to higher BOD, ammonium, and TSS influent concentrations;
- Impacts on wastewater treatment operations and quality; the potential need to adjust NPDES permit conditions and the Metro System pollution control program;
- The financing needed to pay for planned actions;
- Schedules to update the CCAP as more information on climate change and its effect become more available; and
- Any other factors as appropriate.

Another study is planned to develop advanced oceanographic sampling technologies as part of the Plume Tracking Monitoring Program to enhance collection of water quality data in real-time and provide higher resolution maps of plume location and movement.

Lastly, the City of San Diego (2021c) acknowledges the need for additional information to address uncertainty that exists regarding the sources, concentrations, and effects of contaminants of concern (CECs) and persistent organic pollutants (POPs) on coastal waters. After summarizing important efforts being implemented by the State Water Resources Control Board (SWRCB) and SCCWRP, the City proposes to participate in the SCCWRP effort by providing analysis of the PLOO effluent for a target number of CECs (e.g., nitrogen, phosphorus and chlorophyll-*a*). The proposed resolution is designed to provide data on the presence of the target compounds in the PLOO effluent, as well as adding to the scientific community's research efforts. To accomplish this, the City would perform testing of the Point Loma PLOO discharge for target CECs/POPs prior to and after the startup of Pure Water phase 1. The target compounds would be determined in consultation with EPA, NMFS, SCCWRP and others to ensure coordination and to add value to their research efforts. This testing would also be coordinated with the CEC/POP survey being done to fulfill a requirement in the recently adopted permit for discharge of recycled potable reuse water to the potable reservoir system. These studies would provide information on the presence of CEC/POPs in the PLOO discharge, as well as any reductions that occur as a result of Pure Water upstream offloading.

2. ENDANGERED SPECIES ACT: BIOLOGICAL OPINION AND INCIDENTAL TAKE STATEMENT

The ESA establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat upon which they depend. As required by section 7(a)(2) of the ESA, each Federal agency must ensure that its actions are not likely to jeopardize the continued existence of endangered or threatened species or to adversely modify or destroy their designated critical habitat. Per the requirements of the ESA, Federal action agencies consult with NMFS, and section 7(b)(3) requires that, at the conclusion of consultation, NMFS provide an opinion stating how the agency's actions would affect listed species and their critical habitats. If incidental take is reasonably certain to occur, section 7(b)(4) requires NMFS to provide an ITS that specifies the impact of any incidental taking and includes reasonable and prudent measures (RPMs) and terms and conditions to minimize such impacts.

We also determined the proposed action is not likely to adversely affect (NLAA) the following ESA-listed species: North Pacific right whales, sei whales, sperm whales, giant manta rays, Southern Distinct Population Segment (DPS) green sturgeon, oceanic whitetip sharks, Eastern Pacific DPS scalloped hammerhead sharks, Southern California steelhead, and black abalone. These determinations are discussed in the in the "Not Likely to Adversely Affect" Determinations section (Section 2.12). This opinion does not consider effects on critical habitat because none is designated in the action area.

2.1. Analytical Approach

This biological opinion includes a jeopardy analysis. The jeopardy analysis relies upon the regulatory definition of "jeopardize the continued existence of" a listed species, which is "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). Therefore, the jeopardy analysis considers both survival and recovery of the species.

This biological opinion also relies on the regulatory definition of "destruction or adverse modification," which "means a direct or indirect alteration that appreciably diminishes the value of critical habitat as a whole for the conservation of a listed species" (50 CFR 402.02). However, this biological opinion does not include an adverse modification analysis because the the action area does not overlap with any designated critical habitat.

The ESA Section 7 implementing regulations define effects of the action using the term "consequences" (50 CFR 402.02). As explained in the preamble to the final rule revising the definition and adding this term (84 FR 44976, 44977; August 27, 2019), that revision does not change the scope of our analysis, and in this opinion we use the terms "effects" and "consequences" interchangeably.

We use the following approach to determine whether a proposed action is likely to jeopardize listed species:

- Evaluate the rangewide status of the species expected to be adversely affected by the proposed action.
- Evaluate the environmental baseline of the species.
- Evaluate the effects of the proposed action on species and their habitat using an exposure–response approach.
- Evaluate cumulative effects.
- In the integration and synthesis, add the effects of the action and cumulative effects on the environmental baseline, and, in light of the status of the species, analyze whether the proposed action is likely to directly or indirectly reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species.
- If necessary, suggest a reasonable and prudent alternative to the proposed action.

2.2. Rangewide Status of the Species

This opinion examines the status of each species that is likely to be adversely affected by the proposed action. The status is determined by the level of extinction risk that the listed species face, based on parameters considered in documents such as recovery plans, status reviews, and listing decisions. This informs the description of the species’ likelihood of both survival and recovery. The species status section also helps to inform the description of the species’ “reproduction, numbers, or distribution” for the jeopardy analysis. This opinion does not consider effects on critical habitat because none is designated in the action area.

One factor affecting the range-wide status of ESA-listed species and aquatic habitat at large is climate change. Climate change has received considerable attention in recent years, with growing concerns about global warming and the recognition of natural climatic oscillations on varying time scales, such as long-term shifts like the Pacific Decadal Oscillation (PDO) or short-term shifts, like El Niño or La Niña. Evidence suggests that the productivity in the North Pacific (Lowry et al. 1989; Quinn and Niebauer 1995) and other oceans could be affected by changes in the environment. Important ecological functions such as migration, feeding, and breeding locations may be influenced by factors such as ocean currents and water temperature. Any changes in these factors could render currently used habitat areas unsuitable and require displaced individuals to use previously unutilized or non-existing habitats. Changes to climate and oceanographic processes may also lead to decreased productivity or different patterns of prey distribution and availability. Such changes could affect individuals that are dependent on those affected prey.

Multiple studies have detected changes in the abundance, quality, and distribution of whale prey species in association with climate shifts, particularly with ocean warming. The nature and extent of effects have varied across study areas and species; however, in many cases, ocean warming has led to negative effects on these prey species. For instance, in the California Current Ecosystem (CCE), an anomalous warming of the upper ocean and weak upwelling event occurred from 2013–2016, often referred to as the “blob” or the “warm blob.” During this period, sharp decreases in euphausiid biomass were observed, as evidenced by declines in both abundance and body length (Peterson et al. 2017; Harvey et al. 2017). Brodeur et al. (2019)

Brodeur et al. (2019) compared samples collected in the Northern California Current region during years of cool (2011, 2012), warm (2000, 2002), and intermediate (2015, 2016) conditions and found that body condition of northern anchovy, Pacific herring, and Pacific sardine were better in cool years compared to warm years, and significantly so for anchovy and herring. During the anomalous warm blob event, sardine spawned earlier and appeared farther north within the Northern California Current than in previous years (Auth et al. 2018).

Shifts in prey abundance and distributions may lead to corresponding shifts in marine mammal distributions (King et al. 2011). In Monterey Bay, California, such a response was reported for blue, fin, and humpback whales, the densities of which all declined with El Niño-associated declines in euphausiids (Benson et al. 2002). More recently, Santora et al. (2020) outlined how the 2014-2016 marine heat wave in the northeast Pacific Ocean changed humpback whale prey distribution and abundance resulting in a habitat compression for the species with a coastward shift in distribution. By shifting closer to the coast, humpback whales were more likely to encounter coastal fisheries, which has resulted in an increase in humpback whale entanglements in recent years. In another example, there is some evidence from Pacific equatorial waters that sperm whale feeding success and, in turn, calf production rates are negatively affected by increases in sea surface temperature (Smith and Whitehead 1993; Whitehead 1997). Any changes in these factors could render currently used habitat areas unsuitable. Changes to climate and oceanographic processes may also lead to decreased prey productivity and different patterns of prey distribution and availability. Different species of marine mammals will likely react to these changes differently. For example, range size, location, and whether or not specific range areas are used for different life history activities (e.g. feeding, breeding) are likely to affect how each species responds to climate change (Learmonth et al. 2006).

Based upon available information, it is likely that sea turtles are being affected by climate change. Sea turtle species are likely to be affected by rising temperatures that may affect nesting success and skew sex ratios, as some rookeries are already showing a strong female bias as warmer temperatures in the nest chamber leads to more female hatchlings (Chan and Liew 1995; Kaska et al. 2006). Rising sea surface temperatures and sea levels may affect available nesting beach areas as well as ocean productivity. Based on climate change modeling efforts in the eastern tropical Pacific Ocean, for example, Saba et al. (2012) predicted that the Playa Grande (Costa Rica) sea turtle nesting populations would decline 7% per decade over the next 100 years. Changes in beach conditions are expected to be the primary driver of the decline, with hatchling success and emergence rates declining by 50-60% over the next 100 years in that area (Tomillo et al. 2012). Sea turtles are known to travel within specific isotherms and these could be affected by climate change and cause changes in their bioenergetics, thermoregulation, prey availability, and foraging success during the oceanic phase of their migration (Robinson et al. 2008; Saba et al. 2012). While the understanding of how climate change may impact sea turtles is building, there is still uncertainty and limitations surrounding the ability to make precise predictions about or quantify the threat of future effects of climate change on sea turtle populations (Hawkes et al. 2009).

Climate change impacts that could affect abalone and its habitat include ocean acidification and elevated water temperatures. Ocean acidification could result in water quality conditions that reduce larval survival and shell growth and increase shell abnormalities (Crim et al. 2011).

Studies show that effects of ocean acidification are highly species-specific due to differences between species in physiology, adaptability, and exposure to natural variation in ocean pH. There is a large degree of variability and uncertainty in climate change predictions, the timeframe over which changes may occur, and how the species and their habitat may respond. For example, abalone may be able to adapt to ocean acidification because they already experience natural variability in ocean pH, including low pH levels (Feely et al. 2004, 2008, 2009; Hauri et al. 2009). Increasing ocean water temperatures may occur due to global warming, and short-term and longer-term oceanographic conditions (e.g., ENSO or PDO events) may have varying effects on abalone. For example, warmer water temperatures may reduce food availability and quality by reducing macroalgal growth (Hobday et al. 2001; Tegner et al. 2001), and increase susceptibility to withering syndrome (Ben-Horin et al. 2013). At the same time, warmer water temperatures may benefit larval survival if temperatures move toward the optimum temperatures (Leighton 1972). Studies are underway to evaluate the effects of ocean acidification and increasing water temperatures on abalone, and to assess how other factors (e.g., presence of the disease vectors) may affect these interactions.

We consider the ongoing implications of climate change as part of the status of ESA-listed species. Where necessary or appropriate, we consider whether the effects of the proposed action on ESA-listed species could potentially influence the resiliency or adaptability of those species to deal with the climate change effects that we believe are likely over the foreseeable future.

2.2.1. Marine Mammals

2.2.1.1 Blue Whale

Blue whales were listed as endangered worldwide under the precursor to the ESA, the Endangered Species Conservation Act of 1969, and remained on the list of threatened and endangered species after the passage of the ESA in 1973 (35 FR 8491). Currently there is no designated critical habitat for blue whales. Blue whales make seasonal migrations between feeding and breeding locations, with their distribution often linked to patterns of aggregated prey. Like other baleen whales, the seasonal and inter-annual distribution of blue whales is strongly associated with both static and dynamic oceanographic features such as upwelling zones that aggregate krill (*Euphausia pacifica*; see Croll et al. 2005 for a review).

Blue whales are currently separated into three subspecies in the North Pacific, North Atlantic, and Southern Hemisphere. Their population structure has been studied through photo identification, acoustic, and genetic analyses showing both geographic isolation and overlap of some subpopulations. The Marine Mammal Protection Act (MMPA) identifies geographic stocks of marine mammals, which are groups of marine mammals of the same species or smaller taxa in a common spatial arrangement that interbreed when mature. The MMPA requires the monitoring and management of marine mammals on a stock-by-stock basis rather than entire species, populations, or distinct population segments. For this opinion, we will analyze effects at the ESA-listed global population level, but will rely heavily upon information from the near annual stock assessment reports (SARs) for the Eastern North Pacific (ENP) stock of blue whales that is identified as one of the nine blue whale management units in the NMFS 2020 blue whale

recovery plan, as well as the most recent scientific information available regarding the abundance of blue whales along the U.S. west coast.

The blue whales most likely to be observed within the proposed action area are identified as part of the ENP stock. Tagging and photo identification studies have shown that the feeding population off southern California also migrates as far south as the equator to feed in the eastern tropical Pacific (Mate et al. 1999). These findings have been confirmed through vocal analyses, where the same call type representing the ENP stock have been recorded in the Gulf of Alaska south to the Costa Rica Dome (Stafford et al. 2001; Calambokidis et al. 2009). Recently, Irvine et al. (2014) documented the multi-year satellite track of a blue whale first tagged off California. This animal had very strong site fidelity to particular feeding areas in southern and northern California. In fact, this animal made excursions from one prey field to another, suggesting it was foraging on local increases in prey density and further demonstrating the importance of feeding areas off California to the ENP blue whale stock.

Population Status and Trends: Though still depleted compared to historical abundance, blue whale abundance appears to be increasing in most if not all regions during the past several decades, although the data for most areas are sparse and uncertain (Calambokidis and Barlow 2004; Branch et al. 2007). Although there is insufficient data available to assess the present status in most parts of the North Pacific, the feeding stock of blue whales off the U.S. west coast has been estimated by line-transect and mark-recapture methods. Generally, the highest abundance estimates from line-transect surveys occurred in the mid-1990s, when ocean conditions were colder than present-day (Carretta et al. 2021a). Since that time, line-transect abundance estimates within the California Current have declined, while estimates from mark-recapture studies have increased or remained stable (Carretta et al. 2021a). Evidence for a northward shift in blue whale distribution includes increasing numbers of blue whales found in Oregon and Washington waters during 1996-2014 line-transect surveys (Barlow 2016) and satellite tracks of blue whales in Gulf of Alaska and Canadian waters between 1994 and 2007 (Bailey et al. 2009). Calambokidis and Barlow (2020) estimated blue whale abundance for the U.S. west coast at 1,898 whales, based on updated photographic ID data through 2018 using mark-recapture methods. Becker et al. (2020) estimated blue whale abundance at 670 whales, using habitat-based species distribution models from line-transect data collected from 1991 to 2018. The mark-recapture estimate (1,898) is considered the best estimate of abundance for 2018 due to its higher precision and because estimates based on line-transect data reflect only animal densities within the study area at the time surveys are conducted (Carretta et al. 2021a). To put this in context, NMFS (2020) established recovery criteria for the ENP blue whale management unit as at least 2,000 animals for downlisting and 2,500 for de-listing. Population trends must also be stable or increasing for downlisting.¹

Threats: Blue whales experienced intensive whaling throughout the 20th century, and the threat of directed hunting remains. Other threats that may be affecting blue whales with at least a potential for population-level consequences or are significant enough to contribute to the species' extinction risk include ship strikes, entanglement in marine debris and fishing gear, anthropogenic noise, and loss of prey base due to climate and ecosystem change (NMFS 2020a).

¹ Recovery of the globally listed blue whale population is contingent upon all nine management units meeting the relevant criteria described in Recovery Plan for the Blue Whale (NMFS 2020).

It is difficult to estimate the numbers of blue whales possibly killed and injured by fishing gear, because large whales that become entangled in fishing gear may often die later and drift far enough to not strand on land after such incidents. Vessel strikes are also a threat to all large whales, including blue whales, although reported vessel strikes are considered a minimum accounting of the total. The threat to blue whales due to underwater noise, pollutants, marine debris, and habitat degradation, are difficult to quantify. However, there is a growing concern that the increasing levels of anthropogenic noise in the ocean may be a habitat concern for whales, particularly for whales that use low frequency sound to communicate, such as baleen whales.

For the ENP stock, the observed annual incidental mortality and injury rate (0.88/year) from vessel strikes is less than the calculated potential biological removal (PBR: 4.1) for this stock. This rate does not include unidentified large whales struck by vessels, some of which may have been blue whales, nor does it include undetected and unreported vessel strikes of blue whales (Carretta et al. 2021a). Off the U.S. west coast, blue whale distribution overlaps significantly with the transit routes of large commercial vessels, including cruise ships, large tug and barge transport vessels, and oil tankers in the proposed action area. Vessel strike mortality was estimated to be 18 blue whales per year in the U.S. West Coast Exclusive Economic Zone (EEZ), although this estimate includes only the period of July – November when whales are most likely to be present in the U.S. West Coast EEZ (Rockwood et al. 2017). Blue whales have occasionally been documented entangled in pot/trap fisheries and other unidentified fishery gear on the U.S. west coast in recent years. The annual entanglement rate of blue whales (reported and assigned) during 2015-2019 is 1.39 whales annually (Carretta et al. 2021a). Observed and assigned levels of serious injury and mortality due to commercial fisheries (1.39) exceed 10% of the stock's PBR (4.1), thus, commercial fishery take level are not approaching zero mortality and serious injury rate. Considering the effects of estimated vessel strikes, plus the effects of entanglements, human impacts may be exceeding the calculated PBR of 4.1 for this stock. While it is unknown if these same threats are occurring at similar levels throughout the global population of blue whales, the current levels of human impacts in the ENP are occurring at levels that may delay recovery of this stock of blue whales. Such a circumstance, especially if happening in concert with similar or larger impacts throughout their range, means that recovery of the entire species could be delayed altogether. The blue whale recovery plan (NMFS 2020a) describes recommended actions to determine the level of threat fishery entanglements, vessel strikes, and other potential threats pose to the likelihood of survival and recovery of the species.

2.2.1.2 Fin Whale

Fin whales were listed as endangered worldwide under the precursor to the ESA, the Endangered Species Conservation Act of 1969, and remained on the list of threatened and endangered species after the passage of the ESA in 1973 (35 FR 8491). Currently there is no designated critical habitat for fin whales. Fin whales feed on planktonic crustaceans, including *Thysanoessa* sp. (euphausiids) and *Calanus* sp. (copepods), and schooling fish, including herring, capelin, and mackerel (Aguilar 2009). Association with the continental slope is common, perhaps due to abundance of prey (Schorr et al. 2010). However, fin whales aggregate to areas with large amounts of prey regardless of water depth. For example, fin whales can feed in more shallow

waters during the day (less than 330 feet), and feed in deeper waters at night (can be greater than 1,320 feet) (EPA 2017b).

Fin whales are distributed widely in the world's oceans and occur in both the Northern and Southern Hemispheres. In the northern hemisphere, they migrate from high Arctic feeding areas to low latitude breeding and calving areas. In the Atlantic Ocean, fin whales have an extensive distribution from the Gulf of Mexico and Mediterranean Sea northward to the arctic. The North Pacific population summers from the Chukchi Sea to California, and winters from California southward. Fin whales have also been observed in the waters around Hawaii. Fin whales can occur year-round off California, Oregon, and Washington (Carretta et al. 2021a). Recent information suggests that fin whales are present year-round in southern California waters, as evidenced by individually-identified whales being photographed in all four seasons (Falcone and Schorr 2013). For this opinion, we will analyze effects at the ESA-listed global population level but will rely heavily upon information from the near annual SARs for the California-Oregon-Washington (CA/OR/WA) stock of the fin whale, as well as the most recent scientific information available regarding the abundance of fin whales along the U.S. west coast. The fin whales most likely to be observed within the proposed action area are identified as part of the

Population Status and Trends: Although reliable and recent estimates of fin whale abundance are available for large portions of the North Atlantic Ocean, this is not the case for most of the North Pacific Ocean and Southern Hemisphere. The status of populations in both of these ocean basins in terms of present population size relative to "initial" (pre-whaling, or carrying capacity) level is uncertain. Fin whales in the entire North Pacific have been estimated to be less than 38% of historic carrying capacity of the region (Mizroch et al. 1984). Becker et al. (2020) generated species distribution models from fixed and dynamic ocean variables in a generalized additive model framework using 1991-2018 line-transect survey data to estimate density and abundance of cetaceans in the CCE. The best estimate of fin whale abundance off the U.S. West Coast is 11,065 whales (Becker et al. 2020). Indications of recovery in CA coastal waters date back to 1979/80, but there is now strong evidence that fin whale abundance increased in the California Current between 1991 and 2018 based on analysis of line transect surveys (Carretta et al. 2021a). Mean annual abundance increased 7.5% annually during 1991 to 2014 (Carretta et al. 2021a).

Threats: A comprehensive list of general threats to fin whales is detailed in the Recovery Plan (NMFS 2010) and in the most recent 5-year status review (NMFS 2019a). Obvious threats to fin whales besides vessel interactions and fishery entanglements include reduced prey abundance due to overfishing or other factors (including climate change), habitat degradation, and disturbance from low-frequency noise. It is difficult to estimate the numbers of fin whales killed and injured by gear entanglements, because little evidence of entanglement in fishing gear exists, and large whales such as the fin whale may often die later and drift far enough to not strand on land after such incidents. Documented vessel strike deaths and serious injuries are derived from actual counts of fin whale carcasses and should be considered minimum values. Comprehensive coast-wide data on vessel strike deaths and serious injuries assumed to result in death are compiled in annual reports on observed anthropogenic mortality for the 13-year period of 2007-2019 (Carretta et al. 2021a). During this 13 year period, there were 20 observations of fin whale vessel strike deaths and 1 serious injury assumed to result in death or 1.6 fin whales annually

(Carretta et al. 2021a). Vessel strike mortality was recently estimated to be 43 fin whales per year, although this includes only the period from July – November when whales are most likely to be present in the U.S. West Coast EEZ (Rockwood et al. 2017). The threats to fin whales due to underwater noise, pollutants, marine debris, and habitat degradation, are difficult to quantify. However, there is a growing concern that the increasing levels of anthropogenic noise in the ocean may be a habitat concern for fin whales that use low frequency sound to communicate.

For the CA/OR/WA stock of fin whales, considering the effects of estimated vessel strikes (43/yr) and serious injury (2.2/yr) due to fisheries (0.64 /yr), the total incidental mortality from human impacts are less than the calculated PBR of 80 (Carretta et al. 2021a). Total fishery mortality is less than 10% of PBR and, therefore, may be approaching zero mortality and serious injury rate (Carretta et al. 2021a). However, observations alone underestimate true impacts due to incomplete detection of fishery entanglements. In recent years, there have been a number of instances where fin whales were sighted at-sea with indications of injury resulting from interaction with unknown fishing gear and other debris (NMFS WCR stranding data).

2.2.1.3 Humpback Whale, Mexico DPS and Central America DPS

Humpback whales were listed as endangered under the Endangered Species Conservation Act in June 1970 (35 FR 18319), and remained on the list of threatened and endangered species after the passage of the ESA in 1973 (35 FR 8491). A recovery plan for humpbacks was issued in November 1991 (NMFS 1991). On September 8, 2016, NMFS published a final rule dividing the globally listed endangered humpback whale into 14 distinct population segments (DPSs) and categorizing four DPSs as endangered and one as threatened (81 FR 62259). NMFS identified three humpback whale DPSs that may be found off the coasts of Washington, Oregon, California, and southern British Columbia: the Hawaii DPS (not ESA-listed), the Mexico DPS (ESA-listed as threatened), and the Central America DPS (ESA-listed as endangered). DPS abundance and geographic distribution are described below. On April 21, 2021, NMFS designated critical habitat for the Mexico and Central America DPSs in the North Pacific Ocean that include portions of the CCE, including areas off the coasts of Washington, Oregon, California, and Alaska (for the Mexico DPS). However, the proposed action does not occur within designated critical habitat for either the Mexico or Central America DPSs.

NMFS manages humpback whales that occur in waters under U.S. jurisdiction as five separate stocks under the MMPA. Along the U.S. West Coast, all humpback whales are considered part of the CA/OR/WA stock. The CA/OR/WA stock spends the winter (breeding season) primarily in coastal waters of Mexico and Central America, and the summer (foraging season) feeding primarily on euphausiids and small pelagic schooling fishes along the North American west coast from California to British Columbia. For this opinion, we will analyze effects at the ESA-listed DPS level but will rely heavily upon information from the near annual SARs for the CA/OR/WA stock of the humpback whale, as well as the most recent scientific information available regarding the abundance of humpback whales along the U.S. west coast.

Much of what we know about the current status of humpback whale DPSs results from field efforts conducted on all known winter breeding regions (2004-2006) and all known summer feeding areas (2004, 2005) for humpback whales in the North Pacific (Structure of Populations, Levels of Abundance and Status of Humpbacks (SPLASH)). This study, representing one of the

largest international collaborative studies of any whale population ever conducted, was designed to determine the abundance, trends, movements, and population structure of North Pacific humpback whales as well as to examine human impacts on the population (Calambokidis et al. 2008). NMFS has relied upon results from the SPLASH study for abundance estimates as well as movement proportions between wintering (breeding) and summer (foraging) grounds (Bettridge et al. 2015; Wade et al. 2016; Wade 2017), even though the field efforts took place nearly fifteen years ago. For the 2004-2006 humpback populations, the Wade (2021) revised abundance estimate for the Central America DPS is 755 (Coefficient of Variation (CV)=0.242) animals, and the revised abundance estimate for the Mexico DPS is 2,913 (CV=0.066) animals, using the Multistrata model (Nmulti) (which uses both winter and summer data; Table 4 in Wade 2021).

Recently, Calambokidis and Barlow (2020) provided an abundance estimate for the CA/OR/WA stock of humpbacks (and the Eastern North Pacific stock of blue whales), which has been included in the draft 2021 SAR, with revised trend information and other metrics to assist managers in assessing risk of human-related activities to this stock. Capture-recapture models for humpback whales off CA/OR showed a dramatic increase in recent years, with a trend for the population starting in 1989 (~500 animals) through 2018 increasing an average 7.5% per year, with a higher rate of increase in the late 2000s. Abundance estimate for the CA/OR/WA humpback whale stock is 4,973 (CV=0.048), with an Nmin of 4,776 animals.

Based on the information in the draft 2021 SAR (Carretta et al. 2021a) and in Calambokidis and Barlow (2020), it is clear that there have been changes in the abundance and/or distribution of humpback whale DPSs over the last 10-15 years since the data gathered during the SPLASH project was re-analyzed by Wade (2021). In July, 2021, NMFS WCR finalized an approach outlining the most current evaluation of the distribution and relative abundance of ESA-listed DPSs that occur in the waters off the U.S. west coast for use in ESA analyses of Federal actions (NMFS 2021c). Table 2 summarizes the estimated abundance and proportion of each DPS off the U.S. west coast.

Based on NMFS WCR’s July 2021 memo (NMFS 2021c), this biological opinion evaluates effects on both the Central American and Mexico DPSs of humpback whales as both are expected to occur in the action area in the relative proportions described below in Table 2. Specifically, we assume that 42% of the humpback whales present in the California (CA) and Oregon (OR) waters would be Central America DPS, and 58% would be associated with the Mexico DPS. To the extent that effects are evaluated at an individual level, these proportions would be used as the likelihood that the affected animal is from either DPS. Of note, in the southern California where the action area is proposed, one would more likely interact with a humpback whale from the Central America DPS.

Table 2. Current estimates of abundance and relative proportions of the Central America DPS (endangered), Mexico DPS (threatened), and Hawaii DPS (non-listed) found off the U.S. west coast.

Action Area	Probability that a humpback would be from Central America DPS (N = 2006)	Probability that a humpback would be from Mexico DPS (N = 2770)	Probability that a humpback would be from Hawaii (non-listed) DPS
CA/OR*	42%	58%	0%

*Probabilities are based on the assumption that both the listed DPSs have increased 6% per year (which provides a conservative estimate that 4 out of 10 humpbacks could represent the Central America DPS), and given that the potential Nmin of the CA/OR/WA humpback stock (under the MMPA) is 4,776 animals (Calambokidis and Barlow 2020). Note that an action proposed south of the Gulf of the Farallones is more likely to interact with a humpback from the Central America DPS versus a Mexico DPS (Calambokidis et al. 2017).

2.2.1.3.1 Mexico DPS

The Mexico DPS consists of whales that breed along the Pacific coast of mainland Mexico, the Baja California Peninsula and the Revillagigedo Islands. The Mexico DPS feeds across a broad geographic range from California to the Aleutian Islands, with concentrations in California-Oregon, northern Washington – southern British Columbia, northern and western Gulf of Alaska and Bering Sea feeding grounds. This DPS was determined to be discrete based on significant genetic differentiation as well as evidence for low rates of movements among breeding areas in the North Pacific based on sighting data. The Mexico DPS was determined to be significant due to the gap in breeding grounds that would occur if this DPS were to go extinct and the marked degree of genetic divergence to other populations. This DPS also differs from some other North Pacific populations in the ecological characteristics of its feeding areas (Bettridge et al. 2015).

Population Status and Trends: The Mexico DPS of humpback whales forages along the West Coast of North America as far north as the Aleutian Island and Bering Sea, Alaska. (Wade 2017) estimated the abundance of the Mexico DPS during the period of the SPLASH surveys to be 2,806 whales based on a revised analysis of the SPLASH data. More recently, Wade (2021) revised the estimate of the Mexico DPS from SPLASH data to 2,913 animals during this period. These estimates are not considered a reliable estimate of current abundance, however, because they are more than eight years old and humpback whales in the Pacific have recently experienced positive growth rates (NMFS 2021d). No specific estimate of the current growth rate for this DPS is available, but the documented positive growth rates of humpback whales along the U.S. west coast and in the North Pacific at large likely reflect growth in this DPS, given its relative population size. As described in detail in the recent NMFS WCR July 2021 memo, we assume that the population has increased by 6% annually over the last 15 years. Using Wade's (2021) population estimate of 2,913 whales based on information from 2004-2006 and the assumed 6% annual increase, the current abundance estimate for the entire Mexico DPS, only some of which occur along the U.S. west coast, is 6,981 animals.

2.2.1.3.2 Central America DPS

The Central America DPS is composed of whales that breed along the Pacific coast of Costa Rica, Panama, Guatemala, El Salvador, Honduras, and Nicaragua. Whales from this breeding ground feed almost exclusively offshore of California and Oregon in the eastern Pacific, with only a few individuals identified at the northern Washington–southern British Columbia feeding grounds. This DPS was determined to be discrete based on re-sight data as well as findings of significant genetic differentiation between it and other populations in the North Pacific. The genetic composition of the DPS is also unique in that it shares mitochondrial DNA (mtDNA) haplotypes with some Southern Hemisphere DPSs, suggesting it may serve as a conduit for gene flow between the North Pacific and Southern Hemisphere. The breeding ground of this DPS

occupies a unique ecological setting, and its primary feeding ground is in a different marine ecosystem from most other populations. Loss of this population would also result in a significant gap in the range of the species (Bettridge et al. 2015).

Population Status and Trends: The Central America DPS of humpback whales occurs along the U.S. west coast, although individuals are more likely to be found off the coast of California and Oregon. Wade (2017) estimated the abundance of the Central America DPS to be 783 whales during the SPLASH period based on a revised analysis of the SPLASH data. Wade (2021) revised the estimate of the Central America DPS from SPLASH data to 755. These estimates are not considered a reliable estimate of current abundance, however, because they are more than eight years old and humpback whales in the Pacific have recently experienced positive growth rates (NMFS 2021d). No specific estimate of the current growth rate for this DPS is available, but the documented positive growth rates of humpback whales along the U.S. west coast and in the North Pacific at large likely reflect growth in this DPS, given its relative population size. As described in detail in the recent NMFS WCR July 2021 memo, we assume that the population has increased by 6% annually over the last 15 years. Using Wade (2021)'s population estimate of 755 whales based on information from 2004-2006 and the assumed 6% annual increase, the

2.2.1.3.3 Threats to Mexico and Central America DPS

A comprehensive list of general threats to humpback whales is detailed in the Recovery Plan (NMFS 1991) and the Status Review (Bettridge et al. 2015). Similar to blue and fin whales, humpbacks globally are potentially affected by loss of habitat, loss of prey (for a variety of reasons including climate variability), underwater noise, and pollutants. Entanglement in fishing gear poses a threat to individual humpback whales throughout the Pacific. The estimated effects of fisheries on the CA/OR/WA humpback whale stock (that includes the ESA-listed Mexico and Central America DPSs) is likely underestimated. The serious injury or mortality of large whales due to entanglement in gear may go unobserved because whales swim away with a portion of the net, line, buoys, or pots. Humpback whales, especially calves and juveniles, are highly vulnerable to vessel strikes (Stevick 1999) and other interactions with non-fishing vessels. Off the U.S. west coast, humpback whale distribution overlaps significantly with the transit routes of large commercial vessels, including cruise ships, large tug and barge transport vessels, and oil tankers in the proposed action area. Whale watching boats and research activities directed toward whales may have direct or indirect effects on humpback whales as harassment may occur, preferred habitats may be abandoned, and fitness and survivability may be compromised if disturbance levels are too high (NMFS 1991). However, the most recent Status Review concluded that these activities were most likely having a negligible impact on all DPSs currently.

Along the U.S. west coast, total annual human-caused serious injury and mortality of humpback whales is the sum of commercial fishery (25.2/yr)+non-commercial sources (1.4/yr)+estimated vessel strikes (22/yr), or 48.6 humpback whales annually. Considering the effects of estimated vessel strikes plus the effects of entanglements, human impacts may be greater than the PBR allocation of 28.7 animals for U.S. waters (Carretta et al. 2021a). Other than the vessel strike estimates, most data on human-caused serious injury and mortality for this population is based on opportunistic stranding and at-sea sighting data and represents a minimum count of total

effects. There is currently no estimate of the fraction of anthropogenic injuries and deaths to humpback whales that are undocumented on the U.S. west coast. Based on strandings and at sea observations, annual humpback whale mortality and serious injury in commercial fisheries (25.2/yr) is less than the PBR of 28.7; however, if methods were available to correct for undetected serious injury and mortality, total fishery mortality and serious injury would likely exceed PBR. Observed and assigned levels of serious injury and mortality due to commercial fisheries (≥ 25.2) exceed 10% of the stock's PBR (28.7), thus, commercial fishery take levels are not approaching zero mortality and serious injury rate (Carretta et al. 2021a).

2.2.1.4 Gray Whale, Western North Pacific DPS

Western North Pacific (WNP) DPS gray whales were originally listed as endangered under the Endangered Species Conservation Act in June 1970 (35 FR 18319). WNP gray whales remain listed as endangered under the ESA (35 FR 8491). Currently there is no recovery plan for this population. There are two recognized gray whale stocks in the North Pacific: the WNP and the ENP, which is no longer listed under the ESA after being delisted in 1994 (59 FR 31094). Gray whales occur along the eastern and western margins of the North Pacific, generally migrating between summer feeding grounds in high latitudes and winter breeding grounds in lower latitudes. Gray whale migration is typically limited to relatively near shore areas along the North American west coast during the winter and spring months (November-May). Gray whales are bottom feeders, sucking in sediment and eating benthic amphipods.

Historically, the WNP gray whales were considered geographically isolated from the ENP stock; however, recent information suggests overlap between these two stocks, with WNP gray whales migrating along the U.S. West Coast along with ENP gray whales. Information from tagging, photo-identification and genetic studies show that some whales identified in the WNP off Russia have been observed in the ENP, including coastal waters of Canada, the U.S., and Mexico (Lang 2010; Weller et al. 2012; Mate et al. 2015; Urban et al. 2019). Photographs of 379 individuals identified on summer feeding grounds off Russia (316 off Sakhalin; 150 off Kamchatka) were compared to 10,685 individuals identified in Mexico breeding lagoons, with a total of 43 matches found (Urban et al. 2019). The number of whales documented moving between the WNP and ENP represents 14% of gray whales identified off Sakhalin Island and Kamchatka according to Urban et al. (2019). (Cooke et al. 2018) note that the fraction of the WNP population that migrates to the ENP is estimated to be 45-80%.

Population Status and Trends: The estimated population size from photo-ID data for Sakhalin and Kamchatka in 2016 was 290 whales (90% percentile intervals = 271 – 311; Cooke et al. 2018). Systematic counts of gray whales migrating south along the central California coast have been conducted by shore-based observers at Granite Canyon most years since 1967. The current minimum population estimate for non-ESA-listed ENP gray whales is 26,960 (Carretta et al. 2021b). The most recent minimum estimate of endangered WNP gray whale abundance is 271 individuals (Carretta et al. 2021b). At any given time during the migration, WNP gray whales could be part of the approximately 27,000 gray whales migrating through the CCE. However, the probability that any gray whale observed along the U.S. west coast would be a WNP gray whale is extremely small, i.e., less than 1% even if the entire population of WNP gray whales were part of the annual gray whale migration in the ENP.

Threats: The decline of gray whales in the WNP is attributed to commercial hunting off Korea and Japan between the 1890s and 1960s (Carretta et al. 2021b). Today, near shore industrialization and shipping congestion throughout the migratory corridors of the WNP gray whale stock represent risks by increasing the likelihood of exposure to pollutants and vessel strikes as well as general habitat degradation. The summer feeding area off Sakhalin Island is a region rich with offshore oil and gas reserves, and operations of this nature have introduced new sources of underwater noise, including seismic surveys and increased shipping traffic, as well as habitat modification and risks associated with oil spills (Weller et al. 2002). Another significant threat to gray whales in the WNP is incidental catches in coastal net fisheries, along with potential entanglement in other fixed fishing gear in the WNP (Weller et al. 2013; Lowry et al. 2018). An analysis of anthropogenic scarring of gray whales photographed off Sakhalin Island found that at least 18.7% (n=28) of 150 individuals identified between 1994 and 2005 had evidence of previous entanglements in fishing gear (Bradford et al. 2009). Given that some WNP gray whales occur in U.S. waters, there is some probability of WNP gray whales being killed or injured by vessel strikes or entangled in fishing gear within U.S. waters (Carretta et al. 2021b).

Additional threats include hunting and ocean acidification. In 2005, the Makah Indian Tribe requested authorization from NOAA/NMFS, under the MMPA and the Whaling Convention Act, to resume limited hunting of gray whales for ceremonial and subsistence purposes in the coastal portion of their usual and accustomed fishing grounds off Washington State (NOAA 2015). Ocean acidification could reduce the abundance of shell-forming organisms (Fabry et al. 2008; Hall-Spencer et al. 2008), many of which are important in the gray whales' diet (Nerini 1984).

2.2.1.5 Guadalupe Fur Seal

In the U.S., Guadalupe fur seals were listed as threatened under the ESA on December 16, 1985 (50 CFR 51252) and consequently, are listed as depleted and a strategic stock under the MMPA. The population is considered a single stock because all are recent descendants from one breeding colony at Guadalupe Island, Mexico. The state of California lists the Guadalupe fur seal as a fully protected mammal in the Fish and Game Code of California (Chapter 8, Section 4700, d), and it is also listed as a threatened species in the Fish and Game Commission California Code of Regulations (Title 14, Section 670.5, b, 6, H). The Guadalupe fur seal is also protected under the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and Mexican law. Guadalupe Island was declared a pinniped sanctuary by the Mexican government in 1975. Critical habitat has not been designated for this species in the U.S. Recently, likely in part due to their increasing trend and lack of threats, the species was “up-listed from “threatened” to “least concern” under the criteria of the International Union for Conservation of Nature’s (IUCN) Red List of Threatened Species (Aurioles-Gamboa 2015). The most recent information on Guadalupe fur seal description, range, and status can be found in Aurioles-Gamboa (2015), (Carretta et al. 2021b), and most recent Status Review (McCue et al. 2021), and is summarized below.

The Guadalupe fur seal is the only member of the genus *Arctocephalus* in the Northern Hemisphere. By 1897, the Guadalupe fur seal was believed to be extinct, until a fisherman found slightly more than two dozen at Guadalupe Island in 1926. In 1997, a second rookery was

discovered at Isla Benito del Este, Baja California, and a pup was born at San Miguel Island, California (Melin and DeLong 1999). Since 2008, individual adult females, subadult males, and between one and three pups have been observed annually on San Miguel Island (NMFS Alaska Fisheries Science Center unpublished data). Guadalupe fur seals prefer shorelines with abundant large rocks and lava blocks and are often found at the base of steep cliffs and in caves and recesses, which provide protection and cooler temperatures, particularly during the summer breeding season (in Auriolles-Gamboa 2015). There is little information on feeding habitats of the Guadalupe fur seal, but it is likely that they feed on deep-water cephalopods and small schooling fish like their northern fur seal (*Callorhinus ursinus*) relatives (Seagars 1984). Lactating females may travel a thousand miles or more over a two-week period from the breeding colony to forage. They appear to feed mainly at night, at depths of about 20 m (65 feet), with dives lasting approximately 2.5 minutes (Reeves et al. 2002), with one documented deep dive of 82 meters (Gallo-Reynoso et al. 2008).

Researchers know little about the whereabouts of Guadalupe fur seals during the non-breeding season, from September through May, but they are presumably solitary when at sea. While distribution at sea is relatively unknown until recently, Guadalupe fur seals are known to migrate at least 600 km from the rookery sites, based on observations of individuals by Seagars (1984). Recently, in 2016, satellite tags were attached to five pups on Guadalupe Island. Three pups that departed the island traveled north, from 200-1300 kilometers before the tags stopped transmitting. One of those pups was eventually found dead and emaciated in Coos Bay, Oregon (Norris et al. 2017).

In recent years, Guadalupe fur seals have been increasing in numbers at the Channel Islands and increased strandings have been observed along the entire coast of California, including several along the central California coast. In 2015, an Unusual Mortality Event (UME) for Guadalupe fur seals was declared. Guadalupe fur seal strandings began in January 2015 and were eight times higher than the historical average. Strandings have continued since 2015 and remained well above average through 2020. In September, 2021, the UME was closed by NMFS. Strandings are seasonal and generally peak in April through June of each year. Strandings in Oregon and Washington became elevated starting in 2019 and have continued to present. Strandings in these two states in 2019 were five times higher than the historical average (NMFS WCR stranding program data). Most animals were young, around one year old, post-weaning (Norris et al. 2017).

Population Status and Trends: Commercial sealing during the 19th century reduced the once-abundant Guadalupe fur seal to near extinction in 1894. Population size prior to commercial harvest is not known, but estimates range from 20,000 to 100,000 animals (Fleischer 1987). Counts of Guadalupe fur seals have been made sporadically since 1954. A few of these counts were made during the breeding season, but the majority were made at other times of the year. García-Aguilar et al. (2018) estimates the current population size to be approximately one-fifth of historical, pre-exploitation levels. The most recent population estimate of at least 31,091 is based on pup count data collected in 2013 (García-Aguilar et al. 2018; Carretta et al. 2021). These data indicate that the population of Guadalupe fur seals is increasing exponentially at an average growth rate of 5.9% per year (Carretta et al. 2021b). In the U.S., a few Guadalupe fur seals are known to inhabit California sea lion rookeries at the Channel Islands (San Nicolas

Island and San Miguel Island) (Stewart et al. 1987; National Marine Mammal Lab, unpublished data).

Threats: Although the Guadalupe fur seal population is growing, the species is still at risk due to its relatively low population (i.e., compared to other pinniped species found in the California current) and the fact that nearly all pup production occurs on one island. Since the species has recovered from a very small number of individuals, genetic diversity is expected to be low. Feeding grounds occur around the rookeries and the lower part of the California Current, which is influenced by human population centers and contaminant runoff, and extensive oil tanker traffic and offshore oil extraction activity in southern California increasing the risk of an oil spill. Sealing during the 19th century nearly exterminated the species, but with full protection in Mexico and in the U.S., we presume that Guadalupe fur seals are not presently hunted although it is not known if Guadalupe fur seals are currently being illegally killed.

Minimal conflicts with fisheries exist. Gillnet and set-net fisheries likely take some animals, particularly in areas near Guadalupe Island and San Benito Island (Aurioles-Gamboa 2015). Juvenile female Guadalupe fur seals have also stranded in central and northern California with net abrasions around the neck, fish hooks and monofilament line, and polyfilament string (Hanni et al. 1997). Guadalupe fur seals occasionally are observed hooked in the Hawaii shallow set longline fishery. Between 2013 and 2017, there were two serious and two non-serious injuries involving this species (Carretta et al. 2021a).

During El Niño events, Guadalupe fur seals may experience high pup mortality due to storms and hurricanes (Gallo-Reynoso 1994), as well as low prey availability, which is likely a cause for the elevated strandings of malnourished and emaciated pups and subadults off California beginning in 2015. Guadalupe fur seals share much of their haul-out and breeding habitat with California sea lions, which have historically suffered from viral disease outbreaks and could serve as a vector for disease transmission. During periods of low prey availability, both species may compete for resources. Exotic fauna and diseases could be introduced from humans interacting with pinnipeds on the island. Lastly, killer whales and sharks (particularly great white sharks (*Carcharodon carcharias*) have been seen with regularity around Guadalupe Island, particularly during the summer months, and are therefore likely predators of Guadalupe fur seals.

Over the most recent five-year period reviewed (2013-2017), NMFS has documented serious injury and/or mortality of Guadalupe fur seals due to marine debris (possibly discarded fishing gear) and shootings (Carretta et al. 2021a), in addition to the fisheries interactions mentioned above. Guadalupe fur seals are also susceptible to domoic acid toxicity, bacterial pneumonia and other associated effects from emaciation/malnourishment (Norris et al. 2017). Military activities in southern California could affect Guadalupe fur seals through behavioral and physiological effects from mid-frequency active sonar, underwater detonations, and missile launches, as well as from sonic booms felt on the Channel Islands following a rocket launch. Scientific research is conducted on Guadalupe fur seals, primarily on San Miguel Island, including capture and tagging of pups, juveniles, and adult females. There have been no documented injuries or deaths associated with such research. Lastly, with oil production occurring off southern California and within the range of Guadalupe fur seals, the potential for an oil spill exists that could threaten this species, depending on the extent of the spill.

2.2.2. Sea Turtles

2.2.2.1 East Pacific DPS of Green Sea Turtles

In 2016, NMFS finalized new listings for 11 green sea turtle DPSs, including listing the East Pacific DPS as threatened (81 FR 20057). The East Pacific DPS includes turtles that nest on the coast of Mexico which were historically listed under the ESA as endangered. All of the green turtle DPSs were listed as threatened, with the exception of the Central South Pacific DPS, Central West Pacific DPS, and the Mediterranean DPS, which were listed as endangered (Seminoff et al. 2015).

Green turtles are found throughout the world, occurring primarily in tropical, and to a lesser extent, subtropical waters. In the eastern Pacific, greens forage coastally from southern California in the north to Mejillones, Chile in the South. Based on mtDNA analyses, green turtles found on foraging grounds along Chile's coast originate from the Galapagos nesting beaches, while those greens foraging in the Gulf of California originate primarily from the Michoacan nesting stock. Green turtles foraging in southern California and along the Pacific coast of Baja California originate primarily from rookeries of the Islas Revillagigedos (Dutton 2003).

Population Status and Trends: Green turtles that may be found within the action area likely originate from the eastern Pacific. Green turtles in the eastern Pacific were historically considered one of the most depleted populations of green turtles in the world. The primary green turtle nesting grounds in the eastern Pacific are located in Michoacán, Mexico, and the Galapagos Islands, Ecuador (NMFS and USFWS 1998a). Here, green turtles were widespread and abundant prior to commercial exploitation and uncontrolled subsistence harvest of nesters and eggs. Sporadic nesting occurs on the Pacific coast of Costa Rica. Analysis using mtDNA sequences from three key nesting green turtle populations in the eastern Pacific indicates that they may be considered distinct management units: Michoacán, Mexico; Galapagos Islands, Ecuador, and Islas Revillagigedos, Mexico (Dutton 2003).

Information suggests steady increases in nesting at the primary nesting sites in Michoacan, Mexico, and in the Galapagos Islands since the 1990s (Delgado and Nichols 2005; Senko et al. 2011). Colola beach is the most important green turtle nesting area in the eastern Pacific; it accounts for 75% of total nesting in Michoacan and has the longest time series of monitoring data since 1981. Nesting trends at Colola have continued to increase since 2000 with the overall eastern Pacific green turtle population also increasing at other nesting beaches in the Galapagos and Costa Rica (NMFS and USFWS 2007a; Wallace et al. 2010). Based on recent nesting beach monitoring efforts, the current adult female nester population for Colola, Michoacán is over 11,000 females, making this the largest nesting aggregation in the East Pacific DPS comprising nearly 60 percent of the estimated total adult female population (Seminoff et al. 2015).

Two foraging populations of green turtles are found in U.S. waters adjacent to the proposed action area. South San Diego Bay serves as an important habitat for a resident population of up to about 60 juvenile and adult green turtles in this area (Eguchi et al. 2010). There is also an aggregation of green sea turtles that appears to be persistent in the San Gabriel River and

surrounding coastal areas (e.g., Anaheim Bay) in the vicinity of Long Beach, California (Lawson et al. 2011; Crear et al. 2016). This group of turtles has only recently been identified, and research on their abundance, behavior patterns, or relationship with the population in San Diego Bay is still in its infancy. Over the last decade of study, we have identified well over 50 different sea turtles occurring in the San Gabriel River/Anaheim Bay area (NMFS unpublished data) through research or strandings, although the duration of residence and/or transitory patterns of individuals in this area are the subject of ongoing research. Results from genetic sampling during monitoring programs or from strandings suggest that the lesser known Revillagigedo nesting population of green sea turtles is a significant source for southern California foraging (LeRoux et al. 2020).

Threats: A thorough discussion of threats to green turtles worldwide can be found in the most recent status review (Seminoff et al. 2015). Major threats include: coastal development and loss of nesting and foraging habitat; incidental capture by fisheries; and the harvest of eggs, sub-adults, and adults. Climate change is also emerging as a critical issue. Destruction, alteration, and/or degradation of nesting and near shore foraging habitat is occurring throughout the range of green turtles. These problems are particularly acute in areas with substantial or growing coastal development, beach armoring, beachfront lighting, and recreational use of beaches. In addition to damage to the nesting beaches, pollution and effects on foraging habitat is a concern. Pollution run-off can degrade sea grass beds that are the primary forage of green turtles. The majority of green turtles in coastal areas spend their time at depths less than 5 m below the surface (Schofield et al. 2007; Hazel et al. 2009), and hence are vulnerable to being struck by vessels. Collisions with vessels are known to cause significant numbers of deaths every year (NMFS and USFWS 2007a; Seminoff et al. 2015). Marine debris is also a source of concern for green sea turtles especially given their presence in nearshore coastal and estuarine habitats.

The bycatch of green sea turtles, especially in coastal fisheries, is a serious problem because in the Pacific, many of the small-scale artisanal gillnet, setnet, and longline coastal fisheries are not well regulated. These are the fisheries that are active in areas with the highest densities of green turtles (NMFS and USFWS 2007a). The meat and eggs of green turtles has long been favored throughout much of the world that has interacted with this species. As late as the mid-1970s, upwards of 80,000 eggs were harvested every night during the nesting season in Michoacán (Clifton et al. 1982). Although Mexico has implemented bans on the harvest of all turtle species in its waters and on the beaches, poaching of eggs, females on the beach, and animals in coastal waters continues. In some parts of Mexico and the eastern Pacific, consumption of green sea turtles remains a part of the cultural fabric and tradition (NMFS and USFWS 2007a).

Like other sea turtle species, increasing temperatures have the potential to skew sex ratios of hatchlings. Many rookeries are already showing a strong female bias as warmer temperatures in the nest chamber leads to more female hatchlings (Chan and Liew 1995; Kaska et al. 2006). Increased temperatures also lead to higher levels of embryonic mortality (Matsuzawa et al. 2002). An increase in typhoon frequency and severity, a predicted consequence of climate change (Webster 2005), can cause erosion which leads to high nest failure (Van Houtan and Bass 2007). Climate change may also affect green sea turtle feeding. Seagrasses are a major food source for green sea turtles and may be affected by changing water temperature and salinity (Short and Neckles 1999; Duarte 2002).

2.2.2.2 Leatherback Sea Turtles

The leatherback turtle is listed as endangered under the ESA throughout its global range (NMFS and USFWS 2020a). Increases in the number of nesting females have been noted at some sites in the Atlantic, but there have been substantial declines or collapse of some populations throughout the Pacific, such as in Malaysia, Mexico and Costa Rica. The most recent Status Review (NMFS and USFWS 2020a) found that all population segments have been and are impacted, to varying degrees, by habitat loss and modification, overutilization, predation, inadequate regulatory mechanisms, fisheries bycatch, pollution, and climate change. Based on the best available information presented herein, the Pacific leatherback population is characterized by low resiliency and redundancy. The Status Review found that all population segments met the definition of high risk of extinction as a result of reduced nesting female abundance, declining nest trends (for all but the SW Atlantic population), which exhibits extremely low abundance), and numerous, severe threats. With low abundance estimates in all four countries where the species nests in the Western Pacific, and the two countries in the Eastern Pacific, leatherback sea turtles are at an extremely high risk of being extirpated from the Pacific Ocean. Extirpation of Pacific leatherbacks would significantly contract the species' range, thus increasing the already high risk of extinction for the globally-listed entity.

A recovery plan for the U.S. Pacific populations of leatherbacks was completed in 1998 (NMFS and USFWS 1998b). In 2012, NMFS revised critical habitat for leatherbacks to include additional areas within the Pacific Ocean (77 FR 4170). The proposed action does not occur within designated critical habitat for Pacific leatherbacks.

Leatherback turtles lead a completely pelagic existence, foraging widely in temperate and tropical waters except during the nesting season, when gravid females return to tropical beaches to lay eggs. Leatherbacks are highly migratory, exploiting convergence zones and upwelling areas for foraging in the open ocean, along continental margins, and in archipelagic waters (Morreale et al. 1994; Eckert 1999; Benson et al. 2007, 2011).

In the Pacific, leatherback nesting aggregations are found in the eastern and western Pacific. In the eastern Pacific, major nesting sites are located in Mexico, Costa Rica, and to a lesser extent, Nicaragua. Nesting in the western Pacific occurs at numerous beaches in Indonesia, the Solomon Islands, Papua New Guinea, and Vanuatu, with a few nesters reported in Malaysia and only occasional reports of nesting in Thailand and Australia (Eckert et al. 2012). Leatherbacks nesting in Central America and Mexico migrate thousands of miles into tropical and temperate waters of the South Pacific (Eckert 1997; Shillinger et al. 2008). After nesting, females from the Western Pacific nesting beaches make long-distance migrations into a variety of foraging areas including the central and ENP (off of the U.S. west coast), westward to the Sulawesi and Sulu and South China Seas, or northward to the Sea of Japan (Benson et al. 2007, 2011). The IUCN Red List conducted its most recent assessment of the West Pacific Ocean subpopulation in 2013 and listed it as "Critically Endangered" due in part to its continual decline in nesting, the continued threat due to fishing, and the low number of estimated nesting females.

Population Status and Trends: Leatherbacks occur throughout the world and populations and trends vary in different regions and nesting beaches. In 1980, the leatherback population was estimated to consist of approximately 115,000 adult females globally (Pritchard 1982). By 1995, one estimate claimed this global population of adult females had declined to 34,500 (Spotila et al. 1996). Our ability to estimate leatherback population abundance is complicated by the life history of the species. Data collected at nesting beaches are often the best available data but do not provide information on life stages away from the nesting beaches (i.e., immature and mature males and immature females). Additionally, standardized nesting surveys are difficult to maintain over many, consecutive years and at all nesting beaches. Here we provide data that have been consistently collected using a standardized monitoring approach over a recent remigration interval, providing reasonable certainty that such data are representative of recent nesting at the identified beach. Although some data may have been collected at other nesting beaches, monitoring has not been recent, consistent, or standardized, thus limiting our certainty of these data; therefore data from those nesting beaches cannot be used to calculate abundance.

The East Pacific leatherback population has undergone dramatic declines over the last three generations and to date there is no sign of recovery (Wallace et al. 2013; NMFS and USFWS 2020a). In the eastern Pacific, nesting counts indicate that the population has continued to decline since the mid 1990s, leading some researchers to conclude that Pacific leatherbacks are on the verge of extirpation (Spotila et al. 1996; Spotila et al. 2000). In Costa Rica, a negative 15.5 percent trend in nesting females has been documented at Las Baulas (NMFS and USFWS 2020a). In Mexico, a positive trend has been recorded at some nesting beaches (i.e., Barra Cruz/Grande +9.5 percent), but a negative trend has been recorded in other areas (i.e., Cahuitan - 4.3 percent). Overall, the current and potential future trend for the population is uncertain and additional years of data are needed to ascertain if recovery is occurring in Mexico.

The Western Pacific leatherback metapopulation that nests in Indonesia, Papua New Guinea, Solomon Islands, and Vanuatu harbors the last remaining nesting aggregation of significant size in the Pacific. This metapopulation is the source for the leatherbacks that occur off of the U.S. west coast annually. This metapopulation is made up of small nesting aggregations scattered throughout the region, with a dense focal point on the northwest Coast of Papua Barat, Indonesia. The Jamursba Medi and Wermon nesting beaches in this area represent an estimated 50 to 75 percent of all nesting in the West Pacific (NMFS and USFWS 2020a)). Using the best available data for the West Pacific leatherback population (Fitry Pakiding, University of Papua, pers. comm. 2020) and a Bayesian steady-state model, Martin et al. (2020) provided a median estimate of the total number of nesting females (i.e., over one remigration interval) at Jamursba Medi and Wermon beaches of 790 females, with a 95 percent credible interval of 666 to 942 females, as a snapshot of current abundance in 2017. We consider this to be the best available estimate of total adult female abundance at these two nesting beaches in 2017 (based on data from 2014 through 2017). To estimate the total number of nesting females from all nesting beaches in the West Pacific, we need to consider nesting at unmonitored and irregularly monitored beaches. As noted above, an estimated 50 to 75 percent of West Pacific leatherback nesting occurs at Jamursba-Medi and Wermon beaches (Dutton et al. 2007; NMFS and USFWS 2020a). Applying the conservative estimate of 75 percent to the Martin et al. (2020) estimate of 790 nesting females at Jamursba Medi and Wermon beaches, the total number of nesting females in the West Pacific population would be 1,054 females with an overall 95 percent credible interval of 888 to 1,256

females. It should be noted that this estimate (i.e., 1,054) of nesting females for the West Pacific population based on more recent available information is an update of the NMFS and USFWS (2020a) estimate (i.e., 1,277) which was based on a simple calculation that did not provide confidence or credible intervals.

A recent NOAA-funded, World Wildlife Fund (WWF)-Indonesian assessment team identified a new leatherback nesting area in 2017 on three north coast beaches of Buru Island in Central Maluku (WWF 2018 in NMFS and USFWS 2020a). Initial monitoring of these beaches suggest that this 10.6 km stretch of shoreline supports the first substantial nesting population discovered outside of Papua, Indonesia in the last decade. Nesting activity appears to be year round with a primary summer nesting peak (May to July) and a secondary winter peak (December to February). During 2017, 203 nests were documented, of which 120 were damaged by predation (WWF 2018 in NMFS and USFWS 2020a).

Based on the estimates presented in Jones et al. (2012) for all Pacific populations, NMFS inferred an estimated West Pacific leatherback total population size (i.e., juveniles and adults) of 250,000 (95 percent confidence interval 97,000 to 535,000) for 2004. Based on the relative change in the estimates derived from Jones et al. (2012) and the more recent Martin et al. (2020), we estimate the current juvenile and adult population size of the West Pacific leatherback population is around 100,000 sea turtles (95 percent confidence interval 47,000 to 195,000 individuals).

The Western Pacific population exhibits low hatching success and decreasing nest and population trends due to past and current threats, which are likely to further lower abundance and increase the risk of extinction (NMFS and USFWS 2020a). The low estimated nesting female abundance of the West Pacific population places it at elevated risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes ((NMFS and USFWS 2020a). These processes, working alone or in concert, place small populations at a greater extinction risk than large populations, which are better able to absorb impacts to habitat or losses in individuals (NMFS and USFWS 2020a). Low site fidelity and dispersal of nests among various beaches may help to reduce population level impacts from threats which may disproportionately affect one area over another, but may also place nests in locations that are likely unmonitored and not protected from human poaching or predation, thereby increasing threats to the population. Due to its small size, this population has restricted capacity to buffer such losses (NMFS and USFWS 2020a).

The median trend in annual nest counts estimated for Jamursba Medi nesting beaches from data collected from 2001-2017 was -5.7 percent annually (NMFS and USFWS 2020a). In the absence of population trend data on other leatherback life history stages, we consider these trends in annual nest counts an index of the population's growth rate. Martin et al. (2020) estimated the mean and median time until the West Pacific population declines to 50 percent, 25 percent, and 12.5 percent of its current estimated abundance. Results of this modelling effort indicate that the adult female portion of West Pacific leatherbacks nesting at Jamursba-Medi and Wermon beaches are predicted to decline to 50 percent of their current abundance in a mean of about 13 years (95 percent CI from 5 to 26 years) and to 25 percent of their current abundance in a mean of about 24 years (95 percent CI from 13 to 42 years).

Several datasets indicate or support that leatherbacks found off the U.S. west coast are from the western Pacific nesting populations, specifically boreal summer nesters, including: satellite tracking of post-nesting females and foraging males and females; genetic analyses of leatherback turtles caught in U.S. Pacific fisheries or stranded on the U.S. west coast; and stable isotope analysis. Given the relative size of the nesting populations, it is likely that the majority of the animals originate from the Jamursba-Medi nesting beaches, although some may come from the comparatively small number of summer nesters at Wermon in Papua Barat, Indonesia. The Jamursba-Medi nesting population generally exhibits site fidelity to the central California foraging area, and it has been estimated that approximately 30 to 60% of Jamursba-Medi summer nesters may have foraged in waters off California during some recent years (Benson et al. 2011; Seminoff et al. 2012). Previously, surveys in neritic waters off central and northern California estimated that, on average, approximately 180 leatherbacks (both males and females, subadults and adults) would be expected to be found off the California coast each year (Benson et al. 2007).

In recent years, surveys of leatherback abundance off the U.S. west coast have also detected a decline similar to what has been documented at the nesting beaches (Benson et al. 2020). The updated analysis from Benson et al. (2020) estimates the average number of leatherbacks off central California each year has dropped from 128 to 55 since 2003.

Threats: The primary threats identified for leatherbacks are fishery bycatch and effects at or adjacent to the nesting beaches, including degradation of nesting habitat (erosion, logging, elevated sand temperatures, human/animal encroachment), direct harvest, and predation. In the western Pacific, leatherbacks are also subjected to traditional harvest, which was well documented in the 1980s and continues today. Traditional hunters from the Kei Islands continue to kill leatherbacks for consumption and ceremony. Recent surveys indicate that harvest continues with estimates of 431 takes over the past 8 years (53.9/yr), and at least 103 leatherbacks harvested in 2017 (WWF 2018 in NMFS and USFWS 2020a).

Leatherbacks are vulnerable to bycatch in a variety of fisheries, including longline, drift gillnet, set gillnet, bottom trawling, dredge, and pot/trap fisheries that are operated on the high seas or in coastal areas throughout the species' range. Off the U.S. west coast, a large time/area closure was implemented in 2001 to protect Pacific leatherbacks by restricting the California thresher shark/swordfish drift gillnet fishery. This closure significantly reduced bycatch of leatherbacks in that fishery. On the high seas, bycatch in longline fisheries is considered a major threat to leatherbacks (Lewison et al. 2004). In addition to anthropogenic factors, natural threats to nesting beaches and marine habitats such as coastal erosion, seasonal storms, predators, temperature variations, and phenomena such as El Niño also affect the survival and recovery of leatherback populations (Eckert et al. 2012).

There are interactions between leatherbacks and domestic longline fishing for tuna and swordfish based out of Hawaii. Under requirements established in 2004 to minimize sea turtle bycatch (69 FR 17329), vessel operators in the Hawaii-based shallow-set swordfish fishery must use large (sized 18/0 or larger) circle hooks with a maximum of 10 degrees offset and mackerel-type bait. From 2012-2017, the incidental take statement for the Hawaii-based shallow-set fishery was 26

leatherback sea turtles per year, which served as the “hard cap” for the fishery that requires closure of the entire fishery during any year if reached. Recently, the hard cap for leatherback sea turtle bycatch was reset to 16 per year, with the expectations that up to 16 may be caught and 3 may be killed each year and that vessels would be restricted to no more than 2 leatherbacks taken during any one trip (NMFS 2019b). Between 2004 and 2018, there were a total of 105 leatherback sea turtles captured in the fishery, with an estimated 21 leatherback sea turtles killed as a result (NMFS 2019b). In the deep-set longline tuna fishery based out of Hawaii, NMFS exempted the take (interactions or mortalities) of up to 72 interactions and 27 mortalities of leatherbacks over a 3-year period (NMFS 2014). Based on observer data from 2012-2018 (over 20% observer coverage, on average), NMFS estimates that a total of 85 leatherbacks were captured, including 36 mortalities (NMFS 2019b). Since the start of the observer program in American Samoa in 2006 through 2018, the American Samoa longline fishery is estimated to have had 55 interactions, with 38 mortalities (NMFS 2019b).

Estimating the total number of sea turtle interactions in other Pacific fisheries that interact with the same sea turtle populations as U.S. fisheries is difficult because of low observer coverage and inconsistent reporting from international fleets. Lewison et al. (2004) estimated 1,000 to 3,200 leatherback mortalities from pelagic longlining in the Pacific in 2000. Beverly and Chapman (2007) more recently estimated loggerhead and leatherback longline bycatch in the Pacific to be approximately 20% of that estimated by Lewison et al. (2004), which would equate to 200 to 640 leatherbacks during that time period. Chan and Pan (2012) estimated that there were approximately 1,866 total sea turtle interactions of all species in 2009 in the central and North Pacific by comparing swordfish production and turtle bycatch rates from fleets fishing in the central and North Pacific area. In 2015, a workshop was convened to analyze the effectiveness of sea turtle mitigation measures in the tuna Regional Fishery Management Organizations (RFMOs) and 16 countries provided data on observed sea turtle interactions and gear configurations in the Western Central Pacific Ocean. Based on the information gathered there, 331 leatherback sea turtle interactions were reported with a total estimate of 6,620 leatherbacks caught in the region from 1989-2015 in these countries. Most recently, Peatman et al. (2018) estimated that 9,923 leatherbacks were captured in longline fisheries operating in the North Pacific from 2003-2017.

Given that recent developments to reduce sea turtle bycatch in fisheries have been working their way into some international fisheries and the incomplete data sets and reporting that exist, the exact level of current sea turtle bycatch internationally is not clear. However, given the information that is available, we believe that international bycatch of sea turtles in fisheries throughout the Pacific Ocean continues to occur at significant rates, likely several orders of magnitude greater than what NMFS’s documents or anticipates in U.S. domestic Pacific ocean fisheries.

In an attempt to develop a tool for managers to use locally (e.g., in an EEZ) to reduce threats in a particular area of interest, Curtis et al. (2015) developed biological “limit reference points” for western Pacific leatherback turtles in the U.S. West Coast EEZ, similar to a PBR approach calculated for marine mammal stocks. Depending on the model used and the various objectives sought (e.g., achievement of maximum net productivity, or no more than a 10% delay in the time for the population to rebuild) and incorporation of conservative assumptions accounting for

broad uncertainty in abundance and productivity estimates, the limit reference point estimate for human-caused removals in the U.S. West Coast EEZ ranged from 0.8 to 7.7 leatherbacks over 5 years. Although these results are useful for consideration, NMFS is not currently using this approach to manage threats to sea turtles foraging within the U.S. West Coast EEZ, pending further discussion of how this approach or other approaches relate to the standards of the ESA. We anticipate that the management tool presented by Curtis et al. (2015) and other approaches to managing threats to sea turtles will be subject to future discussion by scientific experts.

2.2.2.3 North Pacific DPS Loggerhead Sea Turtles

A recovery plan for the U.S. Pacific populations of loggerheads was completed in 1998 (NMFS and USFWS 1998c) when loggerheads were listed globally as a threatened species under the ESA. In 2011, a final rule was published describing ESA-listings for nine DPSs of loggerhead sea turtles worldwide (76 FR 58868). The North Pacific Ocean DPS of loggerheads, which is the population of loggerheads likely to be exposed to the proposed action, was listed as endangered. Since the loggerhead listing was revised in 2011, a recovery plan for the North Pacific loggerhead DPS has not been completed. However, through a U.S. initiative, three countries (United States, Japan, and Mexico) have been developing a tri-national recovery plan (A. Gutierrez, NMFS, personal communication, 2017).

Loggerheads are circumglobal, inhabiting continental shelves, bays, estuaries, and lagoons in temperate, subtropical, and tropical waters. Major nesting grounds are generally located in temperate and subtropical regions, with scattered nesting in the tropics. Juvenile loggerheads originating from nesting beaches in the western Pacific Ocean appear to use oceanic developmental habitats and move with the predominant ocean gyres for many years before returning to their neritic foraging habitats (Pitman 1990; Bowen et al. 1995; Musick and Limpus 1997). Recent resident times of juvenile North Pacific loggerheads foraging at a known hotspot off Baja California were estimated at over 20 years, with turtles ranging in age from 3 to 24 years old (Tomaszewicz et al. 2015). After spending years foraging in the central and eastern Pacific, loggerheads return to their natal beaches for reproduction (Resendiz et al. 1998; Nichols et al. 2000) and remain in the western Pacific for the remainder of their life cycle (Iwamoto et al. 1985; Kamezaki et al. 1997; Hatase et al. 2002; Conant et al. 2009).

In the western Pacific, the only major nesting beaches are in the southern part of Japan (Dodd 1988). Satellite tracking of juvenile loggerheads indicates the Kuroshio Extension Bifurcation Region in the central Pacific to be an important pelagic foraging area for juvenile loggerheads (Polovina et al. 2006; Howell et al. 2008; Kobayashi et al. 2008). Researchers have identified other important juvenile turtle foraging areas off the coast of Baja California Sur, Mexico (Peckham et al. 2007; Conant et al. 2009). Loggerheads documented off the U.S. west coast are primarily found south of Point Conception, California, in the SCB. South of Point Eugenia on the Pacific coast of Baja California, pelagic red crabs (*Pleuroncodes planipes*) have been found in great numbers, attracting top predators such as tunas, whales, and sea turtles, particularly loggerheads (Pitman 1990; Wingfield et al. 2011; Seminoff et al. 2014).

Population Status and Trends: The North Pacific loggerhead DPS nests primarily in Japan (Kamezaki et al. 2003), although low level nesting may occur outside of Japan in areas

surrounding the South China Sea (Chan et al. 2007; Conant et al. 2009). Along the Japanese coast, nine major nesting beaches (greater than 100 nests per season) and six “submajor” beaches (10–100 nests per season) exist, including Yakushima Island where over 50% percent of nesting occurs (Kamezaki et al. 2003; Jones et al. 2018). Census data from 12 of these 15 beaches provide composite information on longer term trends in the Japanese nesting assemblage. From this data, Kamezaki et al. (2003) concluded a substantial decline (50–90%) in the size of the annual loggerhead nesting population in Japan had occurred since the 1950s. As discussed in the 2011 final ESA listing determination, current nesting in Japan represents a fraction of historical nesting levels (Conant et al. 2009) (76 FR 58868). Nesting declined steeply from an initial peak of approximately 6,638 nests in 1990–1991, to a low of 2,064 nests in 1997. Since that time, nesting has been variable, increasing and decreasing over time as is typical of sea turtle nesting trends. Nesting increased gradually to 5,167 nests in 2005 (Conant et al. 2009), peaked to 11,082 nests in 2008, declined and then has risen steadily to a record high of 15,396 nests in 2013 (Matsuzawa 2009, 2010, 2012)(Y. Matsuzawa pers. comm. 2014). Nesting activity declined in 2014 to less than 10,000 nests, and again in 2015 with less than 5,000 nests laid, but increased slightly in 2016 (NMFS 2019b).

In terms of abundance, Van Houtan (2011) estimated the total number of adult nesting females in the population was 7,138 for the period 2008-2010. An abundance assessment using data available through 2013 was conducted by Casale and Matsuzawa (2015) as part of an IUCN Red List assessment that estimated 8,100 nesting females in the population. More recently, Jones et al. (2018) used a model estimate of 3,632 females nesting in Yakushima, assumed to represent 52% of all nesting females in the population, to estimate the total number of North Pacific loggerhead nesting females at 7,000 (NMFS 2019b). Most recently, Martin et al. (2020) estimated the current loggerhead abundance at 4,541 (95% CI: 4074–5063) for all nesting females in Yakushima. In total, Jones estimated that there are approximately 340,000 loggerhead sea turtles of all ages in the North Pacific population (Jones 2019 as cited in NMFS 2019b).

In a recent consultation completed on the Hawaii-based shallow-set longline fishery (NMFS 2019b), NMFS conducted analyses to estimate the growth rate for the Yakushima portion of the North Pacific loggerhead population, along with the probabilities of this subpopulation reaching abundance thresholds within a 100 year projection period, and time in years (mean, median, & 95% credible interval) to reach the threshold for all runs that fall below the threshold (Jones et al. 2018). The results indicated the current mean growth rate (λ) is 1.024 (95% confidence interval 0.897 to 1.168), which suggest that most trajectories of this subpopulation can be expected to increase slightly in the coming years (NMFS 2019b). Most recently, Martin et al. (2020) used a Bayesian state-space population growth model that estimated an increasing trend for loggerheads (2.3% annually; 95% CI: –11.1% to 15.6%)

As noted above, North Pacific loggerheads have been documented in high numbers off the central Pacific coast of Baja California, Mexico. Aerial surveys conducted from 2005 through 2007 in the Gulf of Ulloa, a known “hot spot,” provided an estimated foraging population of over 43,000 juvenile loggerheads (Seminoff et al. 2014). NMFS conducted aerial surveys of the SCB in 2015 (a year when sea surface temperatures were anomalously warm and an El Niño was occurring) and estimated more than 70,000 loggerheads throughout the area (Eguchi et al. 2018), likely feeding on pelagic red crabs and pyrosomes, the species’ preferred prey. Recent analysis

of loggerhead sea turtle presence in the SCB suggests that loggerhead presence offshore of Southern California is tied not just to warm temperatures, but to persistently warm temperatures over a period of months such as what occurred during the recent large marine heatwave in the Eastern North Pacific Ocean (Welch et al. 2019).

Recent efforts have examined potential relationships between significant climate/environmental variables and influences on turtle populations. Van Houtan and Halley (2011) identified correlations between loggerhead juvenile recruitment and breeding remigrations and two strong environmental influences: sea surface temperature and the PDO index of ocean circulation. The mechanisms that could influence loggerhead survival at important stages may be relevant to understanding past nesting beach trends, and this is a promising avenue of research. However, there are many more anthropogenic and natural factors that may influence sea turtle populations and future trends. Differences in ocean basins, nesting assemblages, demographics, and habitat, among other variables, need to be included in any characterization of status and trends for a particular population or DPS, such as North Pacific loggerheads.

Relating environmental variance into population dynamics is an important component in our attempts to understand the fate of long-lived and highly migratory marine species such as sea turtles. However, we cannot currently reliably predict the magnitude of future climate change and its impacts on North Pacific loggerheads. In addition, as noted by Arendt et al. (2013), Van Houtan and Halley (2011) proposed an alternative to a long-held paradigm that the survivorship of large juveniles and adult sea turtles is more predictive of population change than juvenile recruitment. Van Houtan and Halley (2011) suggested that cohort effects stemming from survival in the first year of life had a greater effect on population growth. Analyses conducted by Arendt et al. (2013) on climate forcing on annual nesting variability of loggerheads in the Northwest Atlantic Ocean showed that trends in annual nest counts are influenced more by remigrants than by neophytes, which contradicts in part the Van Houtan and Halley (2011) study. As summarized above, the North Pacific loggerhead nesting population has been increasing over the last couple of decades, including the most recent years (2010-current) not included in the Van Houtan and Halley (2011) analysis. This may be explained by conservation efforts on the nesting beaches, at the foraging grounds (e.g., Gulf of Ulloa, in Baja California, Mexico), and potentially realized reduction of threats from large-scale fisheries such as longlining.

Threats: A detailed account of threats to loggerhead sea turtles around the world is provided in recent status reviews (Conant et al. 2009; NMFS and USFWS 2020b). The most significant threats facing loggerheads in the North Pacific include coastal development and bycatch in commercial fisheries. Destruction and alteration of loggerhead nesting habitats is occurring throughout the species' range, especially due to coastal development, beach armoring, beachfront lighting, and vehicular/pedestrian traffic. Overall, the NMFS and USFWS have concluded that coastal development and coastal armoring on nesting beaches in Japan are significant threats to the persistence of this DPS (76 FR 58868).

Bycatch in commercial fisheries is a major threat throughout the species range, affecting both juveniles and adults; bycatch occurs in both coastal and pelagic fisheries involving longline, drift gillnet, set-net, bottom trawl, dredge, and pound net gear (Conant et al. 2009). Specifically in the Pacific, bycatch continues to be reported in gillnet and longline fisheries operating in “hotspot”

areas where loggerheads are known to congregate (Peckham et al. 2007). Interactions and mortality associated with coastal and artisanal fisheries in Mexico and the Asian region likely represent the most serious threats to North Pacific loggerheads (Peckham et al. 2007; Conant et al. 2009; Ishihara et al. 2011).

In Mexico, loggerhead mortality has been significantly reduced, particularly in a previously identified hotspot where thousands of loggerheads may forage for many years until reaching maturity. In 2013, Mexico was notified that if it did not establish a regulatory program comparable in effectiveness to that of the United States, Mexico would receive a “negative certification” under section 403(a) of the MSA. This notification was made as a result of documented evidence of hundreds of loggerheads found stranded or bycaught in coastal artisanal fisheries in the Gulf of Ulloa, off the Pacific coast of Baja California. As a result, in 2016, Mexico published new regulations to establish a reserve in the loggerhead hotspot area and set a loggerhead turtle mortality limit for commercial fishing vessels of 90 turtles within the reserve. If that 90 turtle mortality threshold is met, Mexico will suspend gillnet fishing from May through August to protect loggerhead sea turtles. Restrictions on mesh size and soak time were also included to reduce mortalities. After reviewing the regulations, the United States was able to positively certify Mexico in September 2016 (Department of Commerce 2015). This restriction likely reduces loggerhead bycatch by an order of magnitude and addresses one of the primary threats identified in Conant et al. (2009).

There are interactions between North Pacific loggerheads and domestic longline fishing for tuna and swordfish based out of Hawaii. Under requirements established in 2004 to minimize sea turtle bycatch (69 FR 17329), vessel operators in the Hawaii-based shallow-set swordfish fishery must use large (sized 18/0 or larger) circle hooks with a maximum of 10 degrees offset and mackerel-type bait. From 2012-2017, the incidental take statement for the Hawaii-based shallow-set fishery was 34 loggerhead turtles per year, which served as the “hard cap” for the fishery that requires closure of the entire fishery during any year if reach. Recently, the hard cap for loggerhead sea turtle bycatch was removed, with the expectations that up to 36 may be caught and 6 may be killed each year and that vessels would be restricted to no more than 5 loggerheads taken during any one trip (NMFS 2019b). From 2004 to 2018, the Hawaii-based shallow-set fishery captured a total of 177 loggerheads (11.8/year) with 2 observed mortalities (NMFS 2019b).

In the deep-set longline tuna fishery based out of Hawaii, NMFS exempted the take (interactions or mortalities) of up to 18 North Pacific loggerheads over a 3-year period (NMFS 2014). Based on observer data from 2012-2018 (over 20% observer coverage, on average), NMFS estimates that a total of 45 loggerheads were captured, including 30 mortalities (NMFS 2019b).

Estimating the total number of sea turtle interactions in other Pacific fisheries that interact with the same sea turtle populations as U.S. fisheries is difficult because of low observer coverage and inconsistent reporting from international fleets. Lewison et al. (2004) estimated 2,600 – 6,000 loggerhead mortalities from pelagic longlining in the Pacific in 2000. Beverly and Chapman (2007) more recently estimated loggerhead and leatherback longline bycatch in the Pacific to be approximately 20% of that estimated by Lewison et al. (2004), which would equate to between 520 and 1,200 loggerhead mortalities during the year assessed. Chan and Pan (2012) estimated

that there were approximately 1,866 total sea turtle interactions of all species in 2009 in the central and North Pacific by comparing swordfish production and turtle bycatch rates from fleets fishing in the central and North Pacific area. In 2015, a workshop was convened to analyze the effectiveness of sea turtle mitigation measures in the tuna RFMOs and 16 countries provided data on observed sea turtle interactions and gear configurations in the Western Central Pacific Ocean. Based on the information gathered there, 549 loggerhead sea turtle interactions were reported with a total estimate of 10,980 loggerheads caught in the region from 1989-2015 by these countries. Most recently, Peatman et al. (2018) estimated that between 473 to 2,941 loggerheads were captured in longline fisheries operating in the North Pacific from 2003-2017.

Between recent developments to reduce sea turtle bycatch in domestic fisheries that have been working their way into some international fisheries and the incomplete data sets and reporting that exists, the exact level of current sea turtle bycatch internationally is not clear. Given the information that is available, we believe that international bycatch of sea turtles in fisheries throughout the Pacific Ocean continues to occur at significant rates, likely several orders of magnitude greater than what is documented or anticipated in U.S. Pacific Ocean fisheries.

2.2.2.4 Olive Ridley Sea Turtles

A recovery plan for the U.S. Pacific populations of olive ridleys was completed in 1998 (NMFS and USFWS 1998d). A 5-year status review of olive ridley sea turtles was completed in 2014 (NMFS and USFWS 2014). Although the olive ridley sea turtle is regarded as the most abundant sea turtle in the world, olive ridley nesting populations on the Pacific coast of Mexico are listed as endangered under the ESA; all other populations are listed as threatened. We assume that olive ridley turtles that may occur in the action area along the U.S. west coast are most likely from the Pacific Coast of Mexico given the relative proximity of the action area to the Pacific coast of Mexico compared to other nesting populations in the North Pacific Ocean.

Olive ridley sea turtles occur throughout the world, primarily in tropical and sub-tropical waters. Nesting aggregations in the Pacific Ocean are found in the Marianas Islands, Australia, Indonesia, Malaysia, and Japan (western Pacific), and Mexico, Costa Rica, Guatemala, and South America (eastern Pacific). Like leatherback turtles, most olive ridley sea turtles lead a primarily pelagic existence (Plotkin et al. 1993), migrating throughout the Pacific, from their nesting grounds in Mexico and Central America to the deep waters of the Pacific that are used as foraging areas (Plotkin et al. 1994). While olive ridleys generally have a tropical to subtropical range, with a distribution from Baja California, Mexico to Chile (Silva-Batiz et al. 1996), individuals do occasionally venture north, some as far as the Gulf of Alaska (Hodge and Wing 2000). Olive ridleys live within two distinct oceanic regions, including the subtropical gyre and oceanic currents in the Pacific. The gyre contains warm surface waters and a deep thermocline preferred by olive ridleys. The currents bordering the subtropical gyre, the Kuroshio Extension Current, North Equatorial Current and the Equatorial Counter Current, all provide for advantages in movement with zonal currents and location of prey species (Polovina et al. 2004).

Population Status and Trends: It is estimated that there are over one million female olive ridley sea turtles nesting annually along the Pacific coast of Mexico (NMFS and USFWS 2014). Unlike other sea turtle species, most female olive ridleys nest annually. According to the Marine Turtle

Specialist Group of the IUCN, there has been a 50% decline in olive ridleys worldwide since the 1960s, although there have recently been substantial increases at some nesting sites (NMFS and USFWS 2007b). A major nesting population exists in the eastern Pacific on the West Coast of Mexico and Central America. Both of these populations use the north Pacific as foraging grounds (Polovina et al. 2004). Because the proposed action occurs closer to eastern Pacific nesting and foraging sites, we assume that this population would be more likely (i.e., than the western Pacific population) to be affected by the proposed action, and that any affected turtles may have originated from the endangered Mexican breeding population. The eastern Pacific population is thought to be increasing, while there is inadequate information to suggest trends for other populations. Eastern Pacific olive ridleys nest primarily in large aggregations called arribadas on the west coasts of Mexico and Costa Rica. Since reduction or cessation of egg and turtle harvest in both countries in the early 1990s, annual nest totals have increased substantially. On the Mexican coast alone, in 2004-2006, the annual total was estimated at 1,021,500 – 1,206,000 nests annually (NMFS and USFWS 2007b). Eguchi et al. (2007) analyzed sightings of olive ridleys at sea, leading to an estimate of 1,150,000 – 1,620,000 turtles in the eastern tropical Pacific in 1998-2006.

Threats: Threats to olive ridleys are described in the most recent five year status review (NMFS and USFWS 2014). Direct harvest and fishery bycatch are considered the two biggest threats. In the 1950s through the 1970s, it is estimated that millions of olive ridleys were killed for meat and leather and millions of eggs were collected at nesting beaches in Mexico, Costa Rica, and other locations in Central and South America. Harvest was reduced in the 1980's and 1990's, although eggs are still harvested in parts of Costa Rica and there is an illegal harvest of eggs in parts of Central America and India (NMFS and USFWS 2014).

Olive ridleys have been observed as bycatch in a variety of fishing gear including longline, drift gillnet, set gillnet, bottom trawl, dredge, and trap net. Fisheries operating in coastal waters near arribadas can kill tens of thousands of adults. This is evident on the east coast of India where thousands of carcasses wash ashore after drowning in coastal trawl and drift gillnets fishing near the huge arribada (NMFS and USFWS 2007b).

2.2.3. White Abalone

White abalone range from Point Conception, California, to Punta Abreojos, Baja California, Mexico (Bartsch 1940; Cox 1960, 1962; Leighton 1972). Adults occupy open, low relief rocky reefs or boulder habitat surrounded by sand (Hobday and Tegner 2000). Because suitable habitat is patchy, the distribution of white abalone is also patchy (NMFS 2008). They are the deepest living abalone species on the North American West Coast, occupying depths from 5-60m (Cox 1960).

Population Status and Trends: White abalone face a high risk of extinction. NMFS's listed white abalone as endangered under the ESA in 2001 (66 FR 29046; May 29, 2001), primarily due to low densities resulting from historical overfishing. White abalone were subject to serial depletion by the commercial fishery in the early 1970s and suffered the most dramatic declines of the five abalone species (Karpov et al. 2000). During the main period of commercial harvest of white abalone (1969-1981), landings peaked in 1972, but declined to nearly zero by the early

1980s and remained low until the fishery was closed in 1996 (Karpov et al. 2000). Fishery independent surveys also show severe declines in abundance and density. Abundance estimates for the 1960s to 1970s ranged from about 600,000 to 1.7 million white abalone (Tutschulte 1976; Rogers-Bennett et al. 2002), whereas estimates for the 1990s were around 2,000 white abalone, or about 0.1% of estimated pre-exploitation abundance (Hobday et al. 2001). More recent surveys indicate greater numbers than previously estimated (e.g., about 1,900 animals at San Clemente Island and 5,800 animals at Tanner Bank in 2004) (Butler et al. 2006). ROV surveys conducted at Tanner Bank show continued declines in white abalone abundance and density over the period from 2002-2010, with fewer animals in close proximity to one another (Stierhoff et al. 2012).

In recent years, increased survey efforts along the mainland southern California coast has led to more observations of white abalone and evidence of recruitment in the wild. From 2010 to 2016, white abalone ($n = 67$) ranging in size from 130-187 mm shell length (SL) have been observed in areas where they had not been observed for 10 or more years, including off the mainland California coast (e.g., Palos Verdes Peninsula, La Jolla, and Point Loma) (Neuman et al. 2015). These observations show that individuals in the wild have been able to reproduce and recruit successfully, though likely not at a broad enough scale or high enough rate to support recovery.

In Mexico, very little data is available on white abalone populations. White abalone are commercially harvested along with four other abalone species off Baja California. Where information is available, the estimated proportion of white abalone in the catch has varied from less than 1% to 65%, depending on the year and location (Hobday and Tegner 2000). Only two fishery-independent surveys have been conducted. Estimated densities in 1968-1970 ranged from 0.07 to 0.149 abalone per m^2 , whereas no white abalone were found in 1977-1978 (Guzman-Del Proo 1992). Based on the limited data available, white abalone populations in Mexico have likely declined since the 1970s and may be at a level where recruitment failure has already occurred in some areas (Hobday and Tegner 2000).

The fragmented populations that remain in the wild are likely unable to reproduce successfully or at levels needed for recovery (NMFS 2021a). Much progress has been made toward recovery since 2001. Expanded field monitoring off southern California and Mexico supports improved assessments of the species' status in the wild (NMFS 2021a). Recovery efforts are aimed at increasing densities in the wild, to support successful reproduction and establish self-sustaining populations. The increased success and expansion of captive production led to the first ever outplanting of captive-bred white abalone to the wild in 2019 at two sites, including one site off Point Loma within the action area (NMFS 2021a). Several outplanting efforts have been conducted since 2019, with several more planned over the next five years.

Threats: In California, the species' abundance and density have declined substantially, resulting in low reproductive and recruitment success, such that the remaining animals in the wild do not appear to be replacing themselves. The primary threat to the species is their current low densities and spatial distribution, where animals may be too far apart to reproduce successfully or at levels needed for recovery. Complete and partial closures of the abalone fishery have been proposed in Mexico, but we do not know whether they have been adopted and implemented. Illegal harvest of undersized white abalone remains a problem in Mexico, but we have limited information on

the problem’s extent (NMFS 2008). Recovery will involve: (1) protecting the remaining animals in the wild; (2) promoting natural reproduction at a level that can sustain the population, by increasing the abundance and density of white abalone in the wild (e.g., through captive breeding and outplanting); and (3) monitoring wild populations in California and Baja California to assess the species’ status throughout its range.

2.3. Action Area

“Action area” means all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action (50 CFR 402.02).

The action area is the Pacific Ocean encompassing areas where plume has been detected via the benthic monitoring stations that are located in the vicinity of the coasts of La Jolla and Baja California (see Figure 3-5) (City of San Diego 2021a; City of San Diego 2020). The action area encompasses areas where the plume has been detected from both the PLOO and SBOO since it appears the area overlaps at the edges of the monitoring stations and one cannot distinguish the plume from either sources (PLOO and SBOO). This is also consistent with our understanding that the plume distribution and extent are influenced by oceanographic conditions and processes, which result in variable seawater movement within the action area.

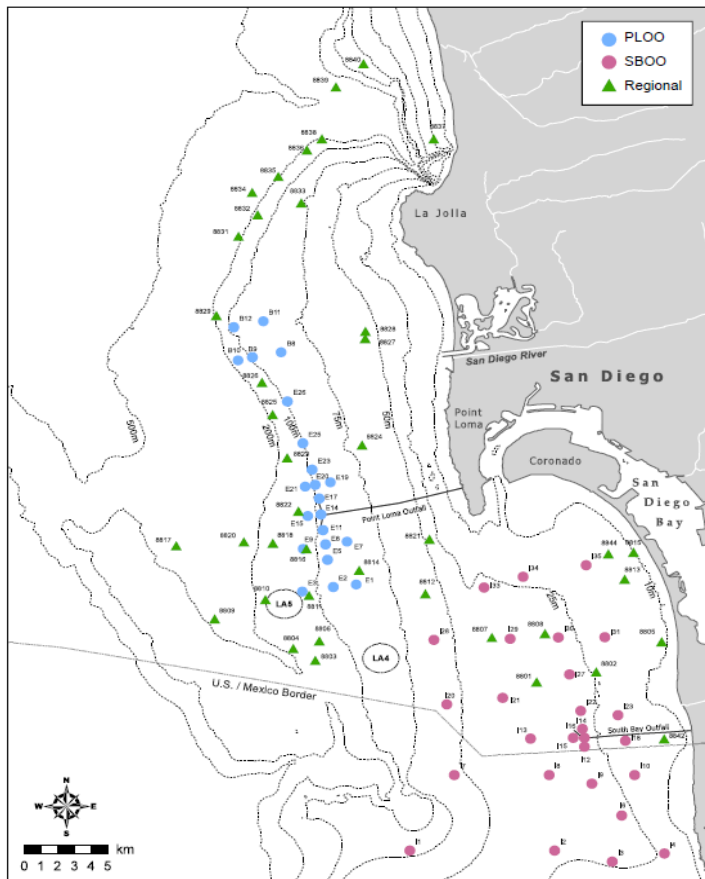


Figure 3. Action area encompassing areas where benthic monitoring stations are located in the vicinity of the coasts of La Jolla and Baja California.

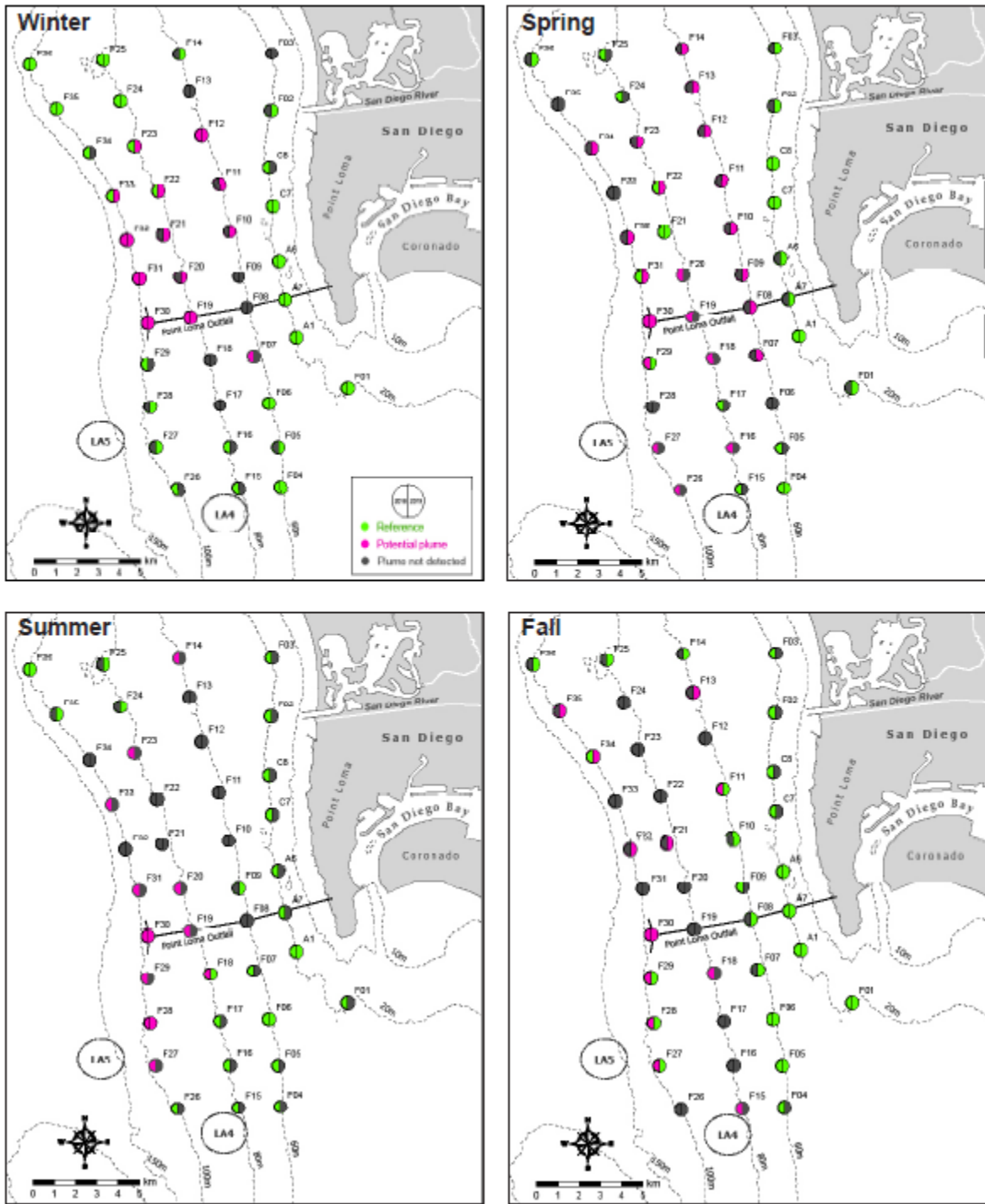


Figure 4. Distribution of stations meeting potential plume criteria (pink) and those used as reference stations (green) near the PLOO during quarterly surveys in 2018 (left half of pie) and 2019 (right half of pie).

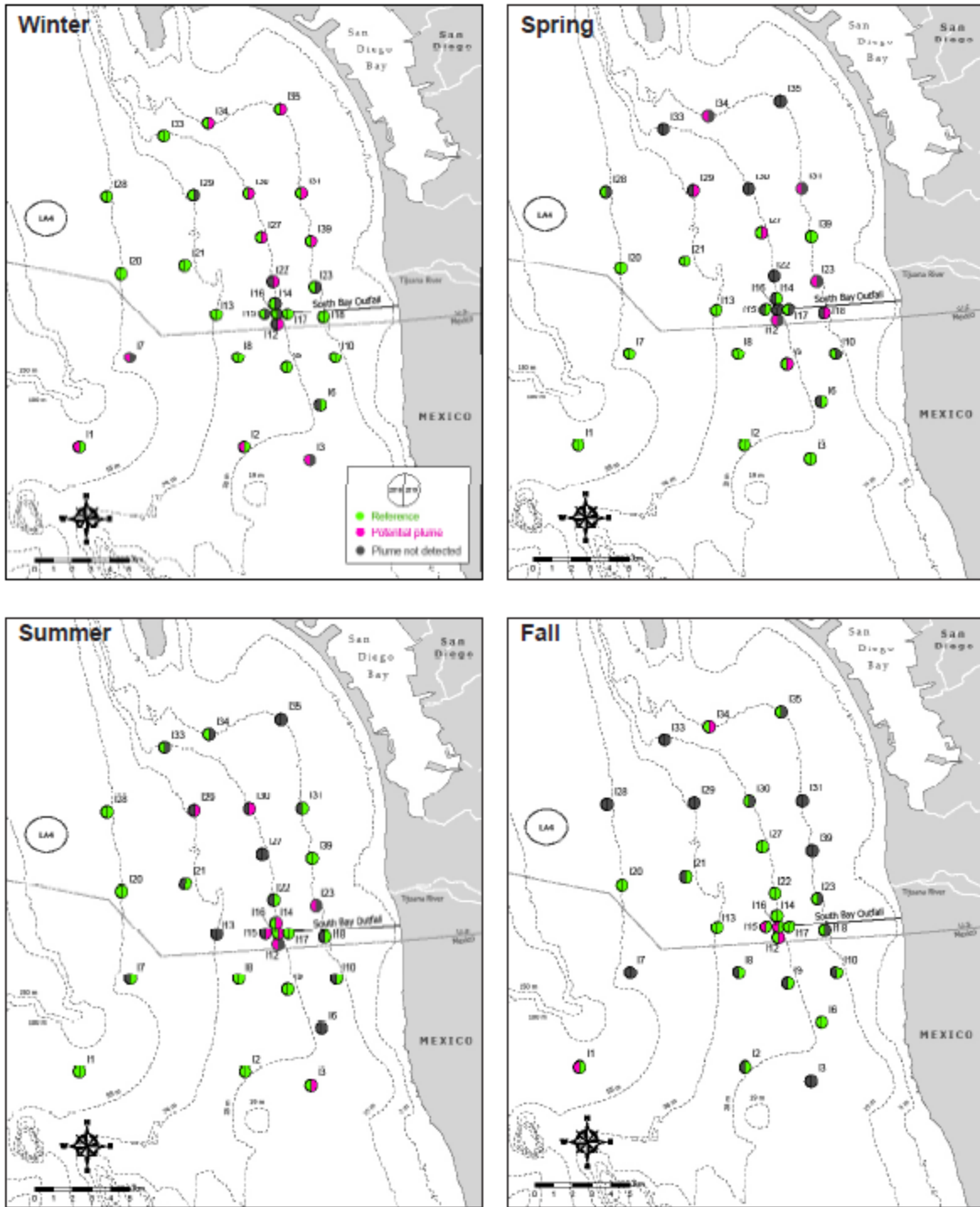


Figure 5. Distribution of stations meeting potential plume criteria (pink) and those used as reference stations (green) near the SBOO during quarterly surveys in 2018 (left half of pie) and 2019 (right half of pie).

The PLOO is located 4.5 miles (7.2 km) offshore at depth of 320 ft (98 m) (Figure 6). It was extended to its present location in 1993, and it is buried in a trench from shore through the surf zone out to a distance of about 2,600 ft (792 m) offshore. Over the next 400 ft (123 m) the pipeline gradually emerges from the rock trench. Beyond 3,000 ft (914 m) offshore, the remainder of the 4.5 mi (7.2 km) pipeline rests on a bed of ballast rock on the sea floor. The end of the pipeline connects to a perforated “Y” diffuser section of two legs, each 2,500 ft (762 m) long. Wastewater is discharged through diffuser ports ranging in depth from 306 ft (93 m) to 320 ft (98 m).

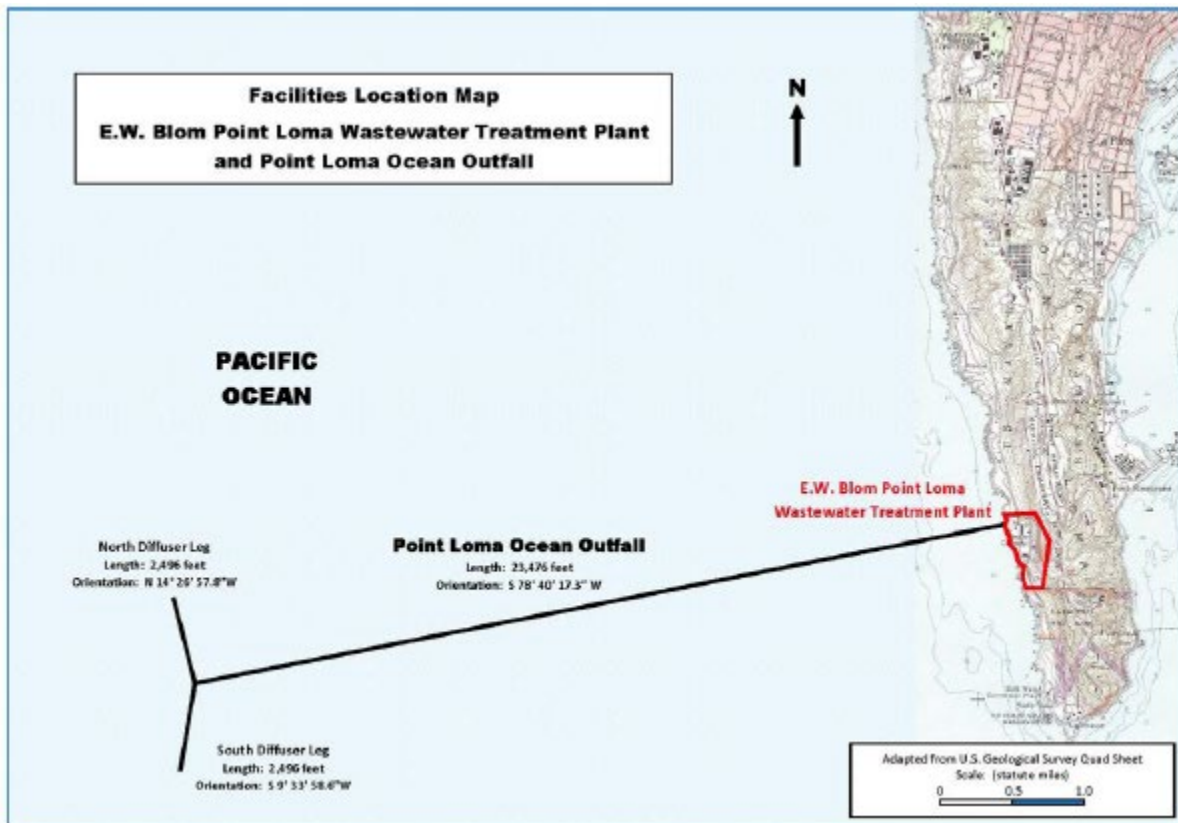


Figure 6. Location of the Point Loma Wastewater Treatment Plan and the PLOO.

The shoreline along Point Loma consists primarily of rocky reef with cobble or sand pocket beaches (City of San Diego 2015a). In the nearshore waters, a large, six-mile long (10 km) kelp bed extends from the tip of Point Loma to the Mission Bay/San Diego River Jetty, from depths of 25 to 90 ft (7.6 to 27 meters) between 0.5 to 1 mile (0.8 – 1.6 km) from shore (City of San Diego 2015a). Beyond the kelp bed, the seafloor gradually slopes downward out to a shelf break at 350 ft (100 meters) depth, beyond which it continues to gradually slope downward to a depth of 1000 ft (305 meters). This shelf area consists primarily of unconsolidated bottom sediments (City of San Diego 2015a). Moreover, for the PLOO, the ZID is estimated to extend 93.5 meters (307 feet) on either side of the diffuser legs and some portions of the action area would likely engender effects than others.

2.4. Environmental Baseline

The “environmental baseline” refers to the condition of the listed species or its designated critical habitat in the action area, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultations, and the impact of State or private actions which are contemporaneous with the consultation in process. The consequences to listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency’s discretion to modify are part of the environmental baseline (50 CFR

As described above, the ESA-listed species that may occur in the action area and be adversely affected by the proposed action are exposed to many similar threats throughout their range. Many of these same threats are also present in the action area (i.e., coastal marine waters off Point Loma from La Jolla to northern Baja California, see Section 2.3 Action Area). Given the large human population and high level of human activity in and around the coastal waters off San Diego, threats such as vessel strikes, disturbance, and habitat degradation also occur in the action area. Although we recognize that many factors affect migratory species during their lifetime, including those occurring outside the action area, we reviewed the stranding records for ESA-listed species within the action area to understand what activities and environmental influences may be affecting these species in the action area. We also review the current state of knowledge on the health of the habitat and environment in the action area, as well as the health of these species and their potential response to environmental and habitat conditions as they enter the action area. This forms a baseline for considering the potential effects of the proposed action from introducing constituents into the environment that may affect ESA-listed species and the quality of the habitat. We also reviewed the ESA consultation record (by conducting a search on NMFS’s Environmental Consultation Organizer and the NOAA Institutional Repository) to identify other Federal actions that have occurred within the action area and have affected ESA-listed species.

2.4.1. Habitat and Environmental Health

The area off Point Loma is characterized by a large, six-mile long (10 km) kelp bed in the nearshore marine environment extending from the tip of Point Loma to the Mission Bay/San Diego River Jetty (City of San Diego 2015a). The kelp bed grows on a pavement-like mudstone/sandstone terrace from depths of about 25 ft (7.6 m) to about 90 ft (27 m) between 0.5 mi (0.8 km) from shore and 1 mi (1.6 km) from shore. Beyond the outer edge of the kelp bed, the seafloor gradually slopes downward to a shelf break at 350 ft, just outside of the 100 m contour line. Beyond the 100 m contour, the seafloor continues to slope downward across the shelf break for another five miles to a depth of 1,000 ft (305 m). This shelf area consists largely of unconsolidated bottom sediments (City of San Diego 2015a).

The action area is within the Southern California Bight, which is influenced by two major oceanic currents: the southward-flowing, cold-water California Current and the northward-flowing, warm-water California Countercurrent. These currents mix in the Southern California Bight and strongly influence patterns of ocean water circulation, sea temperatures, and distributional trends of marine flora and fauna along the southern California coast and Channel Islands (City of San Diego 2014).

Ocean conditions within the action area are affected by both regional- and local-scale currents, including the California Current System, large regional climatic processes (e.g., El Niño Southern Oscillations and Pacific Decadal Oscillations), and local gyres (City of San Diego 2021a). The meandering California Current has a large shoreward component near 32° N, splitting as it approaches the coast, and greatly affects the action area. Recent modeling by Kessouri et al (2020b) confirms previous work (Lynn and Simpson 1987; Mantyla et al. 2008; Rogowski et al. 2012a) by showing that local currents have considerable variability in position, strength, and depth, and can often be coming ashore right toward Pt. Loma. As a result, near-shore currents in the area have been observed by the City (City of San Diego 2016, 2021a, 2021b) variously heading generally north or south directing the effluent towards La Jolla or the Mexican border. Monitoring by the City (City of San Diego 2016, 2021a, 2021b) and Scripps Institution of Oceanography (Rogowski et al. 2012a) also indicates periodic shoreward movement of the PLOO discharge until a depth of approximately 30 meters before turning along-shore.

Seasonal changes in local weather patterns also affect ocean conditions, as the main driver of water column stratification (City of San Diego 2021a). Typically, the dry season from May to September is characterized by relatively warm waters and a more stratified water column, whereas the wet season from October to April is characterized by cooler waters and weaker stratification (Rogowski et al. 2012b, 2013). The transport and distribution of the discharge plume is affected by these oceanographic conditions, as well as more local influences such as outflow plumes from bays, rivers, lagoons, and estuaries; storm water discharge and surface runoff; seasonal upwelling patterns; and variable ocean currents or eddies (City of San Diego 2021a).

The nearshore coastal waters of the action area receive wastes from a variety of human-related sources, such as treated wastewater discharges, stormwater discharges, terrestrial runoff, and outflows from the local rivers and bays (e.g., San Diego River and Bay, Tijuana River) (City of San Diego 2021a). Three wastewater treatment plant (WWTP) facilities discharge into the action area and are sources of pollutants in the offshore environment:

- Point Loma WWTP (the subject of this consultation);
- South Bay International WWTP and South Bay Water Reclamation Plant (WRP) through the SBOO; and
- San Antonio de los Buenos WWTP (in Baja California)

Above facilities that discharge within the U.S. waters are expected to continue to do so into the future under section 402 of the Clean Water Act. For the facilities that discharge into State waters, the State has delegated authority from the EPA to issue the NPDES permits for the

California facilities through the SDRWQCB. The continued discharge of waste and wastewater from these neighboring, non-federally permitted facilities could affect water and sediment quality within the action area. The peak hydraulic capacity from the SBOO is 233 MGD with a total average design capacity of 174 MGD, although the dry weather discharge is typically much less (SDRWQCB 2021). The dry weather discharge primarily consists of effluent from the South Bay International WWTP, as the South Bay WRP recycles most of its tertiary treated water (EPA 2017a). The South Bay International WWTP is owned by the International Boundary Commission (a Federal entity) and treats 25 MGD from the Tijuana River, Mexico². Plans are in development to expand the South Bay International WWTP to 50 MGD, to receive flow that is currently going to the San Antonio de los Buenos WWTP, allowing that to be shut down. The South Bay WRP is permitted to discharge up to 15 MGD (SDRWQCB 2021). Although not a WWTP, University of California, San Diego – Scripps Institution of Oceanography also discharges aquaria wastewater, sea water, filter backwash water, stormwater, and urban runoff to the Pacific ocean. (SDRWQCB 2019).

As described in Section 1.3.1 above, the Point Loma WWTP receives wastewater flows from more than 2.2 million residents and industries in an area encompassing approximately 450 mi² (1,165 km²). The Point Loma WWTP has been in operation since 1963 and originally discharged through an outfall approximately 3.9 km offshore at a depth of about 60 m (City of San Diego 2021a). In the early 1990s, the outfall was extended an additional 3.3 km offshore to its current distance of approximately 4.5 miles (7.2 km) offshore at a depth of 310 ft (94 m). The Point Loma WWTP was upgraded from primary treatment to advanced primary treatment between mid-1985 and July 1986 (City of San Diego 2021a). Treatment processes include: mechanical self-cleaning climber screens; chemical addition to enhance settling; aerated grit removal; sedimentation basins with sludge and scum removal facilities; and effluent disinfection facilities providing chlorination in the effluent channel (EPA 2017a).

Regarding sediment quality, the 2018 Bight Regional Monitoring Program results indicate that multiple sources contribute to pollutants found in the sediment. Past discharge practices (now discontinued) have resulted in detectable levels of DDT and PCBs in almost all bottom-dwelling fish populations throughout southern California (SCCWRP 2012). Highest concentrations of DDT and PCBs are on or near the Palos Verdes shelf off Whites Point in Los Angeles, with fish tissue burdens declining to the north and south across the Southern California Bight. Concentrations of chlorinated hydrocarbons in fish from reference areas are now less than 5% of levels measured two decades ago (Allen et al. 2011). Contaminant burdens in fish tissues (e.g., PCBs, DDTs, and chlorinated hydrocarbons) at Point Loma are comparable to those at reference sites (City of San Diego 2015b, 2021b, 2021a).

2.4.1.1 Water Quality in the Action Area

As described above in section 1.3.2 of the Proposed Action, the City evaluates effluent quality based on TSS, BOD, chlorine, oil and grease, settleable solids, and aldrin. Additional pollutants are also monitored in the effluent but are used only for informational purposes and not for

² https://www.ibwc.gov/mission_operations/sbiwtp.html

compliance determinations. Despite interannual variability, actual mass emissions of total suspended solids have declined steadily since 1995. The permit includes effluent limits for biological oxygen demand (e.g., annual average system-wide percent removal of at least 58%) and total suspended solids (e.g., monthly average system-wide percent removal of at least 80%), which have both been met since 2008.

In the action area, a number of factors influence water quality, including point source discharges (e.g., from WWTPs), regional non-point source discharges, local river outflows, and other local non-point sources (e.g., harbors, marinas, storm drains, and urban runoff) (City of San Diego 2015a). The San Diego Region contains several areas listed as impaired water bodies under section 303(d) of the Clean Water Act, largely due to bacteria indicators. Areas in the vicinity of the Point Loma Outfall that are designated as impaired bodies include the Point Loma Hydrologic Area (for total coliform) and the Scripps Hydrologic Area (for enterococcus, fecal coliform, and total coliform) (EPA 2017a).

Marine mammals, sea turtles, and abalone that are found off the coast of California can be exposed to relatively high levels of contaminants because they are generally long-lived species that are in close proximity to urban areas with high human activity. Here, we describe the essential elements and metals, POPs, and other CECs that are found in the action area and that adversely affect ESA-listed species. Additional information regarding water quality and potential impacts to habitat and marine life can be found in Section 3.2 (Adverse Effects on Essential Fish Habitat).

2.4.1.1.1 Metals and Ammonia

Metals are naturally found in the environment and some are essential to an animals' nutrition. However, human activities can increase the concentrations and metals can become toxic at certain exposure levels. Most metals settle to the ocean floor where they can accumulate in sediment. The City of San Diego (2019) provided effluent monitoring data for 2017, showing varying concentrations of metals including arsenic, chromium, copper, iron, lead, mercury, silver, and zinc. Data for additional years was not provided and were not available in the latest monitoring reports; therefore, we could not assess trends in the concentrations of metals in the effluent over time. As for metals measured in fish, the City of San Diego (2021a) reported that arsenic and selenium were found to exceed their median international standards for human consumption in $\geq 83\%$ of the muscle tissue samples from sport fish collected in the vicinity of the PLOO. In contrast, all muscle tissue samples had concentrations of mercury below USFDA action limits. Historically, elevated levels of metals have remained uncommon in sport fish collected in the vicinity of the PLOO. Overall, there was no evidence of metal contaminant accumulation in fish collected during the 2019 period in the vicinity of the PLOO that could be associated with wastewater discharge, which is consistent with historical findings.

Ammonia is one of several forms of nitrogen existing in aquatic environments and is toxic to aquatic life at certain concentrations. When ammonia is present in water at high enough levels, aquatic organisms have difficulty excreting the toxicant. This leads to a toxic buildup in internal tissues and blood, and the buildup can cause death. Similarly, excess nutrients can increase plant and algal growth leading to eutrophication (EPA 2017b). In 2017, ammonia nitrogen

concentrations in Point Loma WWTP's effluent averaged 40.3 mg/L, with a maximum recorded value of 44.5 mg/L (City of San Diego 2019). The City of San Diego (2019) stated that monitoring for ammonia nitrogen in the vicinity of the PLOO indicates that after initial dilution, concentrations in the plume are at or near detection levels and virtually indistinguishable from natural background levels. Additional monitoring data for ammonia levels in the effluent, receiving waters, or fish tissue was not provided or available in the latest monitoring reports.

2.4.1.1.2 Persistent Organic Pollutants

POPs (such as PCBs, DDT, and different types of flame retardants) can biomagnify, or accumulate up the food chain to a degree where levels in upper trophic-level species can have significantly higher concentrations than that found in the water column or in lower trophic-level species. PCBs were designed for chemical stability and were historically used in paints and sealants, industrial lubricants and coolants in electrical transformers and capacitors, and flame-retardants. There are potentially 209 congeners, or forms, and the chemical structure will influence the volatility, persistence, and toxicity. For example, the more chlorinated PCB congeners are more persistent in the environment than the less chlorinated congeners (Grant and Ross 2002). PCB congeners that are similar in structure to dioxin are highly toxic and can cause cancer, disruption to the immune system, reproductive impairment, endocrine disruption, and developmental problems (WHO (World Health Organization) 2010). Non-dioxin-like PCB congeners are less acutely toxic; however, researchers have reported that they can interfere with hormone-regulated processes (Bonfeld-Jørgensen et al. 2001; Oh et al. 2007) and enhance developmental neurotoxicity (Fischer et al. 2008) and cytotoxicity (Pellacani et al. 2014).

DDTs were primarily used to control insects in commercial and agricultural areas, forests, homes, and gardens. DDTs are persistent in nature and the food web, biomagnify, and are highly toxic to aquatic organisms. The major metabolites, dichlorodiphenyldichloroethylene (DDE) and Dichlorodiphenyldichloroethane (DDD), are also highly persistent and toxic. Eggshell thinning and reproductive dysfunction was linked to DDT exposure in various bird species (reviewed in (Fry 1995). PCBs and DDTs were banned in the 1970s and 1980s due to their toxicity in humans and wildlife.

Although levels of PCBs and DDTs have dramatically decreased in environmental samples since the mid-1970s (Mearns 1988; Lieberg-Clark et al. 1995; Calambokidis et al. 1999; Rigét et al. 2010; Sericano et al. 2014), these compounds continue to be measured in marine biota around the world. Monitoring data for the Point Loma WWTP effluent show PCB levels below the detection limit and below applicable numeric effluent limits and the CA Ocean Plan's water quality objectives (City of San Diego 2021a). Monitoring data for 2017 also showed that DDT levels in the effluent were below the detection limit (City of San Diego 2019). Concentrations of chlorinated pesticides (e.g., DDT, PCB, and PAHs) in the sediments at Point Loma are generally low, except for DDE (a breakdown product of DDT)(City of San Diego 2015b). DDE levels at Point Loma are within the range of sediment concentrations found elsewhere in the SCB, likely due to historical discharges (City of San Diego 2015b).

As for PCB and DDT levels in fish tissue, the City of San Diego (2021a) reported that PCBs were detected in all muscle tissue samples from the PLOO rig fishing zones in 2019 at concentrations ≤ 10.98 ppb. Four samples had PCB levels in exceedance of the Office of Environmental Health Hazard Assessment (OEHHA) threshold of 3.6 ppb. PCBs were also detected in all fish liver tissue samples from the PLOO trawl zones in 2019, at concentrations ≤ 3193.5 ppb. Historically, PCBs have been detected in 89–100% of the liver tissue samples from Sanddabs, Scorpionfish, and Hornyhead Turbot analyzed since 1995, with total PCB concentrations generally within ranges reported elsewhere in the SCB (e.g., Mearns 1988; LACSD 2020; City of San Diego 2021). The City of San Diego (2021a) also reported that DDTs were detected in all samples collected in the vicinity of the PLOO at concentrations ≤ 531.9 ppb. The City of San Diego (2021a) report that concentrations of PCBs and DDTs have been widely variable over time with most being detected at levels within ranges reported elsewhere in the SCB (e.g., Mearns 1988; LACSD 2020; City of San Diego 2021). Overall, levels of PCBs and DDTs near the PLOO are not higher than at reference sites, and there are no existing warnings, advisories, harvest closures, or restrictions on seafood taken from the Point Loma ocean area regarding PCBs and DDTs (City of San Diego 2015b).

Recent decades have brought rising concern over a list of the so-called “emerging” contaminants and other pollutants, including flame retardants (PBDEs and chlorinated organophosphates). PBDEs and organophosphates flame retardants have been used to protect or enhance the properties of plastics, fabrics, furniture and other materials as well as prevent fire or delay its initiation (Pantelaki and Voutsas 2020). Additive flame-retardants can readily dissociate from the products they are added to and discharge into the environment. Preliminary Bight 2018 data indicate non-detectable to low levels of PBDEs in coastal sediments (EPA 2021). However, due to restrictions on the use of brominated forms of flame retardant such as PBDEs, increased use of alternative flame retardants, such as chlorinated organophosphate esters, has most likely occurred to meet flammability standards for many consumer products, such as mattress pads, furniture, or automobile seating (EPA 2015).

Organophosphate esters is one class of the flame retardants increasingly used as manufacturing additives not only as flame retardants, but also as plasticizers, antifoaming agents, electronic equipment, and various other applications (Lin and Sutton 2018; Pantelaki and Voutsas 2020). There are many compounds classified as organophosphates including triphenylphosphine oxide (TPPO), tri-n-butyl phosphate (TNBP), isopropylated triphenyl phosphate (IPPP), tris(chloroethyl) phosphate (TCEP), tris(chloroisopropyl) phosphate (TCIPP), tris (1,3-dichloro-2-propyl) phosphate (TDCPP), tris(chloroethyl) phosphate (TCEP), and tris (chloropropyl)phosphate (TCPP).

Organophosphates esters have been identified as a growing concern due to their common use, widespread exposure, and potential health hazards (EPA 2015; Lin and Sutton 2018). EPA (2015) has identified three chlorinated organophosphates of particular concern: TCEP, TCPP, and TDCPP. Limited data on TCEP and TCPP are available for the Point Loma WWTP. Data from a single study show that TCPP was not detected but TCEP was detected at a concentration of 0.16 micrograms per liter in Point Loma WWTP’s effluent (Vidal-Dorsch et al. 2014). Additional information is available for other areas, including north along the coast in Santa Monica Bay. In 2013, PBDEs, TCEP, TCPP, and TDCPP were consistently detected in the

effluent for the City of Los Angeles' Hyperion Treatment Plant (Hyperion) (EPA 2017b). In a special study conducted in 2019, PBDEs were detected at low levels in Hyperion's effluent, whereas TCEP, TCPP, and TDCPP were detected in all effluent samples for both Hyperion and the Terminal Island Water Reclamation Plant (TIWRP) (LASAN 2020). Average mass loadings were the highest for TCPP in Hyperion's effluent (approximately five pounds per day) (LASAN 2020). The results were consistent with those of Vidal-Dorsch et al. (2012), who also detected TCPP in all effluent samples from four publicly owned southern California treatment works. Further up the coast in San Francisco Bay, a 2018 monitoring study also detected organophosphate flame retardants (including TCPP) at concentrations comparable to or greater than PBDEs (Lin and Sutton 2018).

Tributyltin (TBT) is a persistent pollutant that has been used as an antifoulant on ships, buoys, nets and piers to restrict or retard growth of fouling organisms. Although it may pose a toxic threat to species, bioaccumulation appears to be less than for other persistent pollutants (e.g., PCBs, DDTs, and PBDEs). Butyltins have been measured in nearshore and in the deeper basins in southern California (Venkatesan et al. 1998). TBT and other butyltins were not detected in 2017 effluent monitoring for Point Loma WWTP (City of San Diego 2019). The NPDES permit includes performance goals and mass emission benchmarks, as well as monthly influent and effluent monitoring for TBT.

2.4.1.1.3 Contaminants of Emerging Concern (CECs)

CECs are a risk to the health of humans and marine life, and the environment in general, given their presence and frequency of occurrence. Although some CECs have unknown sources, effluent discharged from WWTPs can be a major source of CECs to the receiving waters. CECs include:

- POPs such as flame retardants mentioned above (PBDEs and organophosphate esters) and other global organic contaminants such as perfluorinated organic acids;
- Pharmaceutical and personal care products (PPCPs), including prescribed drugs (e.g., antidepressants, blood pressure), over-the-counter medications (e.g., ibuprofen), bactericides (e.g., triclosan), sunscreens, synthetic musks;
- Veterinary medicines such as antimicrobials, antibiotics, anti-fungals, growth promoters and hormones;
- Endocrine-disrupting chemicals (EDCs), including estrogen (e.g., 17 α -ethynylestradiol, which also is a PCPP, 17 β -estradiol, testosterone) and androgens (e.g., trenbolone, a veterinary drug), as well as many others (e.g., organochlorine pesticides, alkylphenols) capable of modulating normal hormonal functions and steroidal synthesis in aquatic organisms; and
- Nanomaterials such as carbon nanotubes or nano-scale particulate titanium dioxide, of which little is known about either their environmental fate or effects.

The Point Loma WWTP's effluent has not been routinely monitored for CECs, except for those that are included in the facility's broad category of priority pollutants. Effluent monitoring shows that CECs have been detected in the discharge, including several PPCPs (e.g., ibuprofen,

naproxen), pesticides (e.g., lindane, DEET), hormones (e.g., estradiol, estrone, progesterone, testosterone), and industrial/commercial compounds (e.g., TCEP, benzophenone, bisphenol A, nonylphenol) (City of San Diego 2021c). Mean concentrations varied widely among individual CECs, ranging from not detected to 99 micrograms per liter for PPCPs, from not detected to 0.51 micrograms per liter for pesticides, from not detected to 0.18 micrograms for hormones, and from not detected to 0.9 micrograms per liter for industrial/commercial compounds (City of San Diego 2021c). The City has conducted special studies to evaluate the effects of several of these CECs in the effluent on fish species (City of San Diego 2021c). Levels of PBDE flame retardants in the effluent were not evaluated; however, the City of San Diego (2021c) notes that PBDEs appear to be declining in the environment since being banned from use and WWTPs are no longer considered to be major sources of PBDEs (Dodder et al. 2012). This is consistent with the 2018 Bight Regional Monitoring results that show non detect or low levels of PBDEs in coastal sediment samples (EPA 2021).

Studies suggest that certain PPCPs may also accumulate in marine biota. Synthetic musks and antibacterial chemicals (e.g. Triclosan) have been detected in dolphins and porpoises in coastal waters off Japan and the southeastern United States and in harbor seals off the California coast (Kannan et al. 2005; Nakata 2005; Nakata et al. 2007; Fair et al. 2009). A wider range of PPCPs, including anti-depressants, cholesterol lowering drugs, antihistamines, and drugs affecting blood pressure and cholesterol levels have been detected in tissues of fish from urban areas and sites near WWTPs (Brooks et al. 2005; Ramirez et al. 2009), suggesting possible contamination of prey. As of yet we have no data on concentrations of PPCPs in ESA-listed species or their prey, but they could be a concern because of their widespread occurrence, potential for biomagnification, and biological activity.

In the last few years, microplastics (including microbeads commonly found in personal care products) have been identified as a widespread concern and wastewater effluent has been identified as a source to the marine environment (Talvitie et al. 2015; Ziajahromi et al. 2017). Recent evidence reveals one way microplastics are entering the marine food web is through zooplankton mistaking them for food (Wright et al. 2013; Desforges et al. 2015). Although the impact of microplastics in the food web is largely unknown, chemicals have been found to adsorb to these microplastics, including PCBs and DDT. In fact, some contaminants adsorb to plastics more readily than to sediment, creating an important transport pathway to benthic species (Teuten et al. 2007). Recently, Fossi et al. (2012) detected plastic additives in the blubber of Mediterranean fin whales and suggested these long-lived filter feeders experience chronic exposure to persistent pollutants as a result of microplastic ingestion. Because complete removal from the effluent is not currently possible (Schneiderman 2015), it may be that preventing the source input is the best action to reduce discharge into the aquatic environment (Ziajahromi et al. 2017).

2.4.1.1.4 Harmful Algal Blooms (HABs)

In the SCB, eastern boundary currents drive seasonal upwelling that brings significant amounts of nutrients into shallower coastal waters. In the SCB, there is also a continuous source of nutrients discharged from several WWTPs such as Point Loma, the South Bay Water Reclamation Plant (WRP) and the South Bay International WWTP (jointly through the South

Bay Ocean Outfall or SBOO) and the San Antonio de los Buenos WWTP approximately 5 miles south of the border in Mexico (Howard et al. 2014). Point Loma WWTP alone is permitted to discharge up to 240 MGD of advanced primary treated effluent. In recent years, the average daily discharge has been approximately 140 MGD; this is expected to decrease due to additional water reclamation and recycling over the next few years. The SBOO is permitted to discharge 40 MGD but usually discharges less (~30 MGD) as most of the South Bay WRP flows are reclaimed, while the volume from the San Antonio de los Buenos facility is estimated at 25 MGD.

Nitrogen is the primary nutrient limiting phytoplankton production in coastal waters (Booth 2015) and additions of nitrogen can cause phytoplankton production to increase, potentially reaching levels so high that they become harmful algal blooms (HABs).

HABs in the California Current are most commonly composed of diatoms or dinoflagellates, or a combination of several of these species and the zooplankton that graze upon them (Trainer et al. 2010). There are many known species in the California Current that may develop into HAB levels. The most prevalent in the vicinity of the action area seem to be two diatom groups (*Pseudo-nitzschia delicatissima* group and the *P. seriata* group) based on monitoring data generated by the Southern California Coastal Ocean Observing System (SCCOOS) at Scripps Pier for years 2019-2021 (<https://sccoos.org/harmful-algal-bloom/>). Dinoflagellates such as *Prorocentrum* spp., *Ceratium* spp., *Cochlodinium* spp., and *Dinophysis* spp. are also common. Additional species of dinoflagellates including *Alexandrium* spp., *Cochlodinium* spp., and *Dinophysis* spp have been detected at Scripps Pier at high levels during some parts of the year.

The two diatom groups *P. delicatissima* and *P. seriata* (*P. spp.* when referenced together) produce domoic acid, which has well-documented toxic effects on marine mammals and birds in the SCB and causes amnesiac shellfish poisoning in humans (Trainer et al. 2010). Dinoflagellate species can produce a number of toxins with different effects (Trainer et al. 2010). For example: *L. polyedrum* produces a yessotoxin, a large family of toxins whose potential impacts are being researched; *Dinophysis* and *Prorocentrum* spp. produce okadaic acid and pectenotoxins that cause diarrhetic shellfish poisoning; *Cochlodinium* spp. produce ichthyotoxins; and *Alexandrium* spp. and *Gymnodinium* spp. can produce saxitoxin, which is responsible for Paralytic Shellfish Poisoning (PSP) and fish kills (Gosselin et al. 1989; Lefebvre et al. 2004; Kudela et al. 2010; Trainer et al. 2010; Backer and Miller 2016). Blooms of *Ceratium* spp. and *Akashiwo sanguinea* have been linked to anoxia and the production of hydrogen sulfide (Trainer et al. 2010). Additional information regarding HABs and potential effects on habitat and marine life can be found in Section 3 (Essential Fish Habitat Response).

HAB occurrences appear to be increasing in frequency, duration, size, and severity throughout the SCB and the world (Howard et al. 2012b; Nezlin et al. 2012; Booth 2015). Anderson et al. (2012) notes that there are multiple reasons for this increasing bloom trend. They include natural dispersion of algal species, dispersal via human activities such as ballast water, improved detection of HABs and their toxins, increased aquaculture operations, and stimulation due to cultural eutrophication and climate change.

There is a compelling weight of evidence that nutrients are affecting algal dynamics in the SCB with chronic HAB outbreaks in areas that receive anthropogenic nutrient inputs (Howard et al. 2012b, 2014; Booth 2015). In the past, it was assumed that nitrogen inputs from seasonal upwelling, typically in the spring and early summer months in the SCB, dwarfed the contribution of anthropogenic nitrogen sources. While this is true over the entirety of the SCB at peak upwelling, recent studies challenge this assumption in an important way. Nitrogen inputs from anthropogenic sources, particularly WWTPs, can be substantial and, in some cases, approximately equal to nitrogen inputs from upwelling at the spatial scales relevant to the formation of HABs (Corcoran and Shipe 2011; Howard et al. 2012b, 2017; Booth 2015; Pondella et al. 2016). The largest four WWTPs in the SCB (including the Point Loma facility) accounted for 90% of the total WWTP discharges at the time of these studies (Howard et al. 2014).

Howard et al. (2014) study does not match up perfectly to the action area for this consultation, however, and has important uncertainty in estimating the contribution of nitrogen from upwelling. Howard et al. (2014) analyzed four WWTPs that discharge into the San Diego region, including Point Loma and three small WWTPs (Hale Avenue Resource Recovery Facility, South Bay WRP, and the South Bay International WWTP). The Hale Avenue Resource Recovery facility (~18 MGD/day discharge) is in Escondido, California and discharges out of the action area through a shared outfall in Cardiff, California to the north of the action area. Howard et al. (2014) did not include information on discharges from the San Antonio de los Buenos Wastewater Treatment Plant (~25 MGD/day discharge) in Tijuana, Mexico, which does discharge into the action area and is reported to be in disrepair, resulting in high nutrient loading to the nearshore area.

Howard et al. (2014) also noted that the San Diego region was at the edge of the Regional Oceanic Modeling Systems (ROMS) model boundary, which presents a large amount of uncertainty. In particular, the total contribution of nitrogen from upwelling was likely underestimated in the San Diego region. The wastewater contribution in the study is estimated to be ~3 times higher than the estimated upwelling contribution (7,400 Kg N/km²/year v. 2,400 Kg N/km²/year) which the authors recognized as unlikely. While the information in Howard et al. (2014) is insufficient to precisely assign the nutrient contributions of Point Loma as a percent of the total in the action area, it is evident that the facility's constant contribution of nitrogen to the receiving water is significant and may truly be in the same range as other large WWTP contributions to Santa Monica Bay (~ equal to upwelling) or the San Pedro shelf (~50% of upwelling).

Nitrogen from upwelling is largely in the form of nitrate (98.7%) while nitrogen in effluent is largely ammonium (92%), a reduced form (Howard et al. 2012b, 2014). Concentrations of nitrogen in the effluent plumes are up to three orders of magnitude greater than maximal ambient concentrations and they also entrain deeper, nutrient laden water as they rise into or through the euphotic zone when stratification is weak (Reifel et al. 2013; Seegers et al. 2015). This surfacing most often occurs during the winter months in the SCB. The difference in nitrogen form allows detection of the discharge by tracking the nitrification of the ammonium to nitrate (Booth 2015; McLaughlin et al. 2017). These "hot spots" of ammonium input and nitrification may also be contributing to the overall decline in the dissolved oxygen levels observed in the nearshore regions of the SCB (Booth et al. 2014; Booth 2015; Nezlin et al. 2016) as dissolved oxygen is

used during nitrification. The nutrients are quickly incorporated into the phytoplankton biomass and are not useful as long-term tracers of an effluent plume (Reifel et al. 2013; McLaughlin et al. 2017).

Nezlin et al. (2012) found that all four large WWTPs in the SCB (including Point Loma) had “hot spots” of high offshore chlorophyll-*a* (CHL-*a*), which is indicative of phytoplankton production, and that these conditions occurred throughout most of the year along the south San Diego Region and at the WWTP outfalls. The San Diego area has been identified as a hot spot with longer residence times and higher CHL-*a* levels (Trainer et al. 2010; Nezlin et al. 2012; Smith et al. 2021). Additional nutrients may enter the San Diego region from the south due to the Southern CA Counter Current (Howard et al. 2012b) and from winter runoff.

HAB species are known to persist in the subsurface zone and then be advected into the upper water column during the upwelling season, where the combination of nutrient availability and increasing sunlight may result in a bloom (Trainer et al. 2010; Seegers et al. 2015). WWTP discharges, including discharge from Point Loma, may have the effect of fertilizing or kick-starting the spring time HABs by sustaining or even increasing the duration or population size of HAB species in subsurface water “lenses” associated with the effluent plumes (Trainer et al. 2007; Cochlan et al. 2008; Seeyave et al. 2009; Kudela et al. 2010; Nezlin et al. 2012; Seegers et al. 2015) and by providing nitrogen to the upper water column of the San Diego region when stratification is weak or shallow. These subsurface populations of HAB species can then be uplifted into the surface waters and are a possible explanation for the occurrence of “instant” domoic acid events immediately following upwelling rather than a typical delayed bloom development (Seegers et al. 2015; Smith et al. 2018).

Monitoring of Point Loma’s discharge plume shows the distribution to be primarily alongshore to both the north and south of the outfall (Rogowski et al. 2012a; City of San Diego 2021a) with a periodic inshore component. Previous monitoring shows that ammonia concentrations from the discharge in the vicinity of the PLOO after initial dilution were at or near detection levels and virtually indistinguishable from natural background levels (City of San Diego 2019). This is not surprising given that ammonia levels become more difficult to detect as it is converted to nitrate and as both forms are taken up by phytoplankton. Both forms of nitrogen discharged by the facility, ammonium and nitrate, can support the growth of HAB species. Kudela et al. (2008) showed that ammonium uptake by *A. sanguinea*, a red tide forming dinoflagellate, was approximately threefold higher than uptake of nitrate. Kudela et al. (2010) later showed that *P. spp.* grew equally well or better on reduced nitrogen sources and Howard et al. (2007) showed that *P. australis* could use either nitrate or ammonium simultaneously, a trait that may give it a competitive advantage in areas subjected to wastewater effluent discharges and upwelling.

There are several sources which summarize numerous studies and conclude that reduced forms of nitrogen (ammonium, urea) significantly tilt the phytoplankton community toward the development of HABs (Howard et al. 2012b; Reifel et al. 2013; Booth 2015; Seegers et al. 2015). Schnetzer et al. (2007) cites several studies that examined *P-spp.* and noted that their effective toxicity can be highly variable. These diatom species seem to produce higher levels of domoic acid when under silica or phosphate stress (i.e., the N:P and/or N:Si ratios are higher than or altered from natural conditions) (Anderson et al. 2006; Schnetzer et al. 2013). The

presence of large amounts of nitrogen in the discharge has the effect of unbalancing these ratios at the local level and may be partially responsible for the very potent HABs that have been occurring in the spring for ~10 years in the SCB.

Urea, an organic form of nitrogen that is more commonly found in urban runoff than in WWTP effluent, has been found to produce especially high domoic acid concentrations in *P. australis* (Howard et al. 2007). Urea is a minor component in WWTP discharges in the SCB in contrast to riverine runoff where organic nitrogen forms are much more prevalent (35% of total nitrogen) (Howard et al. 2012b). During wet years, urea entering the San Diego region from the land (e.g., riverine runoff) will be in the less dense, freshwater runoff. This becomes a reservoir of nutrients that influence HAB formation and toxicity when the subsurface species that may be sustained by the PLOO are advected into the upper water column close to shore.

The *P. spp.* are also known to flocculate and form masses large enough to sink to the ocean floor carrying domoic acid with them, which may be ingested by benthic species, thus spreading the toxin within the benthic food web (Schnetzer et al. 2007, 2013; Trainer et al. 2010; Smith et al. 2018, 2021). Rapid transport can be aided by subduction by eddies and there may be a significant topographic influence in the SCB (Kessouri et al. 2020a) leading to benthic hot spots. The SCB 2018 Regional Marine Monitoring Program found widespread domoic acid contamination in the sediments of the SCB with significant detections on the mid-shelf area (67% of this area), although the samples collected in the San Diego region in 2019 were below detection limits (Smith et al. 2021). In all, the toxin was detected in 54% of the SCB shelf habitats sampled. Sediment domoic acid concentrations ranged from 0.57 to 168.0 ng/g sediment over two years of sampling and two different, but similar, detection methods. It is unclear if these concentrations are having direct effects on benthic species in the SCB. Marine worms were found to have high levels of contamination compared to other benthic infauna. This reservoir of domoic acid poses a risk for transfer into the food web, including to ESA-listed species.

As discussed in details in section 2.5.3.2, we anticipate the Point Loma effluent would help encourage HAB formation, and any ESA-listed marine mammals or sea turtles foraging nearby would experience an increased risk of exposure to biotoxins that may impact their health.

2.4.2. Marine Mammals and Sea Turtles

2.4.2.1 Health and Contamination

POPs can be highly lipophilic (i.e., fat soluble) and are primarily stored in the fatty tissues in marine mammals and sea turtles (O’Shea 1999; Aguilar et al. 2002). Therefore, when marine mammals consume contaminated prey, they store the contaminants primarily in their blubber; POPs are stored in the fatty tissues and plasma in sea turtles. POPs can resist metabolic degradation and can remain stored in the blubber or fatty tissues of an individual for extended periods of time. When prey is scarce and when other stressors reduce foraging efficiency, or during times of fasting, a marine mammal metabolizes their blubber lipid stores, causing the pollutants to either become mobilized to other organs or remain in the blubber and become more concentrated (Krahn et al. 2002). Adult females can also transmit large quantities of POPs to

their offspring, particularly during lactation in marine mammals. Mature female sea turtles offload their burdens to their eggs and hatchlings (Van de Merwe et al. 2010; Stewart et al. 2011). The mobilized pollutants can then become bioavailable and may cause adverse health effects. As described above, metals and CECs have widespread occurrence, and some have the potential for biomagnification, and biological activity. However, we have little data on concentration levels in ESA-listed species or their prey of these contaminants. Below, we provide a summary of what information is available on POP levels in ESA-listed marine mammals and sea turtles.

2.4.2.1.1 Marine Mammals

There are numerous studies that have analyzed POPs in marine mammals throughout the world's oceans and throughout the decades (e.g., O'Shea 1999). Here we describe known POP levels that ESA-listed marine mammals have acquired throughout their geographic range, which overlaps with the action area. There are a few studies that have analyzed POPs in marine mammals from the southern California area. These studies have primarily focused on POPs in the blubber of California sea lions, gray whales, humpback whales, northern elephant seals, and harbor seals (Kannan et al. 2004; Elfes et al. 2010; Trumble et al. 2013; Robinson et al. 2013). When no data are available, we describe levels in other populations. This is not an exhaustive literature review, but rather, it provides the general baseline pollutant levels in ESA-listed marine mammals or other marine mammals in the region.

Caution in interpretation should be taken, however, when comparing results among studies and from different populations in different geographic areas. Beyond diet and geographic distribution, there are many factors that influence POP concentrations in an individual, such as age, sex, reproductive history, birth order, body composition, and nutritive condition (Aguilar et al. 1999; Ross et al. 2000; Ylitalo et al. 2001). Methodologies could also vary among studies, which could affect the results and make direct comparisons difficult. Using different sampling techniques, such as biopsy or strandings, will yield different results (Krahn et al. 2001). Because baleen whales make long migrations that are associated with long periods of fasting, fluctuations in the lipid stores occur, which can also affect POP concentrations (Bengtson Nash et al. 2013). Lastly, not all POPs are processed and stored within the same tissues in the body. For example, researchers have suggested that blubber may not be an ideal matrix for examination of organophosphate flame retardant compounds as they are considered a less hydrophobic contaminant compared to other POPs (Sutton et al. 2019).

Only a handful of studies have examined POP levels in baleen whales, and even less is known about POP levels in baleen whales off California. There are more data on POPs in humpback whales than other baleen species in the area. Elfes et al. (2010) compared PCBs, DDTs, and PBDEs, among other POPs, in biopsy samples collected from humpback whales from different feeding areas in the North Pacific and North Atlantic. These feeding areas included the coastal waters off California, Washington, and Alaska, and off the Gulf of Maine. The California feeding group was further divided into the northern and southern regions where the boundary was located at Point Sur.

In general, POP levels were higher in humpback whales from the North Atlantic than whales from the North Pacific (Elfes et al. 2010). However, DDT levels in North Atlantic humpback whales were slightly less than that measured in humpback whales feeding in southern California. DDTs in humpback whales off California were remarkably high, and when compared between two California feeding regions, the whales feeding in the southern region had levels more than six times those measured in whales feeding in northern California. In fact, all POP classes were higher in the blubber of humpback whales off southern California than in other feeding regions in the North Pacific. The authors note this difference was not surprising because this area, which includes the action area, is highly urbanized and impacted by more pollutant inputs (such as wastewater and stormwater) than northern California, and humpback whales demonstrate strong site fidelity to feeding areas.

Humpback whales in Alaskan waters had the lowest concentrations of PCBs, DDTs, and PBDEs compared to that found in the other feeding regions off California and Washington (Elfes et al. 2010). These relatively low levels of POPs in humpback whales are not isolated to the less urbanized waters off Alaska. Stranded juvenile humpback whales in Hawaii had levels that overlapped with the lower end of that found in humpbacks from Alaska (Bachman et al. 2014), which is to be expected given that a large proportion of humpbacks that breed in Hawaii forage off of Alaska. Furthermore, Dorneles et al. (2015) measured POPs in humpbacks from the southern hemisphere (Antarctic Peninsula) and found concentrations were lower than that described in humpbacks from the Northern hemisphere.

Unlike the region-specific POP concentrations found in humpback whales, gray whales appeared to have more of a homogenous POP profile. Dead beached gray whales from Alaska, Washington, and California were analyzed for contaminants (Varanasi et al. 1993). They found no evidence of region-specific differences in the POP concentrations among these gray whales, which is likely due to the fact that gray whales share common migration routes and foraging areas across the population. Between 1996 and 1998, 38 gray whales were biopsy sampled in the coastal waters of Washington (Krahn et al. 2001). Unlike in other species, these whales had higher mean PCBs levels compared to their DDT body burdens (2,100 ng/g lipid and 1,200 ng/g lipid, respectively). When comparing POPs in gray whales and humpback whales off Washington, gray whales had substantially higher PCBs than humpback whales, but slightly less DDTs. Humpback whales in southern California had substantially higher DDT concentrations (4,900 ng/g lipid) than that measured in gray whales (Krahn et al. 2001; Elfes et al. 2010).

Very little data are available for fin and blue whales. Blubber of fin whales off Iceland were measured for PCBs and DDTs (Borrell 1993). Similar to humpback whales, fin whales had relatively low POP levels compared to the toothed whales, largely reflecting their lower trophic level feeding behavior. POPs can also be measured in other matrices besides blubber. Earwax accumulates in some whale species throughout their lives and can be used to measure POPs. For example, POPs were measured in the earwax of a blue whale (Trumble et al. 2013; Robinson et al. 2013). The blue whale earplug was harvested after a ship strike off California. Although we cannot directly compare concentrations in these studies to those that measure POPs in the blubber of whales, it can reveal POP profiles or patterns. Similar to that measured in other species off California, the highest measured POP in the blue whale was that of a DDT metabolite (Robinson et al. 2013).

Although POP levels in baleen whales are lower than levels found in upper trophic level species (such as in toothed whales), the PCB and DDT levels found in humpback whales feeding in southern California and Gulf of Maine (Elfes et al. 2010) were already high enough to warrant further attention. Some individuals had PCB levels at or near the health effects threshold level identified for marine mammals (17,000 ng/g lipid) (Ross et al. 1996; Kannan et al. 2000). These biopsy samples that were at or near the PCB health effects threshold were collected in 2003 and 2004. It is likely that more individuals in this feeding group are currently at or above this threshold as they have accumulated more of these persistent pollutants since that time. Furthermore, previous work had revealed lower reproductive rates in humpbacks that feed off California compared to humpbacks that feed in other North Pacific regions (Steiger and Calambokidis 2000). These elevated levels in humpback whales feeding off southern California waters may be a potential causal factor for these lower reproductive rates (Steiger and Calambokidis 2000; Elfes et al. 2010).

Currently, POP levels in Guadalupe fur seals off California are not known. California sea lions generally share some of the migration habits and patterns as Guadalupe fur seals, and California sea lions eat a variety of prey species similar to that of Guadalupe fur seals. For these reasons, we examine POP levels in California sea lions as a proxy for potential contamination in Guadalupe fur seals. As expected, levels of PCBs and DDTs in dead California sea lions sampled in 2000 were higher than that found in humpback whales and gray whales (Kannan et al. 2004). However, a wide range in pollutant values was found. For example, concentrations of DDTs ranged from 4,100 to 1,400,000 ng/g lipid with no significant difference in mean DDT in animals from southern, central, and northern California (Kannan et al. 2004). Mean PCBs (44,000 ng/g lipid) were three-fold lower than mean DDTs; however, PCBs in California sea lions from southern California were the lowest (17,900 ng/g lipid) although still at the health effects threshold established for PCBs in marine mammals (Ross et al. 1996; Kannan et al. 2000). More recently, Randhawa et al. (2015) examined PCBs and DDTs in California sea lions sampled between 1992 and 2007. For animals that had higher summed PCBs and DDTs, their risk for cancer was eight and six times, respectively, compared to animals with lower levels (Randhawa et al. 2015). Fatal infectious diseases were also more likely in animals with higher body burdens.

Recent studies have reported the emerging presence of organophosphates in biota at different levels of the food web chain (from phytoplankton to invertebrates and fish, up to birds and marine mammals) in urban and industrial sites as well as in remote areas (Pantelaki and Voutsas 2020). Cetacean species that have been identified as carrying organophosphate loads include common dolphins in Spain (Sala et al. 2019), harbor porpoises in the United Kingdom (Papachlimitzou et al. 2015), as well as three dolphin species from South Africa: long-beaked common dolphin, Indian Ocean humpback dolphin, and Indo-Pacific bottlenose dolphin (Aznar-Alemany et al. 2019). Numerous organophosphate flame retardants (e.g., diphenylcresyl phosphate (DCP), triphenyl phosphate (TPP), TPPO, TNBP, IPPP) were also detected in muscle tissues of North Atlantic fin whales as well as their prey in Iceland (Garcia-Garin et al. 2020). Pinnipeds have also been detected carrying organophosphate loads, including ringed and harbor seals from Norway (Hallanger et al. 2015). Additionally, four organophosphate flame retardants (triphenyl phosphate (TPhP), TCPP, TDCPP, TCEP) were detected in blubber of harbor seals near San Francisco Bay (Sutton et al. 2019).

Currently, butyltin concentrations in the ESA-listed species in the action area are not well known, and the extent of current contamination relative to effect thresholds is unknown. The distribution of TBT in the tissues and organs of marine mammals is similar to that of other species, primarily higher in the liver and kidneys and lower in the muscles and blubber (Iwata et al. 1997; Tanabe 1999). Cetaceans distributed near more developed nations have elevated TBT levels compared to cetaceans adjacent to less developed nations (Tanabe et al. 1998). Therefore, it is likely that the ESA-listed marine mammals that may occur in the action area have relatively high TBT concentrations compared to marine mammals in less industrialized regions. Butyltin concentrations in cetaceans off the coasts of Japan and USA are similar. Transplacental transfer of TBT from mother to fetus is relatively low compared to other persistent pollutants. For example, TBT concentrations in the liver of a pregnant female killer whale (150 nanogram per gram wet) was much higher compared to concentrations in the liver of the fetus (26 nanogram per gram wet) (Tanabe et al. 1998). TBTs do not appear to differ between males and females, however, increasing levels have been observed in immature stages of Risso's dolphins (Tanabe 1999).

2.4.2.1.2 Sea Turtles

Although less attention has been paid historically to contaminant levels in sea turtles than marine mammals, there are a few studies that have reported POPs in sea turtles and sea turtle eggs around the world (e.g., Swarthout et al. 2010; van de Merwe et al. 2010; D'Ilio et al. 2011). Similar to that found in other species, maternal transfer of POPs was documented with significant correlations between green and leatherback sea turtle maternal blood and eggs, eggs and hatchling blood, as well as between maternal blood and hatchling blood (Van de Merwe et al. 2010; Stewart et al. 2011). Green sea turtles, loggerhead sea turtles, and hawksbill sea turtles in Japan were measured for PBDEs, PCBs, DDTs, and other organochlorine compounds between 1998 and 2006 (Malarvannan et al. 2011). When comparing PBDE levels among the three turtle species, green turtles had the lowest POP levels and decreasing concentrations were associated with increasing carapace length. Because green sea turtles are omnivores, we would expect them to have lower contaminant loads than other species that feed higher on the food chain. Interestingly, this was not observed in green sea turtles in Southern California. Blood and tissue from green sea turtles in Southern California, including the Long Beach area and in San Diego Bay, were sampled for trace metals, mercury, and POPs (Komoroske et al. 2011; Barraza et al. 2019, 2020). They observed higher plasma levels of several POPs in the green sea turtles than that documented in carnivorous and omnivorous turtles from other areas. However, direct comparison among results with different studies is difficult to interpret because of varying methodologies and sample sizes. In total, studies indicate that POPs may competitively inhibit sex hormone binding, affect hatchling survival, and impair immune function through depressed or increased white blood cell count in sea turtles (Keller and McClellan-Green 2004; Van de Merwe et al. 2010; Stewart et al. 2011; Komoroske et al. 2011; De Andrés et al. 2016).

These relatively higher levels of POPs off California are likely the result of higher concentrations in sediment and biota in the region. Different chemical signatures between populations or groups within a population highlight the influence of foraging locations on exposure. In loggerhead sea turtles off Florida, POP profiles in the blood plasma revealed some loggerheads migrate up and

down along the coast, whereas others remain resident (Ragland et al. 2011). Komoroske et al. (2011) and Barraza et al. (2020) also suggests the higher concentrations in Southern California green sea turtles may also be attributed to potential increased foraging rates as a result from elevated temperatures from power plant discharge, ultimately increasing consumption rates and thus elevated exposure risk. Furthermore, the high DDE levels in San Diego green sea turtles may suggest immunological effects because the levels exceeded lymphocyte proliferation no-effect levels established for loggerheads (Keller et al. 2006). Little is known about contaminant levels in olive ridley and leatherback sea turtles.

2.4.2.2 Strandings

Strandings of ESA-listed marine mammals and turtles have been documented within and vicinity of the action area (NMFS WCR unpublished stranding data). From 2016-2020, the following total number of strandings were documented within the action area (and immediate surrounding area) for each ESA-listed marine mammal species: one humpback whale (it is unknown if this humpback whale belongs to the Mexico or the Central America DPS) and seven Guadalupe fur seals. The cause of many of these strandings is unknown. The one humpback whale appeared to strand due to a vessel strike, and four of the seven Guadalupe fur seals appeared to be suffering from illness and/or malnutrition.

From 2016-2020, the following total number of strandings were documented within the action area (and immediate surrounding area) for each ESA-listed sea turtle species: 55 green turtles, 3 loggerheads, 1 olive ridley, and 3 unidentified sea turtles.³ Of the 55 green turtles, 43 and eight turtles were documented in San Diego Bay and Mission Bay, respectively, both of which are located adjacent to the action area. Twenty-one of the green turtles were struck by vessels, and 5 interacted with recreational and/or commercial fishing gear, including one that had a monofilament line protruding from the turtle's mouth that continued into and out of their gastrointestinal (GI) system. One loggerhead appeared to have been struck by a vessel. The cause of strandings of the remaining sea turtles is unknown.

2.4.3. Abalone

2.4.3.1 Monitoring of Wild Populations and Outplanting Efforts

Comprehensive surveys for white abalone have not yet been conducted throughout the coast off San Diego. Most survey efforts in recent years have concentrated on the area off La Jolla and Point Loma. Since 2010, several surveys have been conducted and have identified several white abalone in these areas (Neuman et al. 2015). In 2017, four of these white abalone individuals were collected and brought into captivity to serve as broodstock for the white abalone captive breeding program, under the Scientific Research and Enhancement Permit 14344-2R issued by NMFS to the Bodega Marine Laboratory (BML) under ESA Section 10(a)(1)(A). These animals were deemed as “singletons” with a low likelihood of reproducing in the wild given their

³ One hawksbill sea turtle stranded in the area during this time, which we consider an anomalous event based on their normal distribution outside the action area.

distance (>10 m) from other white abalone (NMFS 2016). The collection of these individuals to serve as broodstock has the potential to enhance the genetic diversity and productivity of the captive breeding program, to support species recovery.

Since 2019, NMFS has outplanted more than 4,200 captive-bred white abalone at two sites, including on site within the action area, as part of experimental outplanting studies (NMFS 2021a). Subsequent monitoring of the outplant sites is being conducted through dive surveys as well as time-lapse cameras to assess the survival and behavior of the outplanted abalone and the presence of wild adult white abalone.

2.4.3.2 Impacts of Harvest and Other Factors

Factors affecting white abalone within the action area include past fisheries harvest, disease, HABs, and climate change impacts (discussed above).

Limited information is available on the historical presence of white abalone within the action area and the effects of past fisheries harvest on their populations. Commercial landings of white abalone (by weight in shell) in the region from Palos Verdes to Mexico made up only 1.42% of the total white abalone landings in California for the period from 1955 – 1993 (Hobday and Tegner 2000). Past fisheries harvest likely reduced the abundance and density of white abalone within the action area, but we do not have information to evaluate to what extent.

Withering syndrome is a disease caused by a pathogen that infects the gut tissue of abalone and inhibits the animal's ability to digest food. As a result, the animal becomes lethargic, unable to hold onto the substrate as its foot muscle shrinks, and typically dies within a few weeks of exhibiting symptoms. White abalone are known to be susceptible to the disease based on laboratory studies and observations of captive animals (NMFS 2008). One of the animals recently collected from the wild had low levels of infection with the pathogen (Moore 2017); however, white abalone in the wild have not been observed exhibiting symptoms of the disease. Thus, the effects of the disease on wild white abalone are uncertain.

HABs have been linked to abalone mortality events along the California coast. In 2007, a die-off of red abalone at the Monterey Bay Abalone Farm was linked to a bloom of the dinoflagellate *Cochlodinium* (Wilkins 2013). In 2011, a die-off of red abalone and other invertebrate species off Sonoma County was linked to a yessotoxin produced by dinoflagellates in the *Gonyaulax spinifera* species complex (Rogers-Bennett et al. 2012; De Wit et al. 2014a). Mortality was observed at all depths surveyed (0 to 20 m), with greater rates of mortality at shallower sites (De Wit et al. 2014a). We do not know of documented abalone mortality events within the action area that have been linked to HABs. However, poor water quality resulting from the red tide event in 2020 was likely the cause behind the death of one captive black abalone held at the SWFSC La Jolla lab (SWFSC 2021). This indicates that HABs may have affected abalone survival in the action area historically.

2.4.4. Other Federal Actions

We reviewed the ESA consultation record and identified other Federal actions that have occurred within the action area and have affected ESA-listed species. These actions include:

- Monitoring and outplanting of white abalone at subtidal reefs within the action area, to enhance wild populations to self-sustaining levels (NMFS 2019c).
- Fisheries research surveys conducted by the SWFSC throughout the North American west coast, typically from ship-based platforms, with potential lethal and sublethal effects on several ESA-listed sea turtle and fish species from capture or entanglement (NMFS 2020b).
- The Navy’s military readiness training and testing (including research, development, testing, and evaluation) activities in the Hawaii-Southern California Training and Testing Study Area, which includes the action area (NMFS 2018a). Training and testing activities include the use of active sonar and explosives that can adversely affect several ESA-listed marine mammals and sea turtles. Effects include harassment and harm from impulsive and non-impulsive acoustic stressors, as well as mortality and non-lethal injuries resulting from vessel strikes.
- Uniform National Discharge Standards for vessels of the Armed Forces operating in U.S. waters (NMFS 2019d). Discharges conducted under these Standards are expected to affect water quality and exposure of ESA-listed marine mammals, sea turtles, and fish species to contaminants, which could result in non-lethal harm or injury.

2.5. Effects of the Action

Under the ESA, “effects of the action” are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action (see 50 CFR 402.02). A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action (see 50 CFR 402.17). In our analysis, which describes the effects of the proposed action, we considered the factors set forth in 50 CFR 402.17(a) and (b).

For the Effects Analysis, we have identified the following potential effects associated with the discharge of wastewater at Point Loma Wastewater Treatment Plant:

- Toxicity associated with exposure to the discharge plume constituents such as metals and ammonia;
- Accumulation of other contaminants that may persist, be potentially harmful in low amounts, or otherwise emerging as concerns for marine life;
- Exposure to environmental conditions created by the discharge of nutrients, including increased instances of harmful algal blooms.

In Section 3, the Essential Fish Habitat Effects Analysis generally describes and summarizes the impacts that wastewater discharge can have on the environment and ecosystem.

In order to evaluate the potential exposure of ESA-listed species to the proposed action, we consider the presence of ESA-listed species within the action area and the potential exposure of these species to the effects of the discharge (e.g., ZID, discharge plume, and the physical, chemical, or biological effects of the discharge). Potential pathways of exposure for ESA-listed species to effects from Point Loma's effluent discharge include: (1) uptake of pollutants from the water; (2) ingestion of prey that have accumulated pollutants; and (3) exposure to harmful algal blooms resulting from the discharge effects. Then, we evaluate how ESA-listed species may respond to this exposure and how their responses may reduce the fitness of individuals of the affected populations. If a potential reduction in individual fitness is expected, then we consider how the effects on individual reproductive development, growth, and survival may affect the population's growth, reproductive potential, and survival. We also evaluate how these effects may affect the population's recovery potential considering the importance of the population to the species' survival and recovery, as appropriate.

We describe most of the Effects Analysis in general across the ESA-listed marine mammal and sea turtle species groups, due to the overall similarity in how the species are generally exposed to the proposed action at an individual and population level, based on similar long lived and migratory life histories. Following the general synthesis of our understanding of how the proposed action may affect ESA-listed species groups, we consider species-specific information where appropriate and necessary to help describe the potential effects of the proposed action. However, this biological opinion does not include an adverse modification analysis because the the action area does not overlap with any designated critical habitat

2.5.1. Exposure and Response to the Toxicity of Point Loma's Effluent

2.5.1.1 Species Occurrence and Exposure

To evaluate the presence of ESA-listed species in the action area, we considered available scientific, commercial, and public information as well as stranding data to help understand and describe the possible occurrence and exposure of these species to the proposed action. For white abalone, we also considered the distribution of potential habitat that is suitable for white abalone, using habitat as a proxy for the presence of the species where we lacked monitoring data.

2.5.1.1.1 Marine Mammals

Blue, fin, humpback, and gray whales are all generally well-known to be regular visitors to the SCB throughout their lifetimes (juveniles and adults) and are frequently observed transiting or foraging in areas very close to shore, including some within easy sight from land and/or access by recreational boaters, paddlers, etc. Specifically, whale watching companies throughout the SCB are the beneficiaries of the large amount of whale activity occurring in nearshore coastal waters. Individuals of all these whale species are known to visit the coastal areas from La Jolla to Baja California on an annual basis during migrations. Published scientific estimates of cetacean densities on the U.S. west coast (Becker et al. 2020) suggest that the coastal area in California (including the action area) is where densities of blue, fin, and humpback whales can occur in

relatively high proportions under various environmental conditions that occur seasonally and/or during some years.

During their visits to this area, we expect these whales would forage on sardines and anchovies and krill which are all known to occur in the area. Foraging is expected even during visits that may be relatively short, as part of transits during their vast migrations that can cover large areas of the Pacific Ocean. The duration of exposure to the proposed action (duration of visits) for individuals of all species may vary, but are generally expected to range from as little as an hour to several days at a time. Exposure will generally follow seasonal patterns surrounding large-scale migrations, and could occur once per year during a migration, or multiple times for individuals that may be using Southern California waters more regularly or for extended foraging activities.

Gray whale occurrence in the action area is typically associated with biannual migratory transits between summertime foraging grounds in Alaska and winter breeding grounds in Mexico. WNP gray whales that may occur along the U.S. west coast occur in conjunction with the typical gray whale patterns. Especially during the northbound migrations that include mothers and newborn calves, gray whales are frequently observed in and near the Southern California coast each year. The general convention has been that gray whales do not regularly engage in foraging during these migrations, but limited feeding also occurs outside the primary feeding grounds, along their migration route and in some portions of their winter range (Oliver et al. 1983; Nerini 1984; Sánchez-Pacheco et al. 2001). Although the ESA-listed WNP gray whale population are expected to constitute not more than a small fraction of all the gray whales that migrate past and through the action area during a year, the fact that all of those gray whales will pass close to or into the action area makes it highly likely that at least some WNP gray whales will visit the action area during the 5-year period of the proposed action. WNP gray whale exposure is expected to be minimal as the animals would only potentially pass through the action area twice during the biannual migrations for very limited durations lasting no more than a number of hours each time.

As mentioned in Section 2.4 (Environmental Baseline), strandings of humpback whales have occurred in or very near the action area in recent years. As a result, we conclude humpback whale species, specifically the ESA-listed populations of these species, are likely to be in the action area and susceptible to effects associated with the proposed action. As described in Section 2.2 (Rangewide Status of the Species and Critical Habitat), both ESA-listed DPSs of humpback whales are known to be present in California coastal waters and could be expected to occur in the action area occasionally. While we do not expect any individuals to take up extended residence in the action area based on their highly migratory nature, we do expect that some individuals could make numerous or possibly frequent and extended visits to the action area over the course of their relatively long lifetimes. For example, it has been documented that humpback whales have strong site fidelity and individuals feeding in and around the action area will likely return in subsequent years throughout the 5-year period of the proposed action, as is evidenced by variations in patterns of POP accumulation that suggest site fidelity to Southern California (Elfes et al. 2010).

As mentioned in Section 2.4 (Environmental Baseline), Guadalupe fur seal strandings have been documented in or very near the action area, and there has been an increase in strandings of Guadalupe fur seals along the California coast in recent years. These increased strandings began at the start of 2015 and are concurrent with the 2015-2021 Guadalupe fur seal UME. This recent stranding data indicates that Guadalupe fur seals are found in coastal California waters, and we anticipate they are likely to be in the action area and susceptible to effects associated with the proposed action. While we do not have any information that suggests any individuals from this species take up extended residence specifically within the action area, we do expect that individuals could make numerous or possibly frequent and extended visits to the area over the course of relatively long lifetimes. The duration of exposure to the proposed action generally can be expected to range from as little as an hour to several days at a time and could include multiple times for individuals that may use Southern California waters more regularly or for extended foraging activities.

2.5.1.1.2 Sea Turtles

From stranding data, anecdotal sightings, and scientific studies, we know that juvenile and adult green, leatherback, and olive ridley sea turtles occur at least occasionally in the SCB, as do juvenile loggerhead sea turtles. As mentioned in Section 2.4 (Environmental Baseline), strandings of green, loggerhead, and olive ridleys have occurred recently in or very near the action area. Although the SCB is not known to be a persistent or primary foraging or nesting location for leatherbacks, loggerheads, or olive ridley sea turtles, the pelagic ecology of these species occasionally does lead them to migrate through the SCB and potentially into the action area.

Specifically, in La Jolla Shores located along the coast of San Diego County in the vicinity of the action area, green turtles were documented to be using that areas as a foraging site that hosts a small resident aggregation of green turtles based on opportunist sightings (Hanna et al. 2021). Moreover, San Diego Bay located in the vicinity of the action area, is an important foraging and resting area for East Pacific DPS green turtles along the U.S. west coast, with the shallow waters of San Diego Bay providing valuable food resources such as marine algae and seagrass (Hanna et al. 2021; NMFS 2021b). Eguchi et al (2020) reported that the San Diego Bay support as many as 60 green turtles with continuous recruitment of both juveniles and adults. We expect that individual green turtles will make numerous or possibly frequent and extended foraging visits to the action area over the course of relatively long lifetimes of extensive migrations or residence in and around the San Diego and La Jolla area, and that the duration of exposure could last up to many days, weeks, or even months at a time or more.

While we do not have any information that suggests any individual turtles (other than green sea turtle) would take up extended residence specifically within the action area, we do expect that individuals could make numerous or possibly frequent and extended visits to the area over the course of their relatively long lifetimes and migrations in the SCB. As a result, the duration of exposure to the proposed action for individuals of all species may vary, but can generally be expected to range from as little as an hour to several days at a time. Exposure for all such animals would generally follow seasonal patterns surrounding large-scale migrations, and could

occur once per year during a migration, or multiple times for individuals that may be utilizing Southern California waters more regularly or for extended foraging activities.

2.5.1.1.3 Overlap with effects of the discharge – marine mammals and sea turtles

The City of San Diego (2018, 2021c) reported that the effluent plume discharges at a depth of 98 meters and rises in the water column to an average top of the waste field of 40-60 meters deep. The maximum height-of-rise to the top of the wastefield during a month varies from about 50 to 64 meters above the bottom, corresponding to water depths of about 30 to 44 meters (EPA 2017c). Monitoring by the Scripps Institution of Oceanography (Rogowski et al. 2012) detected the plume at a depth of 35 meters at its shallowest. Monitoring from the City of San Diego shows that the plume moves north and/or south alongshore and occasionally shoreward although it has not been detected in shallow waters close to shore (City of San Diego 2016, 2018). When moving to the south and slightly onshore, the PLOO discharge approaches the area typically associated with and monitored for, the SBOO discharge.

For all marine mammal and sea turtle species, potential occurrence and overlap with the effluent discharge can occur essentially anywhere throughout the action area, as all these species are highly mobile and could occur anywhere in the area. As described in Section 2.3 (Action Area), the ZID that represents the boundary of where concentrated effluent mixes with receiving water and the place where permitted effluent limits and performance goals apply, represents a relatively small portion of the action area. For the PLOO, the ZID is estimated to extend 93.5 meters (307 feet) on either side of the diffuser legs (EPA 2017c). Although the portion of the action area that is expected to be exposed to concentrated effluent is small, it is possible for ESA-listed marine mammals and sea turtles, or their prey or forage, to be exposed to the concentrated effluent within the ZID.

2.5.1.1.4 White Abalone

The available information regarding the abundance and distribution of white abalone in the action area, historically or currently, is limited. The available fishery landings data indicate white abalone were present and harvested from the area between Palos Verdes and Mexico, but do not provide further information on the numbers, sizes, or distribution of white abalone in the region historically (Hobday et al. 2001). Fishery-independent data is also limited. Most white abalone monitoring efforts have focused on offshore banks in the SCB. Within the action area, white abalone-focused surveys have primarily been conducted in the area off La Jolla and Point Loma, where several white abalone have been found since 2010 and where experimental white abalone outplanting studies are currently being conducted (see section 2.4.3 Environmental Baseline). Surveys have not been conducted throughout the action area, particularly at deeper rocky reefs (e.g., at depths of 30-60 meters).

Where monitoring data is limited, we use habitat as a proxy for the likely presence of white abalone. White abalone adults occupy open, low relief rocky reefs or boulder habitat surrounded by sand, within depths of 5 to 60 meters (Hobday and Tegner 2000). Most of the subtidal rocky reef habitat exists in nearshore areas off La Jolla and Point Loma (indicated by rocky reef areas

and areas with hard outcrop/pavement and kelp persistence; Figure 7). In summary, several nearshore rocky reefs occur throughout the action area and may support white abalone, with confirmed white abalone presence at a few sites off La Jolla and Point Loma where wild adults have been found and captive-bred juveniles have been outplanted.

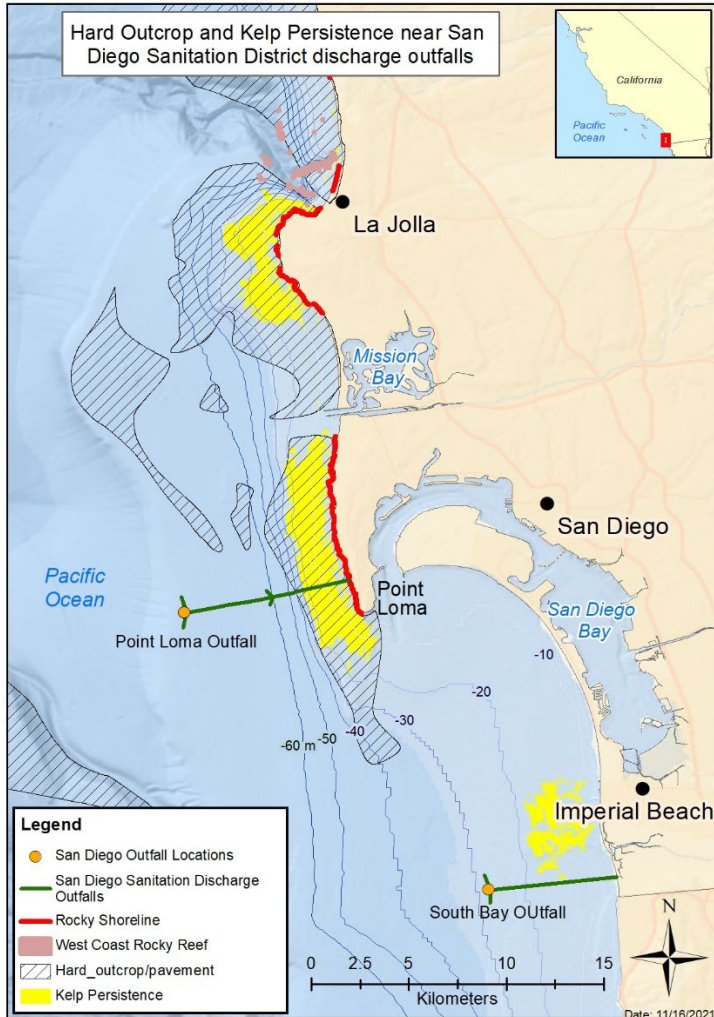


Figure 7. Colored map showing potential white abalone habitat within the action area, primarily consisting of subtidal rocky reefs in nearshore waters (within 20-40 m depth) off La Jolla, the Point Loma peninsula, and Imperial Beach.

2.5.1.1.5 Overlap with effects of the discharge – White abalone

The ZID occurs primarily within deep, soft bottom habitat and does not overlap with potential white abalone habitat. The discharge plume, however, extends throughout the action area, with greater plume probabilities in the vicinity of the outfall and both upcoast/downcoast and lower plume probabilities inshore of the 60 meter contour (see Figure 4 and Figure 5). Based on these plume probabilities, white abalone within the action area may be exposed to the discharge plume to some degree. The plume overlaps with nearshore and offshore rocky habitats where white abalone have been confirmed to occur or may occur, with higher plume concentrations likely

affecting offshore, deeper areas of the reef off Point Loma and lower concentrations in the nearshore reefs off Point Loma and the reefs off La Jolla and Imperial Beach.

In summary, based on the best available information regarding white abalone presence and the extent of the discharge plume and ZID, we conclude that: (a) white abalone off Point Loma and La Jolla may be exposed to constituents from the effluent discharge and their effects; (b) white abalone habitat (and any individuals occurring in that habitat) may also be exposed to the plume and its effects; and (c) because of the sedentary life history of abalone, the risks of exposure are persistent across the entire year.

2.5.1.2 Constituents of Point Loma's Discharge

As described in the NPDES permit, BE/Essential Fish Habitat Assessment (EFHA), and in Section 1.3.2 (Permitted Effluent Limits), effluent limits are implemented for 9 pollutants/parameters: TSS, BOD, chlorine, oil and grease, settleable solids, chronic toxicity, pH, turbidity, and aldrin. Despite interannual variability, actual mass emissions of total suspended solids have declined steadily since 1995. The permit includes effluent limits for biological oxygen demand (e.g., annual average system-wide percent removal of at least 58%) and total suspended solids (e.g., monthly average system-wide percent removal of at least 80%), which have both been met since 2008.

Additional elements and compounds such as metals are also discharged. Other constituents of the discharge that are monitored include metals (arsenic, mercury, and cadmium), PAHs, PCBs, vinyl chloride, endosulfan, and nitrogen (ammonia), and hexachlorocyclohexane. Other constituents of the discharge that are suspected or known to be present in wastewater discharge include: PBDEs, organophosphate esters, TBT; PPCPs including prescribed and over-the-counter medications and numerous other products; and EDCs, including estrogen, androgens, and pesticides. In this Effects Analysis, we consider the potential effect of these pollutants on ESA-listed species.

2.5.1.3 Response of Marine Mammals and Sea Turtles to Exposure to Point Loma's Effluent Plume

For ESA-listed marine mammals and sea turtles, exposure to potentially toxic pollutants from the discharge effluent would primarily occur through the uptake of pollutants from their food sources. In general, direct exposure to constituents such as ammonia and metals in the water column that may occur in the effluent discharge plume of Point Loma does not appear to pose a threat to larger vertebrates that breathe air and have integumentary systems that limit direct uptake from the environment. The permitted effluent limits and performance goals for the proposed permit are set to meet the minimum standards of the California Ocean Plan that have been designed by the EPA and CA SWRCB to protect marine organisms that likely are more immediately or directly sensitive to toxicity from wastewater effluent. Within the ZID, marine mammals and sea turtles could be exposed to relatively higher concentrations of various effluent constituents that are potentially toxic. We cannot precisely estimate exposure times within the ZID for ESA-listed marine mammals and sea turtles given their dynamic movements and

occasional occurrence in the action area, but we anticipate exposure to concentrated effluent in the ZID will be relatively minimal.

Several studies have recorded increased phytoplankton and invertebrates around other offshore wastewater outfalls in Southern California for multiple years (City of Los Angeles 1990; Nezlin et al. 2012; City of Los Angeles Environmental Monitoring Division 2015; EPA 2017b; Kessouri et al. 2021). The increased productivity associated with the effluent plume may attract marine mammal and sea turtle species, which feed on forage fish and invertebrates. This increases the probability of ESA-listed marine mammals and sea turtles foraging in proximity to the outfall and ZID and taking in food sources that may have been exposed to toxic pollutants from the effluent.

The available data indicate that ESA-listed marine mammals and sea turtles are generally not at risk of health effects from most of the compounds or elements (typically metals) measured in Point Loma's effluent. These include ammonia, nickel, silver, and zinc. Some of these compounds are essential elements to nutrition (e.g., nickel and zinc) (Pugh and Becker 2001; Das et al. 2003) and are generally found in low levels in marine mammals and sea turtles distributed throughout the world's oceans (O'Shea 1999; Pugh and Becker 2001). While metals can bioaccumulate in the aquatic environment, most metals (with the exception of methylmercury) do not appear to biomagnify and are regulated and excreted by a host of marine life (Gray 2002). Therefore, limited increases in uptake of these essential elements found in low concentrations in marine mammals and sea turtles are not anticipated to cause adverse health effects for ESA-listed marine mammals and sea turtles. Although silver is not considered an essential element, its toxicity is generally not a concern and it has not been measured often in marine mammals (O'Hara and Becker 2003). Ammonia does not build up in the food chain, and is not anticipated to accumulate in marine mammals and sea turtles.

Other compounds in Point Loma's effluent that may cause adverse health effects but do not appear to biomagnify include: cadmium, chromium, copper, and lead. However, upper trophic-level predators can still accumulate metals even in the absence of biomagnification (Reinfelder et al. 1998). Low levels of arsenic, chromium, copper, and lead have been measured in marine mammal tissues and sea turtles (O'Shea 1999; Saeki et al. 2000; Grant and Ross 2002; Das et al. 2003; Komoroske et al. 2012). Although high cadmium levels are measured in some marine mammals, cadmium is known to combine with metallothionein (a protein molecule) to mitigate the toxic effects (Dietz et al. 1998; Klaassen et al. 2009). Further, no toxic effects of cadmium have been observed in marine mammals or sea turtles to date. Although threshold levels for these metals at which adverse health effects occur are currently unknown for marine mammals and sea turtles, the available data do not indicate that the low levels measured in their tissues pose a health risk (O'Shea 1999). For these reasons, NMFS does not anticipate that ESA-listed marine mammals and sea turtles will experience any toxic health effects associated with most of the potentially toxic compounds and elements found in Point Loma's effluent discharge as a result of occasional exposure to them when foraging in the action area. In the following section (Section 2.5.2 Accumulation of Potentially Harmful Contaminants), we analyze the potential effects associated with other more persistent and/or harmful constituents that may accumulate.

2.5.1.4 Response of White Abalone to Exposure to Point Loma's Effluent Plume

We evaluated how different life stages of white abalone may be affected by exposure to the discharge effluent. Because species-specific information is not available, we used the best available information from studies involving other abalone species. We note that Section 2.5.2 (Accumulation of Potentially Harmful Contaminants) addresses the potential effects associated with other more persistent and/or harmful constituents that may accumulate.

The contaminants identified in the discharge effluent include heavy metals that have been found to have harmful effects on other species of abalone at different life stages, and thus could have harmful effects on white abalone. For example, heavy metals can cause abnormal shell development in larvae at the fertilized egg to veliger stages, depending on the concentrations to which larvae are exposed (Conroy et al. 1996; Gorski and Nugegoda 2006). Increased developmental abnormalities were observed in red abalone (*Haliotis rufescens*) larvae exposed to zinc at concentrations greater than 18 µg/L for 48 hours (Conroy et al. 1996). For blacklip abalone (*H. rubra*) larvae, increased morphological abnormalities were observed after 48-hour exposure to the following heavy metals at or above the indicated concentrations: copper (7µg/L), mercury (21 µg/L), zinc (35 µg/L), iron (4,102 µg/L), cadmium (4,515 µg/L), and lead (5,111 µg/L) (Gorski and Nugegoda 2006).

For juvenile and adult white abalone, several studies have evaluated the effects of water borne and/or dietary exposure of juveniles and adults to copper, zinc, silver, and cadmium. They found that abalone can accumulate these metals in their foot muscle, mantle tissues, and viscera, and experience adverse effects on growth, behavior, and survival (Martin et al. 1977; Liao et al. 2002; Gorski 2006; Huang et al. 2008, 2010; Chen et al. 2011). Martin et al. (1977) exposed adult abalone to concentrations of copper ranging from 10 to 640 µg/L and found 96-hour LD50 values of 65 µg/L for *H. rufescens* and 50 µg/L for *H. cracherodii*, compared to 87 µg/L for *H. rubra* (Gorski 2006). Viant et al. (2002) suggested asphyxial hypoxia and reduced muscle function as possible mechanisms for mortality due to copper exposure. Acute (7-day) and chronic (28-day) toxicity studies exposing *H. diversicolor supertexta* to zinc found reduced growth rates of individuals at 120-125 µg/L and increased mortality at 500-1000 µg/L (Liao et al. 2002; Chen and Liao 2004; Tsai et al. 2004). Chen et al. (2011) modeled the effects of exposure to cadmium and silver on growth and predicted that growth of larvae, juveniles, and adults was inhibited by exposure to levels as low as 10 µg/L Cd and 5 µg/L Ag. In studies exposing *H. diversicolor supertexta* to water borne and dietary silver (5 or 50 µg/L) or cadmium (50 or 500 µg/L), Huang et al. (2010) observed reduced growth and feeding rates in the first few weeks, but similar rates to controls by the end of the 7-week exposure period.

To apply these study results to the Point Loma discharge, we considered the concentrations of these pollutants in the effluent and the concentrations to which white abalone may be exposed. In 2017, cadmium was not detected in Point Loma's discharge effluent, but varying concentrations of copper, mercury, zinc, iron, lead, and silver were detected (City of San Diego 2019). The recorded concentrations were lower than the levels found to kill abalone or cause sublethal effects, except for copper and zinc. The highest recorded values in 2017 for copper (15.3 µg/L) and zinc (41.3 µg/L) exceeded the levels found to cause developmental abnormalities in abalone larvae; however, these values were the concentrations within undiluted effluent. Taking into account the dilution of effluent outside of the ZID, the levels of these metals in the plume are

expected to be well below those documented to kill abalone or sublethal effects on abalone at all life stages. Based on this, we do not expect exposure to the levels of these metals in the plume to negatively affect the development, growth, behavior, or survival of white abalone in the action area.

2.5.2. Accumulation of Potentially Harmful Contaminants

2.5.2.1 POP Loading into the Action Area

POPs are contaminants of concern for ESA-listed species and other marine life because they bioaccumulate, biomagnify, and can be toxic. The legacy organochlorines (e.g. PCBs and DDTs) and the more recent POPs of concern (e.g., PBDEs) have been well documented in the literature to pose a risk to many species. These pollutants are associated with reproductive impairment (Reijnders 1986; Subramanian et al. 1987; Reddy et al. 2001; Schwacke et al. 2002); immunotoxicity (De Swart et al. 1996; Fønnum et al. 2006; Mori et al. 2006); endocrine disruption (Darnerud 2003, 2008; Legler and Brouwer 2003; Legler 2008); neurotoxicity (Darnerud 2003, 2008; Viberg et al. 2003, 2006); and cancer in humans and wildlife (Ylitalo et al. 2005; Bonfeld-Jorgensen et al. 2011). Similar risks may be posed by organophosphate esters: a new generation of flame retardants that is replacing PBDEs and already being detected in areas like California waters and marine waters around the world (Lin and Sutton 2018).

PCBs and DDTs continue to be measured within the SCB, including in the action area and the surrounding region (LACSD 2020). A superfund site for DDT is located on the continental shelf in the vicinity of Palos Verdes; thus, the highest concentrations of DDT were found around this area (SCCWRP 2020; LACSD 2020; EPA 2021) and will most likely be measured throughout SCB for the foreseeable future. PCBs have also been measured throughout SCB. In the early 1980s, the California State Mussel Watch program found that mussels retrieved from the Convair Lagoon within the San Diego Bay had PCB concentrations that exceeded the U.S. Food and Drug Administration's limit for human consumption (San Diego Regional Water Board 2013). Although PCB concentrations declined from 1980 to 2010 in mussel tissue (Melwani et al. 2014), PCBs continue to be detected in sediment and fish collected in the San Diego Bay (Ed Parnell et al. 2008; Neira et al. 2018). Historical discharges of PCBs are also known to persist in sediments due to their long degradation time (LACSD 2020).

The EPA permit includes performance measures for PCBs and DDTs in influent and effluent (sampled once a week) and sediment (sampled twice a year). PCBs and DDTs are also measured in fish tissue annually. As mentioned in section 2.4.1 (Habitat and Environmental Health), the City of San Diego (2021a) reported that PCBs were detected in all muscle and liver tissue samples collected in the vicinity of the PLOO, with some samples having PCB levels in exceedance of the Office of Environmental Health Hazard Assessment (OEHHA) threshold. The potential for exposure to the legacy PCBs and DDT continues to be a concern for ESA-listed species. However, the majority of the exposure likely results from the historical contamination, and it is the persistence of these legacy pollutants that have caused the continuation of effects.

Vidal-Dorsch et al (2014) measured CECs from Point Loma's discharge including flame retardants. PBDEs are being phased out and replaced by the next generation of flame retardants

(i.e., chlorinated organophosphates) (Lin and Sutton 2018), which may have similar toxic effects on ESA-listed species. Moreover, Bekele et al (2019) demonstrated both bioaccumulation and biomagnification of organophosphate flame retardants in the marine food webs. Of particular concern are the following three chlorinated phosphate ester compounds being assessed by EPA (2015) for risks to aquatic organisms and human health: TCEP, TCPP, and TDCPP. ESA-listed species may receive the majority of these pollutants from their diet. To estimate the potential for the proposed action to expose ESA-listed species to these pollutants, it is important to monitor the concentrations of these flame retardants in the effluent and understand how they move through the food web.

To understand and isolate the effects of organophosphate flame retardant loading from Point Loma’s discharge, we estimated the amount of organophosphate flame retardant that is expected to be discharged for the 5 years covered by the proposed NPDES permit. An effluent sample was collected to measure CECs from the PLOO, and TCEP was one of the organophosphate compounds detected (Vidal-Dorsch et al. 2014). However, this study only reported two organophosphates (TCEP and TCPP) and does not allow us to estimate the mass loading of organophosphate flame retardants overall. To estimate the mass loading of the overall organophosphate flame retardants (i.e., combination of numerous organophosphate compounds), we used data from a recent special study at Hyperion and the TIWRP that monitored for these organophosphate flame retardants in the effluent.

As described in Section 2.4.1.1 (Water Quality in the Action Area), organophosphate flame retardants (e.g., TCEP, TCPP, and TDCPP) were detected in all effluent samples for both Hyperion and the TIWRP (LASAN 2020). Average mass loading of the organophosphate flame retardants TCEP, TCPP, and TDCPP combined (mostly TCPP) in Hyperion’s effluent was approximately five pounds per day (Figure 8) (LASAN 2020). This mass loading occurred during a period where the average discharge at Hyperion was approximately 230 MGD (LASAN 2020).

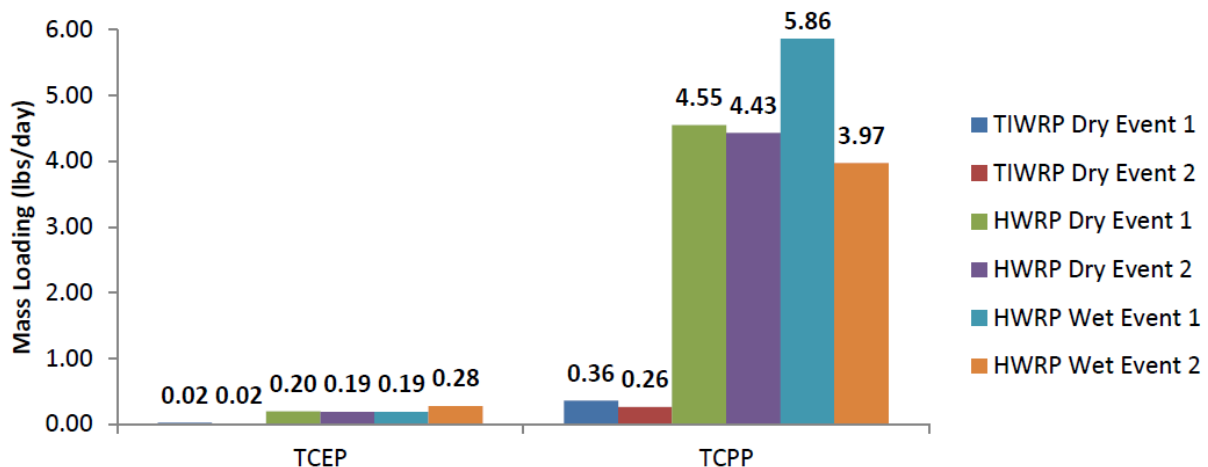


Figure 8. Mass loading calculations of organophosphate flame retardants found in Hyperion and Terminal Island Water Reclamation Plant (TIWRP) effluents (from LASAN 2020).

Point Loma's outfall (i.e., the PLOO) has a capacity of 240 MGD with an average ocean discharge volume of 140 MGD. Because Point Loma and Hyperion are both relatively large WWTP facilities handling domestic, commercial, and industrial wastewater from adjacent highly urbanized areas, we assume that organophosphate flame retardant concentrations and loadings from Point Loma are likely to be relatively equivalent to what has been measured at Hyperion. Therefore, we use the average daily loading for Hyperion (5 lbs/day for 230 MGD) as a surrogate to estimate the loading of organophosphate flame retardants in Point Loma's effluent, using the average flow of 140 MGD and the peak hydraulic capacity of 240 MGD. As a result, we estimate a range of approximately 3 to 5.2 pounds (1.4 – 2.3 kg) per day of organophosphate flame retardants (TCEP, TCPP, and TDCPP combined) may be loaded into the action area as a result of Point Loma's wastewater discharge. This equates to an estimated 1095 to 1898 pounds (511 – 839.5 kg) of organophosphate flame retardants discharged per year. For the total five years of this proposed action, this equates to approximately 5475 to 9490 pounds (2,555 – 4197.5 kg) for the permit cycle.

This estimate of organophosphate loading from the proposed action adds to the long-term accumulation of POPs in the action area that has already occurred from historical discharges under past WWTP operations and other sources.

2.5.2.2 Adverse Marine Mammal and Sea Turtle Health Effects from Exposure to Potentially Harmful Contaminants

Once POPs enter the aquatic system, they readily attach or adsorb to particles (e.g., sediment, dead organic material, plankton, bacteria, microplastics) in the water column rather than dissolving due to the hydrophobic nature of most of these compounds. In general, once the pollutants attach to these particles they may sink down in the water column and accumulate in the sediment, at which point, the sediment acts as a sink and sequesters or buries contaminants rendering the POPs no longer readily available to organisms in the water column. However, the contaminated sediment can act as a source for benthic food webs and begin biomagnifying in the benthic food chain. Not all POPs accumulate in sediment, and some pollutants that enter the aquatic system may directly enter the pelagic food web. The proportional distribution of POPs in the local environment likely varies from site to site based on biotic and abiotic factors.

Recently, researchers from Washington Department of Fish and Wildlife have been tracking the movement of PCBs and other toxic chemicals in Puget Sound, WA, and found comparatively lower levels of these POPs in the sediment, but higher levels in the resident pelagic species. For example, POPs in resident Pacific herring (*Clupea pallasii*) in Puget Sound cannot be predicted by POP levels or trends in the sediment. The three known herring populations in the Puget Sound region reflected different POP patterns, suggesting differential exposure to contaminants and that this difference was related to where these species feed (West et al. 2008). Pacific herring heavily rely on krill, calanoid copepods, and larval invertebrates and fishes. These planktonic species do not have a direct connection to sediment and are likely accumulating POPs directly from the water column (West et al. 2008). These new data and studies from other geographic regions suggest that many of the POPs in the water column do not reach the benthos, but rather are picked up by bacteria or plankton that are then consumed by pelagic organisms, exposing the pelagic food web. This, in addition to deposition in sediments and the benthic food web, is likely

a route by which ESA-listed marine mammals and sea turtles would be exposed to POPs from Point Loma's effluent.

Exposure to some of these contaminants does not need to occur in high concentration to be toxic and has long been recognized as problematic (Carson 1962). Currently, there are no well-developed health effect thresholds for most POPs of concern for marine mammal and sea turtle species. Although it is important to keep in mind that the effects due to POP exposure may potentially be species-specific, dose-dependent, and compound-specific, here we describe toxicology studies that examined the general effects the POP exposurer has had on different species.

POPs such as PCBs, PBDEs, and organophosphate flame retardants are potential endocrine disruptors that can affect thyroid hormone levels, and can cause subtle neurobehavioral effects and reproductive effects in numerous species—both *in vivo* and *in vitro* (Legler and Brouwer 2003; Hall et al. 2003; Darnerud 2008; Legler 2008; Kodavanti et al. 2010; Wang et al. 2020). For example, some POP metabolites are structurally similar to thyroid hormones and these metabolites have disrupted the thyroid hormone homeostasis in laboratory species (Zhou et al. 2001, 2002; Richardson et al. 2008). This type of disruption in thyroid homeostasis is concerning because it can cause developmental neurotoxicity, alter gene expression, reduce the transfer of retinol and T4 (a thyroid hormone) to target organs, and decrease the availability of progesterone (Meerts et al. 2000; Houde et al. 2005; Boas et al. 2006).

Endocrine disruptors can mimic or offset reproductive processes. Consequently, adverse reproductive effects have been associated with POP exposure. Exposure to the congener BDE-99 demonstrated behavioral feminization, permanently impairing spermatogenesis (including reductions in sperm and spermatid counts and smaller testes), and the delay in the onset of puberty and a reduction in the number of ovarian follicles in laboratory species (Hany et al. 1999; Kuriyama et al. 2005; Lilienthal et al. 2006). Some of these exposures were with low doses of POPs, and they caused permanent adverse effects on reproductive processes.

The timing of exposure to POPs can affect the degree of toxicity. The most critical or sensitive period for developmental neurotoxicity appears to occur during the height of the brain growth spurt. For example, neonatal mice exposed to BDE-99 during a critical period of brain development experienced impaired spontaneous behavior (i.e., behavior important for survival such as foraging and predator avoidance). However, mice exposed after the growth spurt did not experience the neurotoxic effects (Eriksson et al. 2002). This study indicates that adverse health effects are not only dose-dependent and species-specific, but also dependent on the timing of exposure. Other studies where animals are exposed to POPs during the defined critical period have shown to cause reductions in sperm and spermatid counts in adult rats, increase hyperactivity in their offspring, cause morphological effects in the thyroid, liver, and kidneys, increase circulating thyroid hormones, and alter spontaneous behavior (Viberg et al. 2003, 2007; Kuriyama et al. 2005). Additionally, neonatal exposure may produce long-term modifications in the cholinergic or neurotransmitter system (Talsness 2008). Therefore, marine mammal calves and pups are likely more susceptible to adverse health effects than are adult whales and pinnipeds because the young are exposed to contaminants during the critical period of development. The influx of toxicants in calves and pups is a cause for concern because the

growth and development of an individual is highly dependent on normal levels of thyroid hormones (Boas et al. 2006).

While POPs can present direct health threats to hormonal regulation, neural development and function, and reproduction as discussed above, they can also alter susceptibilities to infectious diseases. One mechanism of action of inducing contaminant effects is through interactions with the aryl hydrocarbon receptor, generally described as “dioxin-like” effects. “Dioxin-like” contaminants are particularly effective at engendering immunotoxicity across a range of species. PCBs, PBDEs, and DDTs have well documented effects on the immune system in a wide range of experimental animals (Thomas and Hinsdill 1978, 1980; Safe et al. 1989; Dahlman et al. 1994). In the absence of a robust immune system, the individual animal’s health, or its ability to endure and thrive, can become compromised. The immune system is important in patrolling and eliminating cells that undergo malignant transformation. If this immune surveillance is compromised, the potential exists for tumors to develop. For example, St. Lawrence beluga whales have a high occurrence of tumors and lesions, and some evidence of immunosuppression, along with high PCB concentrations (Béland et al. 1993; Martineau et al. 1994). California sea lions that died of carcinoma had higher PCB concentrations compared to California sea lions that died without carcinoma (Ylitalo et al. 2005). Contaminants may play a role in the development of disease by suppressing the immune system or through genotoxic mutation and tumor promotion (Ylitalo et al. 2005).

Data from toxicity testing, epidemiological studies, and risk assessments all suggest that there are health concerns at current exposure levels for organophosphate flame retardants (Blum et al. 2019). There are known animal carcinogens classified as organophosphate esters such as TCEP and TDCPP (EPA 2015) in addition to a potential carcinogenic compound, TCPP (Van der Veen and de Boer 2012). The National Toxicology Program is in the process of evaluating TCPP in a 90-day toxicity study and a 2-year cancer bioassay and developmental toxicity study (EPA 2015). In addition to the carcinogenic effects, TDCPP induces acute-, nerve-, developmental-, reproductive-, hepatic-, nephron-, and endocrine-disrupting toxicity in animals, which has caused increasing concern worldwide (Wang et al. 2020). As for TCPP, the structure is similar to an organophosphate pesticide, an established neurotoxicant (Van der Veen and de Boer 2012), and it has been reported to show developmental and reproductive toxicity in pregnant rats (Ji et al. 2020).

Not many studies have evaluated health effects of organophosphate esters in marine organisms; however, a number of studies demonstrate potential health concern in the aquatic system. TDCP concentrations in San Francisco Bay water have regularly exceeded predicted no effect concentrations for marine settings, suggesting concerns for aquatic toxicity (Lin and Sutton 2018; EPA 2019). In 2015, EPA released a work plan to assess the potential risks of chlorinated organophosphate esters to humans and animals including aquatic organisms (EPA 2015). This study found sublethal effects in fish (unspecified species) including loss of coordination, edema, darkened pigmentation, and hyperventilation which suggest potential for long-term population level concerns (EPA 2015). Exposure to TCPP compounds affect the nervous system in zebrafish and rockfish (Ji et al. 2020; Xia et al. 2021) and the immune system in mussels, *Mytilus galloprovincialis* (Wu et al. 2018). A recent study by Hong et al (2021) found that exposure to TDCIPP and TPhP impaired bone development (e.g., decreased and deformed pectoral fin) in

medaka (*Oryzias melastigma*). Moreover, the concentrations of TDCPP and TPhP detected in harbor seal blubber residing nearby San Francisco Bay were comparable to thresholds for aquatic toxicity (Sutton et al. 2019), suggesting potential health concerns for the harbor seal population.

Less is known about early POP exposure among sea turtles. Recent studies have identified POPs transferred from nesting females to eggs and hatchlings likely have consequences on development. For example, POP concentrations in green sea turtles were significantly negatively correlated with body condition of hatchlings, an indication of effects on development (Van de Merwe et al. 2010). POPs may disrupt normal hormone function by altering the concentrations of circulating thyroid hormone (e.g., Hall et al. 2003) as well as interfere with developmental processes (Eriksson et al. 2002, 2006). Recently, Finlayson et al. (2016) reviewed the available sea turtle toxicological research and identified only 49 papers on sea turtle toxicology, highlighting the need for more toxicological endpoints and mixture effects studies.

Among the four ESA-listed sea turtle species discussed in this opinion, POPs (PCBs, PBDEs, and DDT) were associated with clinical health parameters (i.e., weight, carapace length, hematology, etc.) (Keller et al. 2004; Swarthout et al. 2010; Komoroske et al. 2011; Camacho et al. 2013), fibropapilloma (Aguirre et al. 1994; Keller et al. 2014), hatchling mass and success (Van de Merwe et al. 2010; De Andrés et al. 2016), lymphocyte proliferation (Keller et al. 2006), lysozyme activity (Keller et al. 2006), septicaemia (Orós et al. 2009), cachexia (Orós et al. 2009), and pansteatitis (Orós et al. 2013). Finlayson et al. (2016) also summarized the available in vitro and in vivo toxicity studies for sea turtles. Of the few available, they included alterations to immune response, alterations to sex determination processes and sex reversal, genotoxicity, endocrine disruption, metabolic disruption, and disruption of reproduction (see Finlayson et al. 2016 for a review). POPs may also have subtle effects on the development, size, and fitness of sea turtle eggs and hatchlings, which is important for offshore dispersal, predator avoidance, and ultimately survival and population growth (Van de Merwe et al. 2010; Keller 2013). Although levels at which organophosphate esters induce health effects are unknown in sea turtles, a recent study by Sala et al. (2021) detected TPP, triethyl phosphate (TEP), tris(2-isopropylphenyl) phosphate (T2IPPP), and tris(2-butoxyethyl) phosphate (TBOEP) in tissues of loggerhead turtles.

In addition to the legacy POPs (such as PCBs, DDT, and more recently PBDEs), and organophosphate flame retardants, TBT also acts as an endocrine disruptor and has shown to competitively inhibit aromatase cytochrome P450 activity in humans (Heidrich 2001). Aromatase plays a significant role in sustaining the ratio between male and female hormones during sexual differentiation during embryonic development. TBT inhibits the conversion of androgens to estrogens. Although TBT can significantly inhibit P450 activities, the concentration levels in the liver at which this inhibition occurs is almost 25 times higher than that found in free-ranging marine mammals (Kim et al. 1998). However, some marine mammal populations from the North Pacific, off Japanese coastal waters (e.g. finless porpoise, *Neophocaena phocaenoides*, and Risso's dolphin, *Grampus griseus*), have been documented to contain TBT levels high enough to cause immunotoxicity in laboratory species (Tanabe 1999).

2.5.2.3 Mixture Effects and Non-Linear Dose-Response Curves

Marine organisms are exposed to a number of toxic chemicals off California, and the interactions of these chemicals have the potential to be additive (when the effects from two or more chemicals equal the sum of the effects of the isolated chemicals), synergistic (when the effects from the interaction is greater than the sum of the effects of the isolated chemicals), or antagonistic (when the effects from the interaction is less than the sum of the effects from the isolated chemicals). Although wildlife is rarely exposed to single compounds and health risks are likely elevated as a result of interactions between toxic chemicals, the majority of studies have examined the effects of isolated chemicals. It has only been in recent years that studies have examined health effects from exposure to mixtures of chemicals. For example, a few recent studies have highlighted the importance of evaluating mixture effects (Hallgren and Darnerud 2002; Crofton et al. 2005; Eriksson et al. 2006; Fischer 2008; He et al. 2009b, 2009a, 2010). Mixture effects case studies that have examined effects from the interaction of POPs (e.g., Eriksson et al. 2006; He et al. 2009b, 2009a, 2010) demonstrate that the interaction of pollutants is primarily synergistic and toxicity is enhanced, especially when the exposure to the chemical mixture is at a critical developmental growth period.

The practice of examining only high doses of contaminants, especially endocrine disruptors, may underestimate risk (for a review, see Welshons et al. 2003) because some contaminants can interact at doses below the no observed effect concentrations (NOECs) and produce significant effects (Silva et al. 2002). For example, Crofton et al. (2005) tested the hypothesis that a mixture of thyroid hormone-disrupting chemicals has additive dose-response effects. They demonstrated that the effects from a mixture consisting of thyroid hormone disrupters can be additive at low doses and synergistic at high doses, and more importantly, the highest mixture dose levels were at or below the NOECs of the chemicals. Endocrine disruptors, when isolated, have shown to produce nonlinear (e.g., U-shaped or J shaped) dose-response curves. For example, PBDE concentrations in the blubber of grey seals significantly contributed to thyroid hormone concentrations in their blood (Hall et al. 2003). They found a positive association between PBDEs and circulating thyroid hormones, in contrast to several laboratory studies that have reported a negative correlation. Furthermore, the PBDE concentrations in the grey seals were at much lower doses than were used in laboratory studies, suggesting a hermetic dose-response (or an enhancement of the response at low doses and an inhibition at high doses). TBT can also act synergistically with a PCB congener (PCB-126) known to induce P450, and produce opposite effects than when the chemicals are isolated at higher doses. For example, female mice exposed to high doses of TBT combined with PCB-126 inhibited P450 activity, whereas low doses of TBT combined with the PCB congener enhanced the activity (DeLong and Rice 1997).

A nonlinear dose-response relationship is not uncommon in the literature. Additive or synergistic mixture effects can occur from a wide range of doses; therefore, even low concentrations of persistent pollutants when combined together have the potential to cause adverse health effects in marine organisms. Although it is not clear if contaminant levels in ESA-listed species are at or near a health-effects threshold, we assume that a combination of their current body burdens and their exposure to additional accumulation of POPs from wastewater effluent has a potential to disrupt the reproductive system, the endocrine system, and the immune system within an individual's lifetime.

2.5.2.4 Summary – marine mammals and sea turtles

Effluent from Point Loma WWTP contains potentially harmful contaminants that have been well established to adversely affect laboratory and wildlife species. PCBs and DDTs continue to be measured in the action area through sediment, mussel and fish liver/muscle sampling. The threat of TBT may be uncertain, but it will be subject to monitoring in the effluent. The potential presence of organophosphate flame retardants (e.g., TPPO, TCEP, TCPP) is of concern as well. Recent information indicates that increasing use of organophosphate flame retardants is leading to an increased presence and therefore threat from potential exposure to these constituents. Based on surrogate data from a similar adjacent WWTP, we estimate approximately 1095 to 1898 pounds (511 – 839.5 kg) per year of organophosphate flame retardants may be loaded into the action area as a result of Point Loma’s wastewater discharge. Over the course of the five years of this proposed action, this equates to approximately 5475 to 9490 pounds (2,555 – 4197.5 kg) for the total permit cycle. Once in the aquatic system, these constituents become bioavailable to food webs for uptake by ESA-listed species.

ESA-listed marine mammals and sea turtles are affected by the proposed action indirectly by consuming prey that has accumulated POPs from Point Loma’s effluent, which increases both the potential for adverse health effects to occur and the speed with which they affect the listed animals in the action area. Although baleen whales and sea turtles consume prey at lower trophic levels, and their total body burdens are relatively less than other species, endocrine disruptors do not necessarily need to be in high concentration to cause an effect. Furthermore, there may be synergistic effects between different POPs, likely increasing the health risks to marine mammals and sea turtles. Thus, increasing POP levels in the ESA-listed species only further exacerbates their current susceptibility to adverse health effects, including effects on the exposed animals’ reproductive, endocrine, and immune systems.

As described above, we expect that individuals from all of the ESA-listed marine mammal and sea turtle species that may occur in the action area may make numerous or possibly frequent and extended visits to the area and be exposed to additive accumulation of POPs, increasing the risks of adverse effects that these contaminants are known to present. As described in Section 2.4 (Environmental Baseline), there are numerous other potentially harmful contaminants for ESA-listed marine mammals and sea turtles, and many of those are likely to also be present in Point Loma’s discharge. However, the information describing the levels of most of these contaminants in wastewater discharge (and the extent of their potential harmful effects) is limited. As described in Section 1.3.4 (Special Studies), the City proposes to participate in the SCCWRP effort by providing analysis of the PLOO effluent for a target number of CECs (e.g., nitrogen, phosphorus and chlorophyll-*a*). The City also volunteers to perform testing of the Point Loma PLOO discharge for target CECs/POPs prior to and after the startup of Pure Water phase 1. These studies would provide information on the presence of CEC/POPs in the PLOO discharge that may be harmful to the ESA-listed species as well as what changes may result from Pure Water upstream offloading.

2.5.2.5 Adverse Abalone Health Effects from Exposure to Potentially Harmful Contaminants

There are very few studies evaluating the effects of persistent and potentially harmful contaminants such as POPs on white abalone and other California abalone species, some have been conducted on abalone species worldwide. For example, field studies in Japan found that exposure of *H. madaka* and *H. gigantea* to the organotin compounds TBT and triphenyltin (TPhT) (found in anti-fouling paint) caused ovarian spermatogenesis (masculinization) and altered the timing of reproductive maturity in males and females (Horiguchi et al. 2001, 2005).

Studies have also found adverse effects on larval abalone behavior and development from TBT (Horiguchi et al. 1998). Because abalone rely on synchronous spawning, this altered timing can result in reduced reproductive potential. Gaume et al. (2012) found toxic effects on immune and respiratory cells of *H. tuberculata* when exposed to concentrations of triclosan (an antibacterial agent) ranging from 2 to 10 μ M for 24 to 48 hours. Zhou et al. (2010) exposed *H. diversicolor supertexta* to diallyl phthalate (50 ug/L) and bisphenol A (100 ug/L) for three months and found altered protein expression that could affect physiological functions such as detoxification, immunity, metabolism, and hormonal modulation.

Some of these potentially harmful contaminants have not been detected in Point Loma's effluent, and some have been detected but at lower levels than those described above (City of San Diego 2019, 2021c). Thus, while white abalone within the action area may be exposed to these and other potentially harmful contaminants in Point Loma's effluent, it would likely be at low concentrations because the plume is diffuse in the area of the white abalone habitat and the contaminant loads are low or nonexistent. We do not know how these contaminants may accumulate in abalone over time or what effects this accumulation may have on their health, though effects such as reduced growth and reproductive capacity are possible.

Overall, we expect white abalone within the action area to be exposed to potentially harmful contaminants in Point Loma's effluent and to accumulate these contaminants over time, which may result in adverse effects on individual health. Further studies are needed to evaluate the concentrations of potentially harmful contaminants in the effluent, the concentrations to which white abalone may be exposed, and the effects of these potentially harmful contaminants on abalone at those concentrations over time. As described in section 1.3.4 Special Studies, the City of San Diego plans to analyze specific CECs and POPs in the PLOO effluent as part of a SCCWRP effort and to assess the effects of the Pure Water project on the discharge. These analyses would provide information on some of the CECs and POPs in the PLOO discharge that may be harmful to ESA-listed species.

2.5.3. Harmful Algal Blooms

The discharge of effluent contributes additional nitrogen and other nutrients to the action area off the Southern California coast (e.g., Reifel et al. 2013; Howard et al. 2017; McLaughlin et al. 2017). This promotes HABs that potentially pose a threat to ESA-listed species.

2.5.3.1 Effect of Point Loma's Discharge on HAB Occurrence

Point Loma discharge may have the effect of fertilizing or kick-starting the springtime HABs by sustaining or even increasing populations of HAB species in the subsurface water (Trainer et al. 2007; Cochlan et al. 2008; Seeyave et al. 2009; Kudela et al. 2010; Nezlin et al. 2012; Seegers et al. 2015) providing nitrogen to the euphotic zone of the action area when stratification is weak or shallow. Concentrations of nitrogen (and phosphorus) in the effluent plumes are up to three orders of magnitude greater than maximal ambient concentrations, and they also entrain deeper, nutrient laden water as they rise to the thermocline or close to the surface when stratification is weak (Reifel et al. 2013; Seegers et al. 2015). These conditions can occur during the winter months in the action area. HAB species (*A. catenella*, *P. spp.*) (Trainer et al. 2010; Seegers et al. 2015) are known to persist in the euphotic, subsurface zone and then be advected into the shallow surface waters during the spring upwelling season where the combination of nutrient availability and increasing sunlight may result in a bloom. Nezlin et al. (2012) found that all four large WWTPs in the SCB had “hot spots” of high offshore chlorophyll- α (CHL- α) and that these conditions occurred throughout most of the year.

The City of San Diego (2021c) states that they do not have a direct measure of the euphotic zone specific to the San Diego region. They have previously reported that light transmissivity levels between 75-100% have been recorded to depths of 100 meters as part of the PLOO monitoring program, but light transmissivity is not directly correlated with the euphotic zone. The depth of the euphotic zone is defined as the point where photosynthetic available radiation (PAR) is 1% of its surface value. The euphotic zone is generally considered to be 200 meters deep in open ocean waters (https://oceanservice.noaa.gov/facts/light_travel.html). Mantyla et al. (2008) noted chlorophyll layers at 85 meters deep in offshore waters of the SCB, while a recent paper by Kessouri et al. (2021a) defined the euphotic zone in the SCB as 0-40 meters deep. The lack of seasonal, measured euphotic zone data creates significant uncertainty in evaluating the potential impacts of the nutrients discharged at the PLOO. A contractor for the City reported in 2009 that the effluent plume was observed indirectly by remote sensing in the upper 15-40 feet of the water column (City of San Diego 2015a); Scripps Institution of Oceanography detected the plume at 35 meters deep (Rogowski et al. 2012a); and, EPA reported that the maximum height of the waste field at 30 to 44 meters deep (EPA 2017c). These ranges are likely within the euphotic zone, indicating that the nutrients in the PLOO discharge are at least periodically available for uptake by phytoplankton and can contribute to primary production in the action area.

The physical oceanography in the vicinity of the Point Loma discharge influences the fate and transport of the nutrients and any subsequent phytoplankton or zooplankton that utilize the nutrients to grow. As discussed in the environmental baseline section, ocean conditions within the action area are affected by both regional- and local-scale currents, and local gyres (City of San Diego 2021a). The meandering California Current has a large shoreward component near 32° N; this component splits as it approaches the coast and greatly affects the action area. Recent modeling (Kessouri et al. 2020b) confirms previous work (Lynn and Simpson 1987; Mantyla et al. 2008; Rogowski et al. 2012b) by showing that local currents have considerable variability in position, strength, and depth, and can often be coming ashore right toward Pt. Loma. As a result, near-shore currents in the area have been observed by the City (City of San Diego 2016, 2021a, 2021b) (City of San Diego 2021a, 2021b, 2016) variously heading generally north or south directing the effluent towards La Jolla or the Mexican border. Monitoring by the City (City of San

Diego 2016, 2021a, 2021b) also indicates shoreward movement of the PLOO discharge before turning along-shore. Nezlin et al (2012) identified the south San Diego area as a hot spot area with longer residence time of its water and higher CHL- α levels. Additional nutrients may enter the immediate vicinity of the PLOO from WWTPs to the south (i.e. South Bay Water Reclamation Plant, the International WWTP, and the San Antonio de los Buenos WWTP) and from Tijuana River and San Diego River flows, and other urban stormwater runoff.

Nitrogen from upwelling is largely in the form of nitrate (98.7%) while nitrogen in effluent is largely ammonium (92%), a reduced form (Howard et al. 2012b, 2014). There are several sources that summarize numerous studies and conclude that reduced forms of nitrogen (ammonium, urea) significantly tilt the phytoplankton community toward the development of HABs (Howard et al. 2012b; Reifel et al. 2013; Booth 2015; Seegers et al. 2015). Kudela et al. (2008) showed that ammonium uptake by *A. sanguinea*, a red tide forming dinoflagellate that does not produce a toxin, was approximately threefold higher than uptake of nitrate. Kudela et al. (2010) later showed that *P. spp.* grew equally well or better on reduced nitrogen sources and Howard et al. (2007) showed that *P. australis* could use either nitrate or ammonium simultaneously. Schnetzer et al. (2007) cites several studies that examined *P-spp.* and noted that their effective toxicity can be highly variable. These diatom species seem to produce higher levels of domoic acid when under silica or phosphate stress (i.e. the nitrogen:phosphorus and/or nitrogen:silica ratios are higher than or altered from natural conditions) (Anderson et al. 2006; Schnetzer et al. 2013). The discharge of large amounts of nitrogen in the effluent can have the effect of unbalancing these ratios at the local level and may be partially responsible for the very potent HABs that have been occurring in the spring for ~10 years in SCB (Nezlin et al. 2012).

Howard et al. (2014) estimated that nutrient loading of the San Diego subregion through wastewater effluent increased total nitrogen in the San Diego subregion by about 16,310 pounds (7400 kg) of nitrogen per km² per year. Given the area of the San Diego subregion (1020 km²), this amounts to about 16.6 million pounds (7.5 million kg) of nitrogen over the course of a year. As described in Section 2.4.1.1.4 (Harmful Algal Blooms), this amount of nitrogen was estimated by Howard et al. (2014) to be roughly equal to three times the contribution of upwelling in the San Diego area (as defined therein) although the upwelling estimate is likely an underestimate. Recent estimates of average effluent discharge volume at the PLOO have been approximately 140 MGD (City of San Diego 2021b), which is less than the discharges examined by Howard et al. (2014) but more than twice as large as recent effluent discharges from the SBOO (~40 MGD maximum) and San Antonio de los Buenos WWTP (~25 MGD) combined. An additional facility to the north (University of California, San Diego – Scripps Institution of Oceanography) discharges ~1.25 MGD of mostly aquarium related and filter backwash flows (no “sewage” processed here) that we consider inconsequential in this analysis. Unfortunately, the San Antonio de los Buenos WWTP is in disrepair and we cannot accurately estimate the nutrient loading from it. This makes a relative comparison of the nutrient loading from these three outfalls in the action area uncertain. Although the available information is insufficient to precisely assign the nutrient contributions of Point Loma as a percent of the total in the action area, it is evident that the facility’s constant contribution to the receiving water is significant, and may be in the same range as WWTP nutrient contributions to Santa Monica Bay (~ equal to upwelling) or the San Pedro shelf (~50% of upwelling).

2.5.3.2 Potential Adverse Marine Mammal and Sea Turtle Health Effects from Exposure to HABs

The potential for ESA-listed species to be exposed to biotoxins present in HABs is dependent on the co-occurrence of the harmful taxa present in the action area and ESA-listed species and/or their prey species. If a HAB occurs and exposes the food web (e.g., plankton, small fishes) in an area where ESA-listed species occur, then there is an increased likelihood for ESA-listed species to be exposed to any biotoxins produced. Because the majority of life in the action area depends on phytoplankton, the listed species and their prey are likely to be exposed. As described above, it is likely Point Loma's effluent could function as a seed and kick-start HABs. The SCCOOS tracks HABs in and around the action area. Based on the frequent bloom events along the coast, it is likely that HABs would occur in and around the action area during the five years of the proposed permit. Although it is uncertain what degree Point Loma's effluent would play in any HAB, we anticipate the Point Loma effluent would help encourage HAB formation over the next five years (and in the future, as well). Therefore, any ESA-listed marine mammals or sea turtles foraging nearby would experience an increased risk of exposure to biotoxins.

Four classes of marine algal toxins have been associated with marine mammal mortality and morbidity events: saxitoxin, brevetoxin, ciguatoxin, and domoic acid (Van Dolah et al. 2003). Between 1978 and 2006, the NOAA Fisheries Stranding Network detected 57 of these mortality events nationally. Of those events, 29 were declared UMEs (Gulland 2006). A UME is defined under the MMPA as "a stranding that is unexpected; involves a significant die-off of any marine mammal population; and demands immediate response." In 1991, the marine mammal UME program was established and has since recognized 71 UMEs (Figure 9). Of the 71 UMEs, 18% have been caused by biotoxins from HABs (Figure 9), with the majority being attributed to toxicity from domoic acid and brevetoxin. Other causes of mortality events include viruses, bacteria, parasites, human interactions and oil spills, and changes in ocean conditions (Gulland 2006). Most of the declared UMEs have occurred in California and Florida coastal waters.

Of the four biotoxins, domoic acid can occur in the action area because of the harmful taxa present. Diatoms in the *Pseudo-nitzschia* and the *P. seriata* group appear to be prevalent in the vicinity of Scripps Pier (SCCOOS data, <https://sccoos.org/harmful-algal-bloom/>). The dinoflagellates *Prorocentrum* spp. and *Akashiwo sanguinea* that can produce saxitoxin are also known to be present near Point Loma (<https://sccoos.org/harmful-algal-bloom/>). Phytoplankton assemblages in the action area appear to be dominated by diatoms (i.e., *Pseudo-nitzschia* spp.) in the winter to early spring and by dinoflagellates beginning in late spring (Schnetzer et al. 2013). Dinoflagellate species included *Prorocentrum* spp., *Ceratium* spp., *Cochlodinium* spp., and *Lingulodinium polyedrum* (Schnetzer et al. 2013). These dinoflagellates can produce a number of toxins: *L. polyedrum* produces a yessotoxin; *Prorocentrum* spp. produce okadaic acid and pectenotoxins that cause diarrhetic shellfish poisoning; *Cochlodinium* spp. produce ichthyotoxins; and *Ceratium* spp. have been linked to anoxia and the production of hydrogen sulfide (Trainer et al. 2010).

In mammals, saxitoxins appear to affect the peripheral nervous system, and the primary cause of death in response to exposure is respiratory paralysis. This biotoxin is considered to be responsible for PSP, and can act quickly in species following exposure through accumulation in

prey (O’Hara and O’Shea 2001). Some of the symptoms of exposure to this biotoxin can include lethargy, lack of motor control, paralysis, and death (Van Dolah et al. 2003). Domoic acid causes the syndrome known as Amnesic Shellfish Poisoning. In humans, permanent loss of short term memory was experienced following domoic acid exposure (Van Dolah et al. 2003). Other symptoms in humans from domoic acid have included nausea, vomiting, diarrhea, dizziness, disorientation, lethargy, and seizures (Van Dolah et al. 2003). Signs of domoic acid toxicity in wildlife can include seizures, head weaving, decreased responsiveness to stimuli and scratching behavior (Work et al. 1993; Van Dolah et al. 2003).

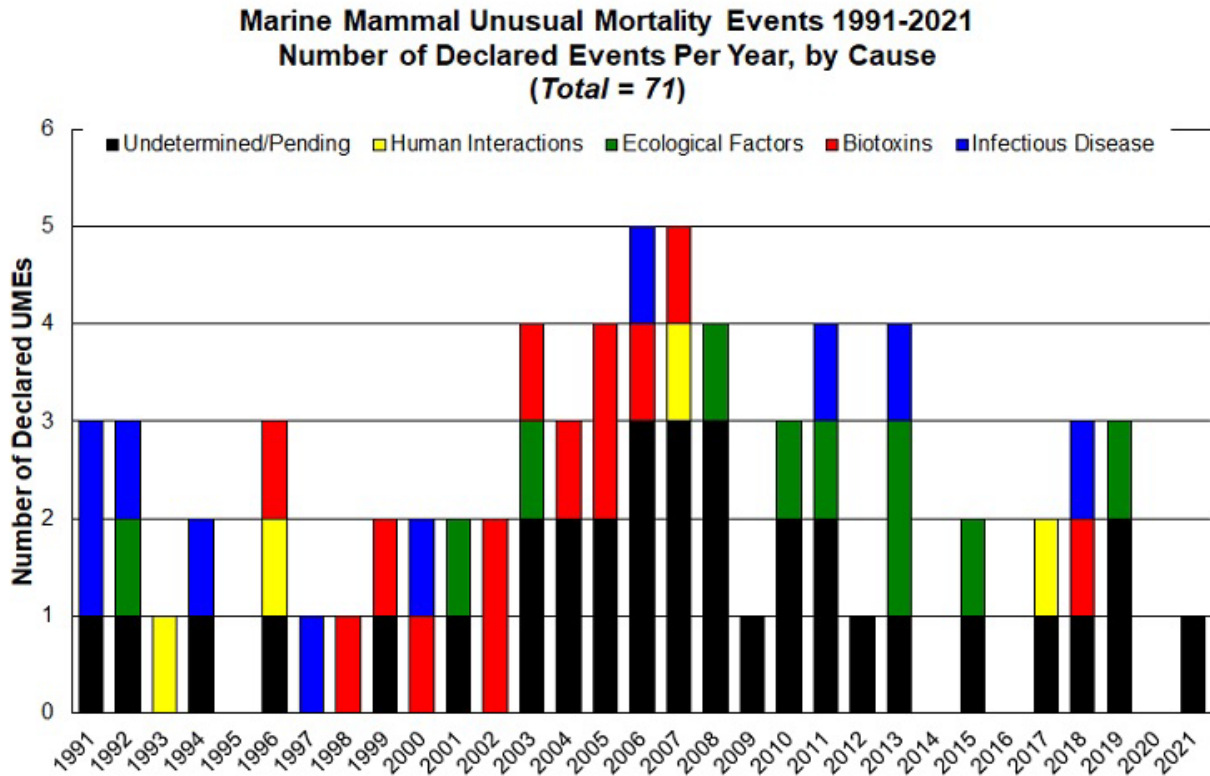


Figure 9. Number of unusual mortality events between 1991 and 2021 per year and by cause (figure reprinted from <http://www.nmfs.noaa.gov/pr/health/mmume/events.html>).

These biotoxins clear rapidly from the blood. Consequently, diagnosis in marine mammals is difficult without a thorough examination. For example, the highly endangered Mediterranean monk seals (*Monachus monachus*) found in coastal waters off West Africa experienced a mortality event in 1997 that reduced the population abundance to almost half its size (Forcada et al. 1999). Osterhaus et al. (1997) had identified morbillivirus in several of the monk seal carcasses, and that was considered a likely cause of the mortality event. However, a subsequent competing theory suggested the mortality event may have been caused by biotoxin exposure (Hernández et al. 1998). Terminally ill individuals exhibited the known clinical symptoms of exposure to saxitoxin (e.g., lethargy, motor incoordination, and paralysis). Furthermore, they found that the time between onset of these clinical signs and death through drowning by paralysis was short. Hernández et al. (1998) also detected high concentrations of the

dinoflagellate *Alexandrium minutum* in the coastal waters as well as in the dead seals. However, because there are no data on the background levels of these toxins in the seals or their prey, and there are no baseline data on the prevalence of virus antibodies, a conclusive diagnosis could not be made (Harwood 1998).

Saxitoxins were also implicated in a mortality event in humpback whales in Cape Cod Massachusetts between November 1987 and January 1988 (Geraci et al. 1989). Fourteen humpback whales died in 5 weeks. During this same time, 2 fin whales and a minke whale (*Balaenoptera acutorostrata*) also stranded. All the humpback whales appeared to be in good condition prior to death, which appears to have occurred quickly. For example, one individual was observed acting normally but within 90 minutes was found dead. Based on examination of the mackerel the whales were consuming, Geraci et al. (1989) estimated the whales were likely consuming approximately 3.2 µg of saxitoxin per kg of body weight. In comparison, the lethal dose for humans is substantially higher at 6-24 µg/kg (Levin 1992) suggesting humpback whales are relatively more sensitive to this biotoxin.

In general, large mammals are more sensitive to bioactive compounds, so extrapolation from human studies is not appropriate for saxitoxins (Stoskopf et al. 2001). The increased vulnerability for humpback whales (and likely other large whales) that were exposed to saxitoxin off Massachusetts in 1987 and 1988 may be due to the fact that a larger proportion of their body weight is blubber (Geraci et al. 1989). Because saxitoxin is water soluble, it will not partition as readily in the blubber. This means there may be a higher concentration of these biotoxins in more sensitive tissues. Geraci et al. (1989) suggests another reason could be from a whale's diving physiology, which concentrates blood to the heart and brain and away from organs used to detoxify, creating higher concentrations of neurotoxin in sensitive tissues. Although the reason for the increased vulnerability of large mammals is uncertain, it is likely that whales feeding in a HAB will be more susceptible to toxic effects than smaller mammals.

The first confirmed domoic acid toxicity in marine mammals occurred in 1998 off the California coast. Seventy California sea lions and one northern fur seal stranded along the central California coast during May and June (Gulland 2000). The sea lions were all noted to be in good physical shape and displayed the clinical symptoms including head weaving, scratching, and seizures. The majority of the stranded sea lions died and domoic acid was detected in the sea lions' urine, feces, and serum. In Monterey Bay, a bloom of *Pseudo-nitzschia australis* occurred and was implicated in the mortality event (Scholin et al. 2000). Closely following the sea lion mortality event, an increased number of sea otter deaths occurred in the same region (Van Dolah et al. 2003).

Following a *P. australis* bloom in Monterey Bay in 2000, 25 gray whales stranded in the San Francisco Bay area (Van Dolah et al. 2003). Approximately half of the whales were sampled for domoic acid, one of which had levels at concentrations that would implicate domoic acid toxicity. Because clearance of this biotoxin is fast, it is not clear if the other whales had been exposed as well. It was previously believed that gray whales typically do not forage during their northern migrations from the nursery grounds to their feeding grounds. However, gray whales have been observed feeding off California and Washington (Van Dolah et al. 2003). Krill were also collected offshore of Monterey Bay following the bloom and identified as a potential vector

for domoic acid to higher trophic level species (Bargu et al. 2002). Based on the maximum domoic acid concentrations measured in krill, Bargu et al. (2002) estimated krill could transfer domoic acid levels up to 62 grams to a blue whale per day, or 0.62 mg per kg.

In 2002, a UME was declared as over 2,000 animals stranded in southern California from April to June with neurological symptoms. The affected species included mostly California sea lions and long-beaked common dolphins (*Delphinus capensis*). The stranding was linked to a bloom of *Pseudo-nitzschia* (Torres de la Riva et al. 2009). Their results also suggested that both inshore and offshore foraging species were affected. In fact, domoic acid was confirmed in 11 out of 11 California sea lions tested, 23 out of 26 common dolphins tested, and a Risso's dolphin (*Grampus griseus*), Cuvier's beaked whale (*Ziphius cavirostris*), gray whale, and a humpback whale (Torres de la Riva et al. 2009).

Some species may have the ability to detect and avoid exposure to biotoxins. For example, the butter clam (*Saxidomus giganteus*) in Alaska is a primary prey eaten by sea otters and is able to retain saxitoxin in its siphon for up to a year (Kvitek and Beitler 1991; Kvitek et al. 1991). In a feeding study with caged sea otters (*Enhydra lutris*), the otters reduced their consumption rates when fed toxic butter clams and appeared selective in consuming sections of the clams less toxic and discarded the more toxic tissues (Kvitek et al. 1991; Van Dolah et al. 2003). However, due to the large number of strandings, it does not appear that baleen whales or pinnipeds have this ability to detect and avoid biotoxins.

HABs are also known to affect sea turtles. Brevetoxins are produced by the dinoflagellate *Karenia brevis* and best known to be responsible for Florida red tides. Between 2005 and 2006, 318 sea turtle strandings were documented off Florida's Gulf of Mexico (a four-fold increase). The cause of death in approximately 90% of the individuals was from red tide intoxication (Fire and Van Dolah 2012) (<http://www.whoi.edu/redtide/page.do?pid=153356>). The data revealed the turtles were consuming contaminated prey and inhaling the toxin.

Less is known about biotoxin exposure and effects in ESA-listed sea turtles that may occur in the action area. Harris et al. (2011) conducted physical examinations on foraging western Pacific leatherback sea turtles from California between 2005 and 2007. They analyzed for domoic acid in plasma and feces and did not detect any domoic acid in plasma above 5 parts per billion nor in the feces at or above 500 parts per billion. They suggested jellyfish might not concentrate domoic acid as readily as forage fish. However, they did detect trace levels of domoic acid in the urine of a fresh dead leatherback that was struck by a propeller off the coast of California in 2008. Harris et al. (2011) emphasized that urine and stomach contents are likely better samples than plasma and feces for evaluating exposure to domoic acid (Tor et al. 2003).

More recent evidence was acquired from a stranded leatherback in Santa Cruz, CA, that was in good body condition and had no obvious cause of death (NMFS WCR strandings data). High domoic acid was measured, and the pattern was consistent with it being metabolized (i.e., increasing in concentration down the gastrointestinal tract and was observed in the intestines, stomach, feces, and bladder). The domoic acid ranges detected in the sea turtle were within the range detected in acutely intoxicated seizing California sea lions. These data reveal sea turtles

can be exposed to and affected by biotoxins, but it is currently unclear whether the effects are similar to that found in birds and marine mammals.

2.5.3.3 Potential Adverse Abalone Health Effects from Exposure to HABs

Abalone mortality events have been linked to HABs along the California coast. For example, in 2007, a *Cochlodinium* bloom killed red abalone at the Monterey abalone farm by causing gill damage and reducing dissolved oxygen levels (Howard et al. 2012a; Wilkins 2013). In 2011, a die-off of abalone and several other invertebrate species off Sonoma County was linked to a bloom of a dinoflagellate in the *Gonyaulax spinifera* species complex that produced high levels of yessotoxin (Rogers-Bennett et al. 2012; De Wit et al. 2014a). In surveys conducted during the mortality event, an average of 25% of the red abalone observed were dead or dying (Rogers-Bennett et al. 2002; De Wit et al. 2014b).

Although these blooms occur at the water's surface, the toxins can affect abalone at depth. In the 2011 die off, red abalone mortalities were observed at all depths surveyed (0 to 20 meters), with the highest percent mortalities observed at 0-5 meters depth (approximately 20-75% mortality) and lower percent mortalities at 10-20 meters depth (less than 10% to nearly 30%) across the four surveyed sites (De Wit et al. 2014a). In this case, the exposure pathway is unclear but abalone may have ingested the dinoflagellate or its cysts on macroalgae (Rogers-Bennett et al. 2012).

In 2020, an unprecedented red tide event occurred in the SCB (including the action area); it involved the proliferation of species such as *L. polyedra* (City of San Diego 2021c). This red tide event caused poor water quality at the SWFSC La Jolla laboratory and probably killed one of the captive black abalone being held at the laboratory (SWFSC 2021).

Both *L. polyedra* and *Cochlodinium* are commonly detected within the action area (see Section 2.4.1.1.4) along with many other HABs species that could affect white abalone within the action area. As explained above, we conclude that the discharge of effluent by the Point Loma WWTP can potentially increase HAB frequency, extent, and intensity in the action area and thereby increase the likelihood that abalone would be adversely affected.

2.5.3.4 Summary

Similar to the analysis of potential effects of adding potentially harmful contaminants like POPs to the environment and increasing the accumulation of these contaminants by ESA-listed species, we conclude that the discharge of effluent by Point Loma can potentially increase HAB frequency, extent and intensity. At this time, we cannot predict the precise extent that Point Loma's effluent discharge contributes to increased probabilities of HABs, or distinguish which HABs may be more or less associated with or influenced by the additional nutrient input created by Point Loma's discharge. What is clear is that (1) HABs pose a significant health risk for ESA-listed marine mammals, sea turtles, and abalone; (2) increasing the probability of HAB occurrence further increases the likelihood of adverse effects from HABs, including impaired health (injury) and mortality; and (3) Point Loma's discharge increases that probability. As

described above, we expect that all of the ESA-listed marine mammal and sea turtle species that may occur in the action area have individuals that may: (a) make numerous or possibly frequent and extended visits to the area; and (b) be exposed to increased frequency or extent of HABs during those visits, increasing the HAB-based risks of adverse effects. White abalone in the area would be exposed to the greater risks associated with increased HAB frequency, extent and intensity.

2.5.4. Risks to Populations

2.5.4.1 Marine Mammals and Sea Turtles

In summary, Point Loma's discharge of effluent poses a risk to ESA-listed marine mammals and sea turtles by exposing individuals to pollutants in the effluent and plume, and to increased HAB frequency, intensity, and extent. The concentrations of metals and most other potentially toxic constituents in the discharge effluent plume are likely to be lower than those typically expected to cause harmful effects for more sensitive species and do not pose much of a threat for direct uptake from the water column or bioaccumulation through the food chain.

On the other hand, studies confirm that marine mammals in particular, and likely sea turtles as well, are susceptible to endocrine disruption and harmful effects from POPs and other potentially harmful constituents that are known or expected to be found in the effluent (e.g., organophosphate flame retardants). The proposed action is likely to increase the body burdens of these contaminants and potentially diminish some animals' health and fitness. However, further studies are needed to evaluate the levels of potentially harmful contaminants found in the effluent and their effects on ESA-listed marine mammals and sea turtles, as well as other marine species.

Finally, HABs have been documented to cause health issues and even kill marine mammals along the California coast. The potential increase in frequency and/or extent of HABs due to the discharge effluent increases mortality risk for marine mammals especially, and possibly sea turtles as well. Further studies are needed to evaluate the composition, frequency, and extent of HABs that occur in the action area, characterize how they relate to the PLOO effluent and evaluate the risk that that combination poses for listed species.

Based on our analysis, we conclude that exposure to the discharge effluent and potential associated environmental effects have the potential to reduce the fitness and survival of ESA-listed marine mammals or sea turtles in the action area. We cannot discount these effects as extremely unlikely to occur or dismiss them as insignificant.

However, given the available information, it is difficult to assess how these effects are expected to affect ESA-listed marine mammals and sea turtles at the population and species level. Long-term effects on individuals, including diminished reproductive capacity and lower survival rates, could result from the continued accumulation of potentially harmful contaminants that would likely be intensified by the proposed action. Increased and augmented HABs resulting from the proposed action are also likely to produce more acute effects, such as physical impairments, reduced foraging, disorientation, and even death among listed animals. These effects are likely to

reduce the abundance of ESA-listed populations directly through removals, and indirectly through lost reproductive capacity or success.

While we can generally describe the distributions of different populations across U.S. West Coast and within the action area, we are unable to describe the specific extent of exposure in terms of the number of individuals from each population that may be affected, and specifically track the level of exposure or response from any individual while it is within the action area or after departure. We expect that individuals from these ESA-listed species will be exposed to Point Loma's discharge, based on evidence that individuals of these species visit the action area. Given the transitory nature of most of these species and their broad distribution in the Pacific, exposure to the proposed action is likely somewhat limited at the population scale to relatively small segments of populations that may occasionally visit or favor the area as opposed to large proportions or entire populations. There may be some exception to this premise for smaller populations like Central American DPS humpback whales where many, if not all, members of the population may visit the SCB on an annual basis and potentially could occur near or within the action area occasionally and be exposed. However, although we can generally describe the extent of the exposure of individuals to the discharge and its effects, we cannot further describe exactly how many individuals or what percentage of these populations will be exposed or potentially affected. In addition, as explained above, the extent of effects (reduced health, changes in reproductive success or capacity, or even death) that can be expected at the individual level are also highly uncertain, making it difficult to anticipate what the population level effects may be. As described in Section 2.5 (Effects of the Action), we generally expect that exposure will be limited to relatively few individuals (adults or juveniles) or small portions of these populations over the duration of permit. Exposure is more likely for individuals that may have some preference for or site fidelity to the action area. Although there is uncertainty in the specific extent of population level exposure, at this time we generally do not anticipate widespread effects across populations that could potentially produce reduced productivity or fitness at a population level for any of these species.

2.5.4.2 Abalone

In summary, the discharge effluent could affect white abalone by exposing individuals to potentially toxic pollutants in the effluent, causing them to accumulate of more persistent pollutants, and exposing them to increased HAB frequency, intensity, and extent. Based on the location of known white abalone populations and suitable white abalone habitat in relation to the discharge point, we expect white abalone to be exposed to diluted concentrations of the effluent plume. We expect the concentrations of heavy metals in the plume to which abalone may be exposed to be well below the levels found to cause harmful effects on other abalone species, and therefore likely to have minimal effects on white abalone. We also expect white abalone to be exposed to low concentrations of more persistent contaminants in the plume, which can accumulate over time and result in adverse effects on the health of individual white abalone. HABs have been known to kill abalone along the California coast. The potential increase in HAB frequency, intensity, and extent due to the discharge effluent poses an increased risk of killing abalone. Further studies are needed to evaluate the composition, frequency, and extent of HABs associated with the discharge effluent and the potential exposure of white abalone to these blooms.

Based on our analysis, we conclude that exposure to the discharge effluent and potential associated environmental effects have the potential to reduce the fitness and survival of individual juvenile and adult white abalone in the action area. We cannot discount these effects as extremely unlikely to occur or dismiss them as insignificant.

However, given the available information, it is difficult to assess how white abalone would be affected at the population and species level. The white abalone in the action area represent only a few of the known, remaining wild individuals along the Southern California coast. In addition, the action area encompasses one of two areas along the Southern California coast where outplanting of white abalone has been conducted since 2019, as part of experimental studies to increase numbers in the wild. Because of this, it is important to understand how the discharge plume may be affecting water quality and white abalone in the action area, to ensure the best conditions for the survival and growth of remaining wild and outplanted white abalone. Long-term effects on individuals, including reduced growth and reproductive capacity, could result from continued accumulation of potentially harmful contaminants, which would be further accelerated by the proposed action. Individuals may also be more likely to be killed by HABs. These effects are likely to reduce the abundance of white abalone and the reproductive capacity or success of the remaining individuals.

The extent of these effects at the individual level are highly uncertain, making it difficult to anticipate what the population level effects may be. However, we expect effects on individual fitness to be minimal and restricted to a few animals only, thereby also limiting effects at the population level. For example, we expect abalone to be exposed to highly diluted, low concentrations of contaminants in the plume, which will likely require a long period of time to accumulate to levels that result in adverse effects on individual health. We do not expect that all abalone in the action area will be exposed to contaminants in the plume at the same concentrations, resulting in varying levels of exposure, uptake, and accumulation across individuals. We also do not expect that all abalone in the action area will be exposed to all HABs that occur within the area. Given the distribution of abalone in the action area and the best available information on past effects, we expect HAB-related impacts, including mortality, to generally be limited to a few abalone in a confined area at any given time, limiting the effects on the population and species as a whole.

2.6. Cumulative Effects

“Cumulative effects” are those effects of future State or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation [50 CFR 402.02 and 402.17(a)]. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

Some continuing non-Federal activities are reasonably certain to contribute to climate effects within the action area. However, it is difficult if not impossible to distinguish between the action area’s future environmental conditions caused by global climate change that are properly part of the environmental baseline *vs.* cumulative effects. Therefore, all relevant future climate-related

environmental conditions in the action area are described earlier in the discussion of environmental baseline (Section 2.4).

As described in the environmental baseline, the San Antonio de los Buenos WWTP facility discharging near the U.S.-Mexico border is expected continue to discharge in the vicinity of the action area. Discharges from this facility may cease in the future, but when that may occur is uncertain.

Cumulative effects could also occur from non-point source pollution, particularly stormwater runoff. Non-point source pollution can bring additional bacteria, pesticides, fertilizers, oil and gas, trash, and heavy metals into the action area. Two river systems drain into the action area: the San Diego River and the Tijuana River. Two other rivers drain into San Diego Bay: the Sweetwater River and the Otay River. These rivers may affect water quality in the action area via tidal exchange between the action area and San Diego Bay.

We did not identify additional state or private activities that are reasonably certain to occur within the action area, do not involve Federal activities, and could result in cumulative effects on ESA-listed species. Oil spills and the introduction of other pathogens and parasites could occur within the time frame of the permit that could affect ESA-listed species within the action area. However, the potential effects are difficult to evaluate at this time, given the unpredictability and uncertainty in the timing, location, scope, and severity of such events. Spills can result in very different effects depending on many factors, including the type and amount of oil spilled, and local conditions. In addition, although we have examples of how other pathogens have affected ESA-listed species in other parts of the world, there are many uncertainties regarding whether and when these pathogens could spread to California and the effects on ESA-listed species.

2.7. Integration and Synthesis

The Integration and Synthesis section is the final step in assessing the risk that the proposed action poses to species and critical habitat. In this section, we add the effects of the action (Section 2.5) to the environmental baseline (Section 2.4) and the cumulative effects (Section 2.6), taking into account the status of the species and critical habitat (Section 2.2), to formulate the agency's biological opinion as to whether the proposed action is likely to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing its numbers, reproduction, or distribution.

We aggregate the Integration and Synthesis across species groups (e.g., marine mammals and sea turtles) for two reasons: (1) overall similarities in how some ESA-listed species are exposed to the proposed action at an individual and population level; and (2) uncertainty regarding the occurrence and magnitude of adverse effects that may result from the proposed action, limiting our ability to describe expected effects for each species individually. We provide a general synthesis of our understanding of how the proposed action may affect ESA-listed species and, where appropriate and necessary, we consider and describe any species-specific risks relevant to concluding this biological opinion.

The proposed action is EPA's reissuance of a NPDES permit for the Point Loma wastewater treatment plant. The purpose of the NPDES permit is to authorize the discharge of modified secondary (i.e., advanced primary) treated wastewater to the Pacific Ocean from the 23,472 foot long, 310-foot deep Point Loma Ocean Outfall. The NPDES permit would be valid for a period of five years.

2.7.1. Marine Mammals and Sea Turtles

As described in Section 2.5 (Effects of the Action), we do not anticipate that ESA-listed marine mammals and sea turtles will experience any adverse health effects associated with most of the potentially toxic compounds and elements found in Point Loma's effluent discharge plume as a result of occasional exposure when foraging in the action area. We base this conclusion on the limited exposure to concentrated amounts of these constituents and/or minimal risks the exposure may pose to their health.

However, as described in Section 2.4 (Environmental Baseline) and Section 2.5 (Effects of the Action), ESA-listed marine mammals and sea turtles that may occasionally occur in the action area are susceptible to diminished health and reduced fitness as a result of exposure to potentially harmful contaminants, including POPs such as organophosphate flame retardants. Individuals of these species may already carry loads of potentially harmful contaminants prior to exposure (or as a result of previous exposure) to the proposed action; these existing loads could already be compromising overall health and fitness. We recognize that Point Loma's discharge may contain numerous other contaminants that could potentially harm ESA-listed species, but the lack of information on these contaminants, their effects, and their concentrations limits our ability to analyze those effects further.

As described in Section 2.5 (Effects of the Action), we expect the proposed action will increase the amount of POPs and other potentially harmful contaminants that are released into the environment. This will ultimately increase or expedite the accumulation of these potentially harmful constituents in the bodies of ESA-listed marine mammals and sea turtles feeding in the action area, increasing the potential for and rate of adverse health effects for these species. The occurrence and magnitude of exposure and adverse effects that we expect to result from the discharge of potentially harmful contaminants is uncertain, partly because levels of some POPs and other potentially harmful contaminants in the effluent have not been extensively monitored, and partly because the various individuals' potential exposure and response are highly variable.

In order to address this uncertainty, the proposed action includes initiation of special studies that we expect will begin to monitor and describe the discharge of some of these potentially harmful contaminants. As this information is collected in the future, we expect to be better able to assess the relative impact and contribution of Point Loma's discharge to increasing contaminant levels of ESA-listed species. Given what is already known about the harmful nature of these constituents described in this opinion, we also expect that these monitoring efforts will encourage the EPA and Point Loma to investigate measures to minimize the discharge of potentially harmful contaminants during future permit actions.

As described in Section 2.4 (Environmental Baseline) and Section 2.5 (Effects of the Action), ESA-listed marine mammals and sea turtles that may occasionally occur in the action area are susceptible to diminished health, reduced fitness, and even mortality, from exposure to HABs, including HABs that may occur in the action area. As described in Section 2.5 (Effects of the Action), the proposed action increases the probability of HABs occurring within the action area, increasing the probability of diminished health, reduced fitness, and even mortality, of ESA-listed marine mammals and sea turtles that occasionally occur within the action area. We do not have a precise understanding of how much Point Loma's discharge may increase the probability of HABs in the action area, or a way to assess if particular blooms are associated with the proposed action and the nutrient input created by Point Loma's discharge. The proposed permit includes monitoring requirements for the influent, effluent, and receiving waters, the intent of which is to increase our understanding of the nitrogen dynamics and point loading that result from Point Loma's discharge and relate that knowledge to HAB occurrence in the action area. The proposed permit also includes a special study of nutrients in the area near the PLOO that may help illuminate the relationship between HABs and nutrient inputs in the action area.

Due to uncertainty associated with these two potential avenues for adverse effects at an individual level, we are also uncertain as to the relative occurrence and magnitude of these adverse effects at the population level for the ESA-listed marine mammals and sea turtles that may be exposed to the proposed action. As described in Section 2.5 (Effects of the Action), we generally expect that exposure will be limited to relatively few individuals (adults or juveniles) or small portions of these populations over the duration of permit. Exposure is more likely for individuals that may have some preference for or site fidelity to the action area. Although there is uncertainty in the specific extent of population level exposure, at this time we generally do not anticipate widespread effects across populations that could potentially produce reduced productivity or fitness at a population level for any of these species.

As described in Section 2.4 (Environmental Baseline) and Section 2.6 (Cumulative Effects), we anticipate that most of the factors that have been affecting the quality and health of the environment within the action area are likely to continue into the future over the duration of the proposed permit. The effects from these factors pose potential continuing threats to the health of ESA-listed marine mammals and sea turtles that may visit the action area and to the action area as a whole. Climate change could influence the migration and distribution of prey species, the relative exposure of various individuals and ESA-listed populations within the action area, and increase the probability and/or magnitude of HAB occurrence in the action area over time. However, these climate change effects are unlikely to factor into the 5-year proposed action time frame considered in this opinion.

There is substantial uncertainty in the specific occurrence and magnitude of expected effects based on the available information. Additional information is needed to support a better understanding of these potential effects and inform future analyses. For example, additional information is needed regarding: (a) the levels of POPs and other potentially harmful constituents in the discharge effluent and their effects on ESA-listed marine mammals and sea turtles, and (b) the effects of the discharge effluent on the frequency and extent of HABs within the action area that may harm ESA-listed marine mammals and sea turtles.

2.7.1.1 Blue Whale

Over the course of the proposed action, we anticipate that some individual blue whales may occasionally enter the action area and be harmed by the proposed action, especially during the summer months. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the PLOO is a gradient and overlap of listed species with the action area is a transitory in nature. Although the ENP stock of blue whales is relatively small (1,898 individuals), exposure to the proposed action will likely be limited to a small number of individuals and the population that may be affected constitutes only a small portion of the globally-listed blue whale species. At this time, additional information is needed to more fully evaluate the exposure of blue whales to Point Loma's discharge and the anticipated effects at an individual and population level. EPA's proposed permit requires monitoring and studies that would address key questions regarding the effects of Point Loma's discharge on the action area and any blue whales that may occur there. The data generated will support improved effects analyses in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that the EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits.

We do not expect the proposed action to reduce the likelihood of survival and recovery of blue whales, based on: (a) our current understanding of the action's potential effects even given the acknowledged uncertainties regarding the magnitude and intensity of those effects on the species' status; (b) the measures that have been proposed to address these uncertainties; and (c) the prospect of developing actions to minimize the effects in future consultations, using information gathered under these measures.

2.7.1.2 Fin Whale

Over the course of the proposed action, we anticipate that some individual fin whales may occasionally enter the action area and be harmed by the proposed action at any time during the year. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the PLOO is a gradient and overlap of listed species with the action area is a transitory in nature. The CA/OR/WA stock of fin whales is estimated to consist of 11,065 individuals, although exposure to the proposed action will likely be relatively limited to a small number of individuals, and the population that may be affected constitutes only a portion of the globally-listed fin whale species. At this time, additional information is needed to more fully evaluate the exposure of fin whales to Point Loma's discharge and the anticipated effects at an individual and population level. EPA's proposed permit requires monitoring and studies that would address key questions regarding the effects of Point Loma's discharge on the action area and any fin whales that may occur there. The data generated will support improved effects analyses in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current

permit cycle. When that information becomes available, we anticipate that the EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits.

We do not expect the e proposed action to reduce the likelihood of survival and recovery of fin whales, based on: (a) our current understanding of the action’s potential effects even given the acknowledged uncertainties regarding the magnitude and intensity of those effects on the species’ status; (b) the measures that have been proposed to address these uncertainties; and (c) the prospect of developing actions to minimize the effects in future consultations, using information gathered under these measures.

2.7.1.3 Humpback Whale; Mexico DPS

Over the course of the proposed action, we anticipate that some individual humpback whales may occasionally enter the action area and be harmed by the proposed action, especially during the spring, summer, and fall months. However, these fitness effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the PLOO is a gradient and overlap of listed species with the action area is a transitory in nature. Based on contaminant signatures described above, there are likely individual humpback whales that favor or frequent foraging sites in Southern California that could include the action area. These individuals will be at increased risk of diminished health and fitness, and even death. The Mexico DPS is estimated to consist of 6,981 individuals, but humpback whales in the action area would more likely consist of the other DPS (Central America DPS). However, this Mexican DPS could occur in the action area given their general migratory movements along the U.S. west coast.

At this time, additional information is needed to more fully evaluate the exposure of the Mexico DPS humpback whales to Point Loma’s discharge and the anticipated effects at an individual and population level. EPA’s proposed permit requires monitoring and studies that would address key questions regarding the effects of Point Loma’s discharge on the action area and any humpback whales that may occur there. The data generated will support improved effects analyses in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that the EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits. In addition, in the future NMFS will be developing further scientific information regarding the distribution of ESA-listed humpback whales. This information will support an improved understanding of the potential exposure of the Mexican DPS humpback whales to actions throughout their range, including specifically their presence and abundance in the SCB.

We do not expect the proposed action to reduce the likelihood of survival and recovery of the Mexico DPS of humpback whales, based on: (a) our current understanding of the action’s potential effects even given the acknowledged uncertainties regarding the magnitude and intensity of those effects on the species’ status; (b) the measures that have been proposed to address these uncertainties; and (c) the prospect of developing actions to minimize the effects in future consultations, using information gathered under these measures.

2.7.1.4 Humpback Whale; Central America DPS

Similarly to the Mexican DPS of humpback whales, we anticipate that some individual Central America DPS humpback whales may occasionally enter the action area and be harmed by the proposed action, especially during the spring, summer, and fall months. Based on contaminant signatures described above, there are likely individual humpback whales that favor or frequent foraging sites in Southern California that could include the action area. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the PLOO is a gradient and overlap of listed species with the action area is a transitory in nature. The Central America DPS is estimated to consist of 1,809 individuals, and they could occur in the action area given their general migratory movements along the U.S. west coast.

As described above for the Mexican DPS, additional information is also needed to more fully evaluate the exposure of the Central America DPS of humpback whales to Point Loma's discharge and the anticipated effects at an individual and population level. EPA's proposed permit requires monitoring and studies that would address key questions regarding the effects of Point Loma's discharge on the action area and any humpback whales that may occur there. The data generated will support improved effects analyses in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits. In addition, in the future NMFS will be developing further scientific information regarding the distribution of ESA-listed humpback whales. This information will support an improved understanding of the potential exposure of Central America DPS humpback whales to actions throughout their range, including specifically their presence and abundance in the SCB.

We do not expect the proposed action to reduce the likelihood of survival and recovery of the Central America DPS of humpback whales, based on: (a) our current understanding of the action's potential effects even given the acknowledged uncertainties regarding the magnitude and intensity of those effects on the species' status; (b) the measures that have been proposed to address these uncertainties; and (c) the prospect of developing actions to minimize the effects in future consultations, using information gathered under these measures.

2.7.1.5 Gray Whales; WNP DPS

Over the course of the proposed action, we anticipate that some individual WNP gray whales may occasionally enter the action area and be harmed by the proposed action during the winter and spring migrations each year. As described before, there is a small likelihood (less than 1% chance) that any individual gray whale that may enter the action area could belong to the WNP population of gray whales. It is likely that at least one or more WNP gray whales would enter the action area during the proposed action and be at risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals

only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the PLOO is a gradient and overlap of listed species with the action area is a transitory in nature. The WNP population of gray whales is very small (271 individuals), but exposure to the proposed action will likely be extremely limited given their migratory behavior through such a small action area, the limited number of WNP gray whales that may occur in the action area, and the limited potential for foraging to occur.

At this time, additional information is needed to more fully evaluate the exposure of WNP gray whales to Point Loma's discharge and the anticipated effects at an individual and population level. EPA's proposed permit requires monitoring and studies that would address key questions regarding the effects of Point Loma's discharge on the action area and any WNP gray whales that may occur there. The data generated will support improved effects analyses in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that the EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits.

We do not expect the proposed action to reduce the likelihood of survival and recovery of WNP gray whales, based on: (a) our current understanding of the actions' potential effects - even given the acknowledged uncertainties regarding the magnitude and intensity of those effects on the species' status; (b) the measures that have been proposed to address these uncertainties; and (c) the prospect of developing actions to minimize the effects in future consultations, using information gathered under these measures.

2.7.1.6 Guadalupe Fur Seals

Over the course of the proposed action, we anticipate that some individual Guadalupe fur seals may occasionally enter the action area and be harmed by the proposed action, especially during the summer months. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the PLOO is a gradient and overlap of listed species with the action area is a transitory in nature. The Guadalupe fur seal population is estimated to be at least 31,091 individuals (Carretta et al. 2021b), although exposure to the proposed action will likely be limited to a small number of individuals and a small portion of the population. At this time, additional information is needed to more fully evaluate the exposure of Guadalupe fur seals to Point Loma's discharge and the anticipated effects at an individual and population level. EPA's proposed permit requires monitoring and studies that would address key questions regarding the effects of Point Loma's discharge on the action area and any Guadalupe fur seals that may occur there. The data generated will support improved effects analyses in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that the EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits.

We do not expect the proposed action to reduce the likelihood of survival and recovery of Guadalupe fur seals, based on: (a) our current understanding of the action's potential effects even given the acknowledged uncertainties regarding the magnitude and intensity of those effects on the species' status; (b) the measures that have been proposed to address these uncertainties; and (c) the prospect of developing actions to minimize the effects in future consultations, using information gathered under these measures.

2.7.1.7 Green Sea Turtle; East Pacific DPS

Over the course of the proposed action, we anticipate that some individual East Pacific green sea turtles may be present in the action area and be harmed by the proposed action. As described above, we expect that some individual green turtles reside in or make frequent or extended visits to the action area. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the PLOO is a gradient and overlap of listed species with the action area is a transitory in nature. Although there are no estimates for the total abundance of East Pacific green sea turtle DPS, the number of nesting females in one of the primary nesting areas exceeds 11,000 individuals. We expect that exposure will be limited to only a small subset of individuals from the East Pacific DPS; however, green turtles are likely at an increased risk of exposure to the proposed action compared to other ESA-listed sea turtles, given their known occurrence in and around the action area.

At this time, additional information is needed to more fully evaluate the exposure of green sea turtles to Point Loma's discharge and the anticipated effects at an individual and population level. EPA's proposed permit requires monitoring and studies that would address key questions regarding the effects of Point Loma's discharge on the action area and any green sea turtles that may occur there. The data generated will support improved effects analyses in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that the EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits.

We do not expect the proposed action to reduce the likelihood of survival and recovery of green sea turtles, based on: (a) our current understanding of the action's potential effects even given the acknowledged uncertainties regarding the magnitude and intensity of those effects on the species' status; (b) the measures that have been proposed to address these uncertainties; and (c) the prospect of developing actions to minimize the effects in future consultations, using information gathered under these measures

2.7.1.8 Leatherback Sea Turtle

Over the course of the proposed action, we anticipate that some individual leatherback sea turtles may occasionally visit the action area and be harmed by the proposed action. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness

effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the PLOO is a gradient and overlap of listed species with the action area is a transitory in nature. While there are no estimates for the total abundance of leatherback sea turtles within the population that may occur in the action area, the number of annual nesting females in western Pacific has been recently estimated at 1,054. We expect that exposure will be limited to only a small number of individuals, constituting only a portion of the population that may be affected and a portion of the globally-listed leatherback sea turtle species, although there is concern that the western Pacific population is in a state of decline, at high risk of going extinct, and to date there is no sign of recovery. However, the overall risks of exposure of the population to this proposed action are relatively low, given that the SCB is not a primary foraging location for this species and the species is not known to show site fidelity to the SCB.

At this time, additional information is needed to more fully evaluate the exposure of leatherback sea turtles to Point Loma's discharge and the anticipated effects at an individual and population level. EPA's proposed permit requires monitoring and studies that would address key questions regarding the effects of Point Loma's discharge on the action area and any leatherback sea turtles that may occur there. The data generated will support improved effects analyses in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that the EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits.

We do not expect the proposed action to reduce the likelihood of survival and recovery of leatherback sea turtles, based on: (a) our current understanding of the action's potential effects even given the acknowledged uncertainties regarding the magnitude and intensity of those effects on the species' status; (b) the measures that have been proposed to address these uncertainties; and (c) the prospect of developing actions to minimize the effects in future consultations, using information gathered under these measures.

2.7.1.9 Loggerhead Sea Turtle; North Pacific Ocean DPS

Over the course of the proposed action, we anticipate that some individual juvenile North Pacific Ocean DPS loggerhead sea turtles may occasionally visit the action area and be harmed by the proposed action. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the PLOO is a gradient and overlap of listed species with the action area is a transitory in nature. While there are no estimates for the total abundance of North Pacific DPS loggerhead sea turtles that may occur in the action area, the total number of adult females in the population was recently estimated at around 7,000, and it is estimated that there are approximately 340,000 loggerhead sea turtles of all ages in the North Pacific. Our expectation is that the relative exposure of this population will be limited to a small number of individuals (juveniles) and a small portion of the DPS. At this time, additional information is needed to more fully evaluate the exposure of loggerhead sea turtles to Point Loma's discharge and the anticipated effects at an individual and population level. EPA's proposed permit requires

monitoring and studies that would address key questions regarding the effects of Point Loma's discharge on the action area and any loggerhead sea turtles that may occur there. The data generated will support improved effects analyses in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that the EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits.

We do not expect the proposed action to reduce the likelihood of survival and recovery of loggerhead sea turtles, based on: (a) our current understanding of the action's potential effects even given the acknowledged uncertainties regarding the magnitude and intensity of those effects on the species' status; (b) the measures that have been proposed to address these uncertainties; and (c) the prospect of developing actions to minimize the effects in future consultations, using information gathered under these measures.

2.7.1.10 Olive Ridley Sea Turtle

Over the course of the proposed action, we anticipate that some individual olive ridley sea turtles, most likely from Mexican nesting beach origins, may occasionally visit the action area and be harmed by the proposed action. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the PLOO is a gradient and overlap of listed species with the action area is a transitory in nature. While there is no specific estimate of abundance for the Mexican nesting beach population, the total abundance of olive ridleys in the eastern tropical Pacific exceeds one million individuals, which includes hundreds of thousands of individuals from the Mexican nesting beach population. We expect that exposure to the proposed action will be limited to a small number of individuals and a small portion of the population. At this time, additional information is needed to more fully evaluate the exposure of olive ridley sea turtles to Point Loma's discharge and the anticipated effects at an individual and population level. EPA's proposed permit requires monitoring and studies that would address key questions regarding the effects of Point Loma's discharge on the action area and any olive ridley sea turtles that may occur there. The data generated will support improved effects analyses in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits.

We do not expect the proposed action to reduce the likelihood of survival and recovery of olive ridley sea turtles, based on: (a) our current understanding of the action's potential effects even given the acknowledged uncertainties regarding the magnitude and intensity of those effects on the species' status; (b) the measures that have been proposed to address these uncertainties; and (c) the prospect of developing actions to minimize the effects in future consultations, using information gathered under these measures.

2.7.2. Abalone

As described in Section 2.2 Rangewide Status of the Species and Critical Habitat, white abalone have declined significantly throughout their range and face a high risk of extinction, primarily due to overfishing and the resulting low local densities. As described in Section 2.4 Environmental Baseline, the action area contains a few remaining wild white abalone, as well as captive-bred white abalone that have been outplanted for experimental studies. These white abalone have already experienced years of exposure to discharges from wastewater treatment plants (including Point Loma), stormwater runoff, and adjacent rivers. Continued discharges are expected over the five-year period, adding to the pollutant load to which the white abalone will be exposed. The white abalone will also continue to experience the effects of warming water temperatures and ocean acidification within the action area.

As described in Section 2.5 (Effects of the Action), the proposed action would result in continued exposure of white abalone to the effluent plume, potential accumulation of harmful contaminants, and to HABs that may occur more frequently and to a larger extent due to the discharge. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only.

In general, the levels of heavy metals and persistent contaminants that have been reported in Point Loma's effluent are lower than the levels found to significantly reduce survival, growth, and reproductive development in abalone. In addition, white abalone are likely to be exposed to highly diluted, low concentrations of these contaminants in the plume. For the more persistent contaminants, however, it is unknown how exposure may increase accumulation in individual white abalone and thus potentially adversely affect individual abalone. We note that the effects of many of the CECs and other potentially harmful contaminants in the effluent have yet to be evaluated, singly or in combination with one another. We expect that the continued discharge of effluent under the proposed action is likely to increase the uptake of potentially harmful contaminants by white abalone within the action area, but the degree of exposure and the effects on the health of individuals and the population as a whole are highly uncertain.

As described in Section 2.4 (Environmental Baseline) and Section 2.5 (Effects of the Action), we expect the discharge of effluent to increase the probability of HABs in the action area, but do not have information to assess if particular blooms are associated with the proposed action. We do not expect that all abalone in the action area will be exposed to all HABs that occur within the area. If oceanographic conditions expose abalone to a HAB, then there is a reasonable potential for some abalone to die. Based on the best available information on past effects and the distribution of abalone in the action area, we would expect any HAB-related mortality of abalone to consist of no more than a few individuals in a confined area, limiting the effects on the population and species as a whole.

In summary, the proposed action may adversely affect survival, growth, and reproductive development of white abalone within the action area, further exacerbating the risks of low density and reduced reproductive capacity for this population. These effects would be in addition to the ongoing effects of other discharges into the action area, warming water temperatures, ocean acidification, and other threats. Of particular concern is the potential for the

proposed action to cause persistent contaminants to accululate in white abalone, which can result in adverse effects on growth, reproduction, and survival, as well as the effect of the proposed action on the frequency and extent of HABs within the action area, which can result in increased risk of mortality for white abalone. Based on the best available information, we expect exposure to harmful contaminants in the plume to be limited to low concentrations; accumulation may vary for individual abalone and, given the low concentrations, may require a long period of time to reach levels that result in adverse effects on individual health. We also expect HAB-related mortality of white abalone to be limited to a few individuals in a confined area. We acknowledge the high degree of uncertainty regarding the specific occurrence and magnitude of expected impacts based on the available information.

Additional information is needed to better understand the potential effects of the effluent on white abalone and inform future analyses. This information includes: abundance and distribution of white abalone in the action area, levels of more persistent and potentially harmful contaminants in the effluent and their effects on white abalone, and the effects of the effluent on the frequency and extent of HABs within the action area that may harm white abalone. EPA's proposed permit requires monitoring and studies that would address key questions regarding the effects of Point Loma's discharge on the action area and white abalone within the area. The data generated will support improved effects analyses in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that the EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits.

We do not expect the proposed action to reduce the likelihood of survival and recovery of white abalone, based on: (a) our current understanding of the action's potential effects even given the acknowledged uncertainties regarding the magnitude and intensity of those effects on the species' status; (b) the measures that have been proposed to address these uncertainties; and (c) the prospect of developing actions to minimize the effects in future consultations, using information gathered under these measures.

2.8. Conclusion

After reviewing and analyzing the current status of the listed species, the environmental baseline within the action area, the effects of the proposed action, the effects of other activities caused by the proposed action, and the cumulative effects, it is NMFS's biological opinion that the proposed action is not likely to jeopardize the continued existence of blue whales, fin whales, Mexico DPS and Central America DPS humpback whales, WNP DPS of gray whales, Guadalupe fur seals, East Pacific DPS green sea turtles, leatherback sea turtles, North Pacific Ocean DPS loggerhead sea turtles, olive ridley sea turtles, or white abalone. No critical habitat has been designated or proposed for these species in the action area; therefore, none was analyzed.

2.9. Incidental Take Statement

Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. “Take” is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. “Harm” is further defined by regulation to include significant habitat modification or degradation that actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including breeding, spawning, rearing, migrating, feeding, or sheltering (50 CFR 222.102). “Harass” is further defined by interim guidance as to “create the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering.” “Incidental take” is defined by regulation as takings that result from, but are not the purpose of, carrying out an otherwise lawful activity conducted by the Federal agency or applicant (50 CFR 402.02). Section 7(b)(4) and section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this ITS.

2.9.1. Amount or Extent of Take

In the biological opinion, NMFS determined that incidental take is reasonably certain to occur as follows:

We anticipate that all individual ESA-listed marine mammals, sea turtles, and white abalone residing or feeding in the action area would uptake and/or accumulate potentially harmful contaminants including POPs such as organophosphate flame retardants. This uptake and/or accumulation would increase their body burden of these contaminants and the risk of incurring adverse effects on their growth, reproduction, and overall health and survival over a shorter period of time than would otherwise occur absent the proposed action. We expect all ESA-listed individuals that may enter or reside in the action area would be at increased risk of experiencing this effect, but we expect that adverse effects would generally be limited to relatively few individuals (adults or juveniles) from these populations.

We cannot further enumerate the anticipated take of ESA-listed species from the proposed action, due to uncertainty in the number of individuals that may be subject to exposure and uncertainty in the response and level of harm that would occur for individuals exposed from each ESA-listed species. Instead, we can describe the extent of take associated with the potential accumulation of potentially harmful contaminants by relating the extent of take to the amount of these potentially harmful contaminants being discharged into the action area by Point Loma. While there are many potentially harmful contaminants, our analysis focused on the apparently increasing threat associated with accumulation of organophosphate flame retardants, given the recent literature describing the potential harm organophosphate flame retardants can have on numerous ESA-listed species, and its known association with wastewater discharge in general. Consequently, we elect to use the extent of organophosphate flame retardant discharge as a surrogate to describe the extent of take associated with risks of increased contaminant levels for ESA-listed species as a result of the proposed action.

We have therefore quantified the potential incidental take of the proposed action in terms of the total loading of organophosphate flame retardants that we expect to be discharged by Point Loma. As we described in Section 2.5 (Effects of the Action), the levels of organophosphate flame retardants that are discharged by Point Loma have not been documented historically. Using available information from a similar adjacent WWTP, we estimated that Point Loma could discharge up to approximately 1898 pounds (839.5 kg) of organophosphate flame retardant discharge per year. These organophosphate flame retardants are released into the ecosystem and are potentially bioavailable for uptake into the food web and ESA-listed species. For the total five years of this proposed action, the incidental take, therefore, equates to the discharge of up to approximately 9490 pounds (4197.5 kg) of these organophosphate flame retardants for the permit cycle.

The proposed action includes development of special studies to evaluate the levels of CECs, including specifically these organophosphate flame retardants, in the effluent and mass loadings to the receiving water. Through this special study and the requirements placed upon Point Loma by the EPA, we expect Point Loma to be able to monitor the discharge of organophosphate flame retardants relative to the amount of their discharge that has been assumed and described above and to report the annual monitoring data to the EPA.

We also anticipate that all individual ESA-listed marine mammals, sea turtles, and white abalone residing or feeding in the action area would face increased risks of exposure to HABs because the frequency and/or extent of HABs is likely to increase. However, we expect that adverse effects would generally be limited to relatively few individuals (adults or juveniles) of these populations.

At this time, we cannot predict the precise extent that Point Loma's effluent discharge contributes to increased probabilities of HABs, or distinguish which HABs may be more or less associated or influenced by the additional nutrient input from Point Loma's discharge. Consequently, we cannot further enumerate the anticipated take of ESA-listed species from the proposed action. Instead, we can describe the extent of take associated with increased probabilities of harmful effects from exposure to HABs by relating the extent of the increased probability of HABs to the amount of nitrogen, specifically in the form of ammonia, that is being released into the action area. We elect to use the extent of ammonia discharged as a surrogate to describe the extent of take associated with risks of increased probability of HAB exposure for ESA-listed species as a result of the proposed action. We use ammonia because it has been routinely monitored in the effluent (on a weekly basis) in contrast to other forms of nitrogen. Ammonia is also estimated by Howard et al. (2014) as making up 92% of the nitrogen in effluents discharged by WWTPs in the SCB, and therefore likely accounts for the vast majority of nitrogen discharged by this WWTP.

We have therefore quantified the potential incidental take of the proposed action in terms of ammonia that we expect to be discharged by Point Loma. In 2017, the maximum recorded value of ammonia nitrogen concentrations in Point Loma WWTP's effluent was 44.5 mg/L (City of San Diego 2019). This value of ammonia nitrogen concentration is useful to calculate an expectation for ammonia discharge based on current flows that we can use as a take surrogate. Assuming that the wastewater treatment processes at Point Loma remain approximately the same

in their effectiveness at removing ammonia from the wastestream, an annual average discharge rate range of 140 and 240 MGD should result in an annual mass emission discharge rate up to between 8,610 and 14,760 metric tons per year of ammonia⁴.

As part of the proposed action, the EPA requires Point Loma to monitor the influent, effluent, and receiving waters for parameters that include the several forms of nitrogen (e.g., ammonia, nitrate nitrogen, nitrite nitrogen). As described above, our surrogate for the extent of take associated with risks of increased probability of HAB exposure is described in terms of the extent of nitrogen discharged as ammonia. Ultimately, through the required monitoring under the proposed permit, which includes analysis of the nitrogen species being treated and discharged by Point Loma, we expect Point Loma to be able to monitor nitrogen levels in the discharge and estimate the total loading of nitrogen to the action area relative to the amount that has been assumed and described above and to report the annual total nitrogen monitoring data to the EPA.

2.9.2. Effect of the Take

In the biological opinion, NMFS determined that the amount or extent of anticipated take, coupled with other effects of the proposed action, is not likely to result in jeopardy to the species or destruction or adverse modification of critical habitat.

2.9.3. Reasonable and Prudent Measures

“Reasonable and prudent measures” (RPM) are measures that are necessary or appropriate to minimize the impact of the amount or extent of incidental take (50 CFR 402.02).

1. EPA shall monitor, document, and report the extent of incidental take of ESA-listed species resulting from Point Loma’s discharge using the surrogates described in Section 2.9.1 of this biological opinion.

⁴ The following was used to convert 44.5 mg/L to pounds per day using the assumed range of discharge flow rates: [mg/L]*[MGD]*[8.34 pounds/gallon]. For converting pounds to metric tons, we used a 0.000454 pounds to metric tons conversion factor.

2.9.4. Terms and Conditions

In order to be exempt from the prohibitions of section 9 of the ESA, the Federal action agency must comply (or must ensure that any applicant complies) with the following terms and conditions. The EPA or any applicant has a continuing duty to monitor the impacts of incidental take and must report the progress of the action and its impact on the species as specified in this ITS (50 CFR 402.14). If the entity to whom a term and condition is directed does not comply with the following terms and conditions, protective coverage for the proposed action would likely lapse.

The following terms and conditions implement RPM 1:

1a. EPA shall require that Point Loma to collect the necessary data to determine levels of organophosphate flame retardants in the effluent and the loading of receiving water using sampling and analysis protocols that are consistent with or equivalent to those used in studies by other wastewater dischargers (as referred to in this opinion).

1b. EPA shall require that Point Loma collect the necessary data to determine the various forms and total nitrogen loading from the facility's discharge into the receiving water (e.g., ammonia, nitrate nitrogen, nitrite nitrogen and organic nitrogen). While the extent of nitrogen loading from Point Loma's discharge can be approximated by monitoring ammonia as described in section 2.9.1 (Amount or Extent of Take), a comprehensive monitoring of all nitrogen forms is needed to fully understand the nitrogen dynamics of Point Loma's discharge and the total nitrogen loading that results from Point Loma's discharge into the action area. These monitoring results would improve our understanding of the proposed action's contribution to nutrient loading and HABs in the action area. These monitoring results will also assist efforts by the EPA and Point Loma to investigate measures to minimize the discharge of nutrients that may increase the probability of HAB occurrence in the action area during future permit actions.

In order to support this, the EPA shall require Point Loma to increase the frequency of the effluent nitrate and organic nitrogen sampling to match the ammonia sampling (weekly in Table E-4 Effluent Monitoring) by an appropriate method in order to calculate a mass emission rate that accounts for and characterizes all forms of nitrogen in the effluent. Also, the EPA shall require Point Loma to measure nitrogen forms in receiving waters, to extent possible. These results will produce a more consistent and robust dataset that can be used in regional efforts (e.g. the Southern California Bight Regional Monitoring Program and the BEC-ROMS model). This monitoring increase may be achieved through development and implementation of a special study work plan required by a forthcoming NPDES permit modification, or other means determined appropriate by the EPA.

1c. In order to assess the contribution of the PLOO to algal bloom formation in the action area, the EPA shall require the applicant to conduct a multi-year study, approved by the EPA and NMFS, that measures and/or determines the seasonal depth of the euphotic zone in the area of the PLOO as it relates to the action area. This requirement may be achieved through development and implementation of a new special study work plan required by the NPDES permit, or other means determined appropriate by the EPA.

1d. EPA shall report the following to NMFS WCR within 180 days after the permit expiration date or at the time of permit renewal and consultation with NMFS: the estimated discharge of organophosphate flame retardants (pounds or kg) by Point Loma into the action area per year; the estimated total levels of ammonia and total nitrogen (pounds/tons or kg/metric tons) discharged by Point Loma into the action area per year; and results of the seasonal depth of the euphotic zone in the area of the PLOO.

EPA may require Point Loma to directly submit the report to NMFS, provided that the EPA also receives it. The report shall be submitted to the NMFS WCR Protected Resources Division's Long Beach Office Branch Chief (Penny Ruvelas) at the following addresses:

- Electronically: Penny.Ruvelas@noaa.gov
- By mail: NMFS West Coast Region
501 West Ocean Boulevard, Suite 4200
Long Beach, California 90802

1e. EPA shall notify NMFS WCR if Point Loma's estimated annual discharge of organophosphate flame retardants and/or estimated annual discharge of nitrogen (as ammonia) exceeds the total amounts/levels during the permit term that have been assumed and described above immediately when monitoring results indicate this has occurred.

2.10. Conservation Recommendations

Section 7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Specifically, "conservation recommendations" are suggestions regarding discretionary measures to minimize or avoid adverse effects of a proposed action on listed species or critical habitat or regarding the development of information (50 CFR 402.02).

Harmful Algal Blooms

The following conservation recommendations related to HABs in the action area would provide information for future consultations and address questions related to the effects of Point Loma's discharge on the frequency and extent of HABs in the action area and SCB.

1. EPA should support additional data collection within the action area and the SCB to help understand the potential influence on harmful algal bloom dynamics from Point Loma's discharge. This could include:
 - a) Studying the generation of nitrogen form, timing, and mass balance data from upwelling and stormwater runoff events in the coastal areas between La Jolla and Baja California (including areas off of Point Loma) to couple with the required generation of nitrogen data from Point Loma's discharge and feed into regional modeling efforts (e.g. Southern California Bight Regional Monitoring Program).
 - b) Assessing what HAB species are in the coastal areas between La Jolla and Baja California (including areas off of Point Loma); whether the HAB species are being maintained within the subsurface plume; and whether they are manifesting concurrently with *P. spp.* and high domoic acid levels, or if *P. spp.* tends to bloom first and therefore reduce the prevalence of other HAB species. This work may be conducted by Point Loma or through the five-year Regional Bight Monitoring Program that examines multiple WWTPs within the area.
 - c) Synthesizing the results from additional data collection, monitoring and/or evaluation can be provided to NMFS in a report or reports, submitted on a schedule to be determined.
 - d) Having the EPA ask Point Loma investigate the potential for denitrification at the Point Loma WWTP as flows to the facility are reduced by increased water recycling projects planned by the City. Decreased flows at Point Loma WWTP may allow for some areas of the WWTP to be repurposed for denitrification processes that would further reduce nitrogen loading to the receiving waters and potential impacts related to HABs.

2.11. Reinitiation of Consultation

This concludes formal consultation for NPDES permit (#CA0107409) reissuance for the Point Loma Wastewater Treatment Plant and Ocean Outfall.

Under 50 CFR 402.16(a): "Reinitiation of consultation is required and shall be requested by the Federal agency or by the Service where discretionary Federal agency involvement or control over the action has been retained or is authorized by law and: (1) If the amount or extent of taking specified in the incidental take statement is exceeded; (2) If new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not previously considered; (3) If the identified action is subsequently modified in a manner that causes an effect to the listed species or critical habitat that was not considered in the biological opinion or written concurrence; or (4) If a new species is listed or critical habitat designated that may be affected by the identified action."

In this biological opinion, we describe the extent of take of the proposed action in terms of the amount of potentially harmful contaminants discharged into the action area by Point Loma

WWTP, specifically the total loading of organophosphate flame retardants. We estimated that Point Loma WWTP discharges approximately 1898 pounds (839.5 kg) of organophosphate flame retardants (TCEP, TCPP, and TDCPP combined) into the action area each year. If Point Loma WWTP's discharge of these organophosphate flame retardants per year is determined to be greater than this estimate (through the special studies required by the EPA or other means), then we may determine that the extent of take of the proposed action that has been anticipated in this biological opinion has been exceeded.

We also describe the extent of take of the proposed action in terms of the amount of nutrients discharged into the action area by the Point Loma WWTP, specifically ammonia. We estimate that the facility discharges up to 14,760 metric tons per year of ammonia into the action area per year. If Point Loma's discharge of ammonia per year is determined to be greater than this estimate (through the monitoring required by the EPA or other means), then we may determine that the extent of take of the proposed action that has been anticipated in this biological opinion has been exceeded.

In addition to the extent of take, we identify numerous uncertainties regarding the exposure of ESA-listed species to the proposed action and the effects of this exposure. If an event or events transpire such that HABs in the action area are identified as causing significant harm and/or mortality to ESA-listed species, we may determine that the extent of take associated with Point Loma's potential contribution to HABs and resulting effects to ESA-listed species has been exceeded, pending available information about the HAB event or events. In addition, we recognize that the state of science continues to develop regarding contaminants, HABs, wastewater discharge, and ESA-listed species. We also expect additional information to become available through studies undertaken in association with the proposed action and conservation recommendations provided in this biological opinion. We will consider new information as it becomes available and, based on that information, may determine that the extent of take of the proposed action that has been anticipated in this biological opinion has been exceeded.

2.12. "Not Likely to Adversely Affect" Determinations

Under the ESA, "effects of the action" are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action (50 CFR 402.02).

In our analysis, which describes the effects of the proposed action, we considered 50 CFR 402.17(a) and (b). When evaluating whether the proposed action is not likely to adversely affect listed species or critical habitat, NMFS considers whether the effects are expected to be completely beneficial, insignificant, or discountable. Completely beneficial effects are contemporaneous positive effects without any adverse effects to the species or critical habitat. Insignificant effects relate to the size of the effect and should never reach the scale where take occurs. Effects are considered discountable if they are extremely unlikely to occur.

We do not anticipate the proposed action to adversely affect North Pacific right whales, sei whales, sperm whales, giant manta rays, Southern DPS green sturgeon, oceanic whitetip sharks, East Pacific DPS scalloped hammerhead sharks, Southern California DPS steelhead, or black abalone. This opinion does not consider effects on critical habitat because none is designated in the action area.

In our effects analysis, we identified three potential stressors to result from Point Loma's permitted discharge of wastewater: (1) uptake of pollutants from the water; (2) ingestion of prey that have accumulated pollutants; and (3) exposure to harmful algal blooms resulting from the discharge. In this section, we analyze each species, or species group, as applicable relative to all of these potential stressors.

2.12.1. Other ESA listed marine mammals

2.12.1.1 North Pacific Right Whales

North Pacific right whales are extremely rare in the action area: only 14 North Pacific right whales have been sighted off California since 1950 (NMFS 2017). As a result, we do not expect these whales to be exposed to any stressors deriving from the proposed action.

2.12.1.2 Sei Whales

Similarly, no sei whales have been sighted during dedicated marine mammal surveys by NMFS in Southern California since 1991 (Carretta et al. 2021a). Therefore, due to their rarity, we do not expect these whales to be exposed to any stressors deriving from the proposed action.

2.12.1.3 Sperm Whales

Sperm whales reach peak abundance in waters offshore California from April through mid-June, and from the end of August through mid-November (Carretta et al. 2021a). Sperm whales are typically found foraging offshore in deep waters and/or canyons and are more commonly sighted off central California. Sperm whales primarily prey on medium and large-sized squid (e.g., the giant squid) and fishes (e.g., sharks) that occupy deep ocean waters. Sperm whale occurrence in nearshore waters of Southern California is rare; no sperm whales have been sighted during dedicated marine mammal surveys by NMFS in Southern California since 1991 (Carretta et al. 2021a). Although the action area does include some areas of deep water canyons, sperm whales are very unlikely to forage in the action area and none of their probable prey would be directly connected to the food web in nearshore waters that are most likely to be impacted by wastewater discharge. As a result, we do not expect these whales to be exposed to any stressors deriving from the proposed action.

2.12.1.4 Conclusion for other ESA-listed marine mammals

Based on the above, we do not expect that any of these species would be exposed to any effects stemming from the proposed action. Given these circumstances, NMFS finds that the potential for the proposed action to affect these species is discountable and determines that the proposed action may affect but is not likely to adversely affect North Pacific right whales, sei whales, and sperm whales.

2.12.2. Giant manta ray

Giant manta rays are slow growing and highly migratory, with small, fragmented populations distributed throughout the world's oceans. They inhabit tropical, subtropical, and temperate waters and are commonly found offshore and in productive coastal areas, although they can also be observed in estuaries, inlets, bays, and intercoastal waterways. As filter feeders, they eat large quantities of zooplankton. The main threat to the species is targeted and incidental catch in commercial fisheries. NMFS listed the giant manta ray as threatened under the ESA in 2018 (83 FR 2916; January 22, 2018).

In the eastern Pacific, giant manta rays have been documented as far north as southern California; however, sightings in this area are sporadic (Miller and Klimovich 2017). In U.S. west coast fisheries, the species is occasionally observed as bycatch in the California drift gillnet fishery that targets swordfish and thresher sharks; however, they have only been observed in low numbers and only during El Niño events (Miller and Klimovich 2017). Observer records for this fishery from 1990-2006 contain only 14 documented observations of giant manta rays, with 36% of individuals released alive. These observations equate to an estimated (extrapolated) catch of 90 giant manta rays for the entire period (95% CI: 26 – 182; CV = 0.48) (Larese and Coan 2008). No giant manta rays have been observed in the California drift gillnet fishery since 2010 (data available from: <https://www.fisheries.noaa.gov/west-coast/fisheries-observers/west-coast-region-observer-program#data-summaries-and-reports>).

Although the action area is within the known range of the giant manta rays, it is located at the extreme northern end of their range. As discussed above, the species' presence off southern California has been rarely documented. Although the presence of giant manta rays in the action area is possible, the possibility of such an occurrence during the course of the proposed action is extremely unlikely. Consequently, the risks of exposure to the proposed action are very low.

Given these circumstances, NMFS finds that the potential for the proposed action to affect this species is discountable and determines that the proposed action may affect but is not likely to adversely affect giant manta rays.

2.12.3. Green sturgeon, Southern DPS

Green sturgeon are anadromous, long-lived, bottom-oriented (demersal) fish that range from the Bering Sea, Alaska (Colway and Stevenson 2007) to El Socorro, Baja California, Mexico (Rosales-Casián and Almeda-Jáuregui 2009). They are one of the most marine-oriented and

widely distributed of the sturgeon, spending much of their lives migrating between estuaries along the coast. Relatively little is known about how green sturgeon use coastal marine habitats. While in the ocean, they generally occur between 0 and 110 meters, spending most of their time in areas between 20 to 80 meters in depth (Erickson and Hightower 2007; Huff et al. 2011). While in marine waters, they may be feeding or simply migrating between estuaries.

In 2006, NMFS identified two DPSs of green sturgeon (71 FR 17757; April 7, 2006): a Northern DPS consisting of populations originating from coastal watersheds northward of and including the Eel River, and a Southern DPS consisting of populations originating from coastal watersheds south of the Eel River. NMFS listed the Southern DPS as threatened under the ESA, but determined that ESA listing was not warranted for the Northern DPS. Threats to the Southern DPS include the loss of access to historical spawning habitat, impaired spawning and rearing habitats, and fisheries bycatch.

Both Northern and Southern DPS green sturgeon make extended migrations along the coast and co-occur in marine waters. In general, green sturgeon presence is limited south of Monterey Bay. Documented records of green sturgeon (mainly from fishery interactions) indicate single records of green sturgeon encounters south of Monterey Bay in 1941 (between Huntington Beach and Newport), 1957 (just north of Point Vicente, Los Angeles County), 1991 (north of Santa Barbara), and 1993 (off San Pedro) (NMFS 2011). One green sturgeon was observed off Baja California in 2008, about 200 km south of Bahía de Todos Santos (Rosales-Casián and Almeda-Jáuregui 2009). We are not aware of any additional records of green sturgeon in the vicinity of the action area. Thus, while the available information indicates green sturgeon could occur in the action area, the reports are infrequent and speak to the rarity of the species in the area. None of the green sturgeon observed south of Monterey Bay were identified to DPS.

Based on the best available information, Southern DPS green sturgeon are likely extremely rare in Southern California, with a very low probability of occurring in the action area and being exposed to Point Loma's discharge effluent. The likelihood that Southern DPS green sturgeon would stay in this specific area for any length of time, thereby being exposed to potentially harmful effluent, is also low given the rarity of observations this far south in recent decades. Given these circumstances, NMFS finds that the potential for the proposed action to affect this species is discountable, and determines that the proposed action may affect but is not likely to adversely affect Southern DPS green sturgeon.

2.12.4. Oceanic whitetip shark

NMFS listed the oceanic whitetip shark as threatened under the ESA in 2018 (83 FR 4153; January 30, 2018). Oceanic whitetip sharks are long-lived, pelagic, surface-dwelling top predators with late maturation and low to moderate productivity. They are found in tropical and subtropical oceans throughout the world, including the Pacific Ocean, where they have declined by approximately 80 to 95% since the mid-1990s (Young et al. 2018). In the eastern Pacific Ocean, the species' range extends from southern California to Peru (Compagno 1984). Oceanic whitetip sharks typically occur offshore in the open ocean, on the outer continental shelf, or around oceanic islands in deep water greater than 184 meters (Young et al. 2018). The primary

threat to oceanic whitetip sharks is bycatch in commercial fisheries. Their tendency to remain at the surface makes them particularly susceptible to interactions with fisheries.

Although oceanic whitetip sharks can occur as far north as southern California, their distribution is concentrated farther south and in more tropical waters (Young et al. 2018). Observer data for fisheries along the U.S. west coast have never recorded any observed encounters with oceanic whitetip sharks (Young et al. 2018). For example, the California drift gillnet fishery operates off the U.S. west coast and targets swordfish and common thresher sharks. Observers for this fishery did not observe any oceanic whitetip sharks in 8,698 sets conducted over a 25-year period from 1990-2015 (Young et al. 2018). A review of more recent observer records also show no observations of oceanic whitetip sharks for this fishery from 2015-2021 (data available from: <https://www.fisheries.noaa.gov/west-coast/fisheries-observers/west-coast-region-observer-program#data-summaries-and-reports>).

Based on the rare occurrence of oceanic whitetip sharks off the southern California coast, we expect the likelihood of their presence in the action area to be extremely low. Given these circumstances, NMFS finds that the potential for the proposed action to affect this species is discountable, and determines that the proposed action may affect but is not likely to adversely affect oceanic whitetip sharks.

2.12.5. Scalloped hammerhead shark, Eastern Pacific DPS

The scalloped hammerhead shark can be found in coastal warm temperate and tropical seas worldwide. The scalloped hammerhead shark occurs over continental and insular shelves, as well as adjacent deep waters, but is seldom found in waters cooler than 22° C (Compagno 1984). It ranges from the intertidal and surface to depths of up to 450–512 meters (Klimley 1993), with occasional dives to even deeper waters (Jorgensen et al. 2009). It has also been documented entering enclosed bays and estuaries (Compagno 1984). Distribution in the eastern Pacific extends from the coast of Southern California, including the Gulf of California, to Ecuador and possibly Peru, to the offshore waters around Hawaii and Tahiti (Miller et al. 2014).

The 2014 Status Review Report (Miller et al. 2014) identified six DPSs of the worldwide scalloped hammerhead population. Four were listed under the ESA, including the Eastern Pacific DPS, which is listed as endangered largely due to existing threats associated with commercial fisheries catch and bycatch throughout the DPS (79 FR 38214; July 3, 2014). The Central Pacific DPS was not listed under the ESA due primarily to the relative lack of threats facing this DPS and the presence of productive pupping grounds in Hawaii (79 FR 38214; July 3, 2014). Abundance data from the eastern Pacific are limited, but available information suggests that the Eastern Pacific DPS is declining (79 FR 38214; July 3, 2014). Although precise population estimates are not available in the eastern Pacific, estimates based on assumptions related to genetic and demographic parameters have been made for populations in Baja and Pacific Panama that suggest combined totals in these two populations is at least in the tens of millions (Duncan et al. 2006; Miller et al. 2014).

Although the action area is within the known range of the Eastern Pacific DPS of scalloped hammerhead sharks, it is located at the extreme northern end of their range and their presence

anywhere off California has been only rarely documented. To date, no scalloped hammerheads have been documented as captured in fisheries along the U.S. west coast (NMFS 2015). Although scalloped hammerheads could occur in the action area it is extremely unlikely given that scalloped hammerheads sharks favor warmer waters more often located in lower latitudes. Consequently, the risks of exposure to the proposed action are very low. Given these circumstances, NMFS finds that the potential for the proposed action to affect this species is discountable, and determines that the proposed action may affect but is not likely to adversely affect scalloped hammerhead sharks.

2.12.6. Steelhead, Southern California DPS

The Southern California Steelhead DPS was listed as an endangered species under the ESA in 1997 (62 FR 43937) and subsequently affirmed in 2006 (71 FR 834) and 2014 (79 FR 20802). The geographic range of this DPS extends from the Santa Maria River, near Santa Maria, to the California–Mexico border.

Southern California steelhead are categorized as “winter run” because adult migration from the ocean into freshwater rivers and streams generally occurs between December and April (Fukushima and Lesh 1998), arriving in reproductive condition and spawning shortly thereafter. Juvenile steelhead rear in freshwater for one to three years before migrating to the ocean (as smolts), usually in late winter and spring, and grow to reach maturity at age two to five before returning to freshwater to spawn. The timing of emigration is influenced by a variety of parameters such as photoperiod, temperature, breaching of sandbars at the river’s mouth and streamflow.

In the action area, we expect individual adult and juvenile steelhead to occasionally be present while migrating to/from known steelhead watersheds (San Diego, Sweetwater, Otay, and Tijuana rivers; Figure 10) tributary to the San Diego Region. Based on our understanding, their presence in the action area is expected to be intermittent and brief (hours to a few days). Juvenile steelhead rapidly migrate off-shore upon entering the ocean, swimming hundreds of miles from their natal river or stream (Light et al. 1989; Daly et al. 2014). Migration rates of juvenile steelhead in the marine environment, assuming constant movement in a straight line, have been reported to range from about 1 mile per day to as high as 26 miles per day (Daly et al. 2014). Two juvenile steelhead that migrated from a southern California river and detected (acoustic tag) offshore of northern California were estimated to migrate at a rate of 20 and 12 miles per day (Kelly 2012).

Additionally, adult and juvenile steelhead migrating through the action area in coastal waters are expected to primarily occupy the upper water column. Tagging studies on the vertical distribution of adult steelhead have shown that adult steelhead spend on average approximately 95% of the time within 20 feet of the ocean surface, and 72% of the time within 3 feet of the surface (Ruggerone et al. 1990). Juvenile steelhead also appear to primarily occupy the upper water (e.g., 3 feet from the surface) based on the prey species they consume (Percy et al. 1990).

Given their location in the very upper part of the water column steelhead would have limited exposure to the discharge plume. Because the Point Loma discharge point is well below the depth that juvenile and adult steelhead are reported to occupy, we expect that steelhead are

extremely unlikely to encounter the high effluent concentrations expected within the ZID. Any exposure of steelhead to the wastewater effluent would be limited to low concentrations and short duration, and would not be expected to result in lethal or sub-lethal effects. Also, the potential exposure duration would most likely be on the order of minutes rather than days based on steelhead swimming speed and distribution of the effluent plume, further reducing the likelihood of lethal or sub-lethal effects on steelhead.

Wastewater discharges of persistent bio-accumulative constituents (e.g., DDT, PCBs and flame retardants) can potentially cause lethal or sub-lethal effects on steelhead, by inhibiting growth or increasing disease susceptibility. These effluent constituents are likely concentrated in sea-floor sediments in the action area near the wastewater discharge point, well below the depth that juvenile and adult steelhead are reported to occupy. Although these constituents may be introduced to the upper water column via the food web and consumed by steelhead, the brief exposure (hours to a few days) is expected to limit accumulation of these constituents to levels below those sufficient to have lethal or sub-lethal effects on adult or juvenile steelhead.

In a study on the effects of PBDE exposure on disease susceptibility (Arkoosh et al. 2010), juvenile salmon were fed a diet containing various concentrations of PBDEs for 40 days and then exposed to a marine bacterial pathogen. The cumulative mortality during the 40-day feeding period of the three treatments did not exceed 2% in any treatment group, and there were no significant differences in cumulative mortality among the treatment groups. Although not statistically significant, a slight downward trend in weight and other condition factors was observed for fish fed PBDE diets relative to fish fed the control diet. Although this study demonstrates potential lethal or sub-lethal consequences from exposure to persistent bio-accumulative effluent constituents, the frequency and duration that adult or juvenile steelhead may be exposed to these constituents (i.e., hours to a few days) is not expected to be sufficient to result in lethal or sub-lethal effects.

The proposed action may promote the occurrence of HABs and result in potential indirect effects on steelhead. However, studies indicate that fish, including steelhead, are tolerant to domoic acid exposure, showing no neurological symptoms following oral exposure (circle, upside-down, and spiral swimming) (Lefebvre et al. 2007). Another consequence that may result from HABs is reduced or depleted dissolved oxygen in the water column, causing lethal or sub-lethal effects to fish (e.g., hypoxia). However, the occurrence of reduced or depleted dissolved oxygen appears to occur in confined basins or deeper in the water column where subsurface species and/or decomposition occurs. Ocean surface and near-surface dissolved oxygen is moderated through atmospheric exchange and mixing (wind and wave action). Because juvenile and adult steelhead are reported to occupy the near-surface water column, exposure to reduced or depleted concentrations of dissolved oxygen is not expected to occur.

Thus, the proposed action is not expected to have either lethal or measurable sub-lethal adverse effects on steelhead. Based on this analysis, we find that potential adverse effects of the proposed action on Southern California steelhead are insignificant and determine that the proposed action may affect, but is not likely to adversely affect this species.

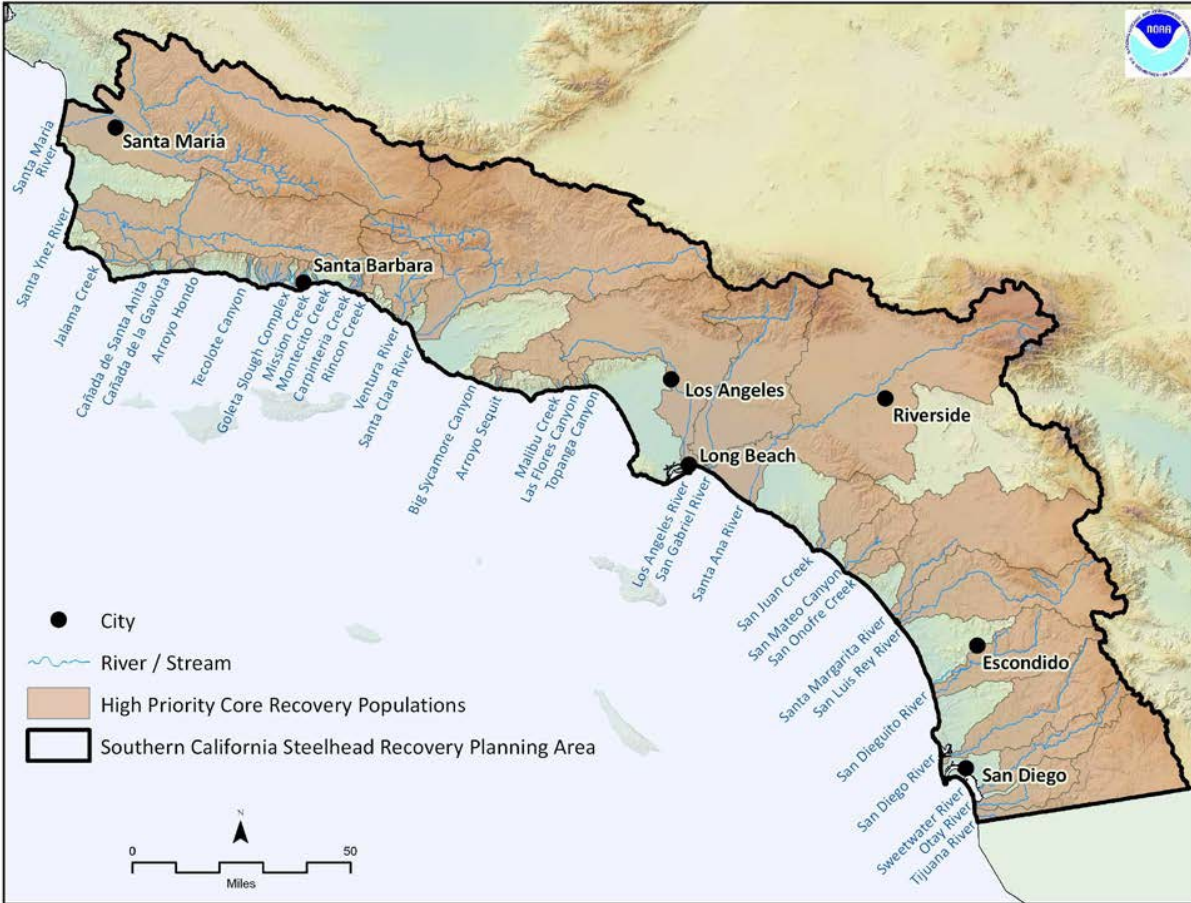


Figure 10. Southern California Steelhead Recovery Planning Area; maps shows the rivers and streams within the range of Southern California Steelhead (Figure 1-1 from Southern California Steelhead Recovery Plan (NMFS 2012a)).

2.12.7. Black abalone

Black abalone range from Point Arena, California, to Bahía Tortugas, Baja California, Mexico (74 FR 1937; January 14, 2009). Black abalone occupy rocky intertidal and subtidal reefs from the upper intertidal to six meters depth. They are most commonly observed in the mid to lower intertidal, in habitats with complex surfaces and deep crevices (Leighton 2005).

NMFS listed black abalone as endangered under the ESA in 2009 (74 FR 1937; January 14, 2009), primarily due to severe declines caused by a disease called withering syndrome. This disease resulted in mass mortalities and the loss of populations throughout southern and south-central California. All populations south of Monterey County declined in abundance by more than 80%; those south of Point Conception declined by more than 90%, and some have become locally extirpated (Neuman et al. 2010). Affected populations remain locally extirpated or persist at low densities, although a few populations have shown signs of recruitment (NMFS 2018b). Overall, black abalone remain in danger of extinction.

Although the action area is within the species' range and contains suitable rocky intertidal and shallow subtidal habitat, black abalone are not likely to occur within the action area. Data from long-term monitoring surveys show no records of black abalone within the area since at least 2005 (NMFS 2011). In 2015, surveys were conducted at four rocky intertidal sites along the shore between La Jolla and Point Loma and found no black abalone (Eckdahl 2015).

In addition, monitoring of the discharge plume indicates that dispersal is primarily alongshore to both the north and south of the outfall, with limited dispersal toward shore (City of San Diego 2021a). We expect any exposure of black abalone habitat to be limited to very diluted concentrations of the discharge plume, based on the depth of the outfall and distance from shore. Based on this, if black abalone were to be exposed to the plume, we expect the exposure to be minimal and unlikely to result in adverse effects.

As described in Section 2.5.3 of the Effects Analysis (Harmful Algal Blooms), Point Loma's discharge could contribute to and promote HABs within the action area. Given the location of suitable black abalone habitat relative to the location of the outfall, it is unlikely that black abalone would be exposed to HABs resulting from or promoted by Point Loma's discharge. Given these circumstances, NMFS finds that the potential for the proposed action to affect this species is discountable, and determines that the proposed action may affect but is not likely to adversely affect black abalone.

3. MAGNUSON–STEVENS FISHERY CONSERVATION AND MANAGEMENT ACT ESSENTIAL FISH HABITAT RESPONSE

Section 305(b) of the MSA directs Federal agencies to consult with NMFS on all actions or proposed actions that may adversely affect EFH. Under the MSA, this consultation is intended to promote the conservation of EFH as necessary to support sustainable fisheries and the managed species' contribution to a healthy ecosystem. For the purposes of the MSA, EFH means "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity", and includes the physical, biological, and chemical properties that are used by fish (50 CFR 600.10). Adverse effect means any impact that reduces quality or quantity of EFH, and may include direct or indirect physical, chemical, or biological alteration of the waters or substrate and loss of (or injury to) benthic organisms, prey species and their habitat, and other ecosystem components, if such modifications reduce the quality or quantity of EFH. Adverse effects on EFH may result from actions occurring within EFH or outside of it and may include site-specific or EFH-wide impacts, including individual, cumulative, or synergistic consequences of actions (50 CFR 600.810). Section 305(b) of the MSA also requires NMFS to recommend measures that can be taken by the action agency to conserve EFH. Such recommendations may include measures to avoid, minimize, mitigate, or otherwise offset the adverse effects of the action on EFH [CFR 600.905(b)].

This analysis is based, in part, on the EFH assessment provided by the EPA and descriptions of EFH for Pacific Coast Groundfish (PFMC 2020), coastal pelagic species (CPS) (PFMC 2021), and highly migratory species (HMS) (PFMC 2018) contained in the fishery management plans developed by the PFMC and approved by the Secretary of Commerce.

3.1. Essential Fish Habitat Affected by the Project

The proposed project occurs within EFH for various federally managed fish species within the Pacific Coast Groundfish, CPS, and HMS Fishery Management Plans (FMPs). In addition, the proposed project occurs within, or in the vicinity of, rocky reef and canopy kelp habitats, which are designated as habitat areas of particular concern (HAPCs) for various federally managed fish species within the Pacific Coast Groundfish FMP. HAPCs are described in the regulations as subsets of EFH that are rare, particularly susceptible to human-induced degradation, especially ecologically important, or located in an environmentally stressed area (or some combination of those factors). Designated HAPCs are not afforded any additional regulatory protection under MSA; however, Federal projects that may adversely impact HAPCs are more carefully scrutinized during the consultation process.

3.2. Adverse Effects on Essential Fish Habitat

Point-source discharges from municipal sewage treatment facilities (i.e., wastewater discharge) or storm water discharges can adversely affect EFH by: (1) reducing habitat functions necessary for growth to maturity; (2) modifying community structure; (3) bioaccumulation; and (4) modifying habitat. At certain concentrations, wastewater discharge can alter ecosystem properties, including diversity, nutrient and energy transfer, productivity, connectivity, and species richness (Hanson et al. 2003). These discharges can impair life functions among finfish, shellfish, and related organisms (e.g., growth and egg development, visual acuity, swimming speed, equilibrium, feeding rate, response time to stimuli, predation rate, photosynthetic rate, spawning seasons, migration routes, and resistance to disease and parasites). Point-source discharges may affect the growth, survival, and condition of EFH-managed species (and their prey species) if high levels of contaminants (e.g., chlorinated hydrocarbons, trace metals, PAHs, pesticides, and herbicides) are discharged. If contaminants are present, they may be absorbed across the gills or concentrated through bioaccumulation as contaminated prey is consumed (Raco-Rands 1996).

As described above in Section 1.3.2 (Permitted Effluent Limits), the NPDES permit establishes effluent limitations for the following nine pollutants/parameters: TSS, BOD, chlorine, oil and grease, settleable solids, chronic toxicity, pH, turbidity, and aldrin. Additional pollutants are monitored in the effluent (e.g., metals such as arsenic, mercury, lead, copper, chromium, and cadmium; ammonia; endosulfan; hexachlorocyclohexane; phthalate; ethylbenzene; toluene; benzene; PAHs; PCBs; and vinyl chloride); however, the permit states that these additional pollutants are not WQBELs for these parameters. The monitoring of these additional pollutants is used only for informational purposes and are not compliance determinations. In the following sections, we evaluate the adverse effects on EFH from pollutants discharged from the PLOO and discuss ongoing studies to better understand those effects and efforts to address them, such as compliance with existing water quality standards. Much of the information used in this effects analysis was taken from EFH Assessments provided by the EPA for this and other similar wastewater treatment projects, as well as the EPA's Technical Decision Document for the existing Point Loma Wastewater Treatment Plant NPDES permit.

3.2.1. Metals

Metals are known to bioaccumulate in marine organisms and can cause a variety of chronic health problems and physical anomalies at elevated concentrations. To both evaluate whether trace metals (and toxic organic compounds) are found at elevated concentrations in the area of the outfall and identify trends, EPA examined sediment monitoring data for pre-discharge (1991-1993) and discharge monitoring surveys (1994-2013) conducted during July, at the depth of the outfall along the 98 meter contour. EPA's review included the following ten metals: arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc. Concentrations of the analyzed metals were compared with non-regulatory sediment quality guidelines developed by NOAA. These sediment quality guideline concentrations represent the 10th percentile, or Effects Range-Low (ERL), and 50th percentile, or Effects Range-Median (ERM), of a toxicological effects database that has been compiled by NOAA for each parameter. The ERL is indicative of the concentrations below which adverse effects rarely occur, and the ERM is representative of the concentrations above which effects frequently occur. Mean concentrations of all metals with an established ERL were below that threshold (selenium lacks an ERL threshold). In addition, with the exception of silver in 2006, all ten metals were either below, or comparable to, average background levels for mid-depth sediments summarized for the 2008 Southern California Bight survey. As a result, the Point Loma Wastewater Treatment Plant is meeting its water quality objectives in accordance with the California Ocean Plan with regard to trace metals.

In addition, to evaluate contaminant levels in fish near the PLOO, samples are taken from rockfish and flatfish at various distances from the outfall and at reference stations to the north and south. Concentrations of metals in liver and muscle tissue of these fish are consistent with levels detected throughout the SCB, and there is no apparent relationship between metal levels and proximity to the outfall. Moreover, the existing permit maintains effluent monitoring requirements for metals (and other contaminants) to ensure that the Point Loma Wastewater Treatment Plant complies with permit effluent limitations, performance goals, and other permit requirements to achieve water quality standards in the receiving water. Performance goals (e.g., six-month median, maximum daily, instantaneous maximum) are based on the California Ocean Plan objectives for the protection of marine aquatic life.

3.2.2. Toxicity

Various pollutants, including ammonia, pesticides, petroleum-based contaminants, and metals, can adversely affect EFH through acute (i.e., lethal) or chronic (i.e., sublethal) toxicity (Hanson et al. 2003). Initial dilution, the process that causes the rapid and irreversible turbulent mixing of wastewater with ocean water around the point of discharge, is rapid and energetic, with timescales of seconds to minutes. The EFH Assessment asserts that any changes in the physical or chemical properties in surrounding ocean waters is limited to the immediate vicinity of the outfall due to strong local currents and high initial dilution (>200:1).

Under its existing NPDES permit, the City conducts the required effluent monitoring for the priority toxic and non-conventional pollutants listed in Table B of the California Ocean Plan and

“remaining priority pollutants.” The City submitted effluent data for metals, ammonia, and toxic organic chemicals from 2009 through 2013 as part of the most recent permit application process. Applying the minimum monthly average initial dilution value of 204:1, the Point Loma Wastewater Treatment Plant achieved 100% compliance with applicable State water quality standards and the EPA’s water quality criteria for toxics

To assess and protect against impacts caused by the aggregate toxic effect of the discharge of pollutants and the toxic effect of individual chemicals without water quality criteria, whole effluent toxicity tests are employed in a laboratory. These tests expose sensitive organisms to effluent concentrations and assess any impacts on mortality, growth, or reproduction. Test organisms are usually early life stages of surrogate organisms representative of those found in the environment. Because the EPA has determined that no species or test method is always the most sensitive, the proposed permit requires re-screening of the standard test species every two years to ensure the most sensitive test species is used in evaluating the toxicity of the effluent. The City conducted sensitivity screening tests and determined that red abalone (*Haliotis rufescens*) and giant kelp (*Macrocystis pyrifera*) were the most sensitive organisms for chronic toxicity testing. Test results from 2009-2015 using the minimum monthly initial dilution value of 204:1 revealed no exceedances of the chronic toxicity objective for red abalone, and only three exceedances for giant kelp, which the EPA notes is a very low failure rate. Follow-up toxicity testing in response to the exceedances demonstrated compliance. The existing permit maintains chronic toxicity effluent limitations, and specific requirements (e.g., additional testing, source identification, and measures to reduce toxicity, as appropriate) if the effluent limitation for chronic toxicity is exceeded.

For acute toxicity testing, topsmelt (*Atherinops affinis*) were deemed the most sensitive organism. Again using the minimum monthly initial dilution value of 204:1, test results from 2009-2015 show no exceedances of the acute toxicity objective. Thus, test results provided by the City demonstrate that the Point Loma Wastewater Treatment Plant has been compliant with permit standards for both acute and chronic toxicity. It is worth noting that Permit Order No. R9-2017-0007, effective October 1, 2017, removed performance goals and monitoring requirements for acute toxicity since chronic toxicity is more stringent and a chemical at low concentration could have chronic effects but no acute effects. As a result, the EPA retained effluent limitations for chronic toxicity only.

3.2.3. Nutrients and HABs

As described above in Section 2.4.1.1.4 (Harmful Algal Blooms, in the Environmental Baseline) and Section 2.5.3 (Harmful Algal Blooms, in the Effects Analysis), nutrient loading can cause increased plant and algal growth leading to eutrophication and increased instances of HABs.

As noted previously, *P. spp.* are domoic acid-producing diatoms that are the most frequently noted HAB species in the action area. Domoic acid is a water soluble neurotoxin that accumulates in shellfish and planktivorous fish such as anchovy and sardine (Lefebvre et al. 2007, 2012; Smith et al. 2018). Although the effects on piscivorous birds and marine mammals are well documented and wide spread (Schnetzler et al. 2013; Smith et al. 2018), impacts on other species are less certain. Behavioral or schooling impacts on fish species are not believed to be

occurring and laboratory work has shown that fish species ingesting domoic-acid-producing phytoplankton seem to be able to isolate and eventually depurate the compound (Lefebvre et al. 2007, 2012). It is unknown if there is a metabolic cost to this process for the fish. When domoic acid was microinjected into zebrafish eggs, the study did not find any effects on egg hatching and development (Vasconcelos et al. 2010). However, we could not readily find any studies reporting effects on fish egg or larval development under realistic exposure scenarios in our literature search. Vasconcelos et al. (2010) also reported that most studies showed little or no effects to survival or reproduction to several species of mollusks and crustaceans. Yet Liu et al. (2007) found significantly compromised growth and survival of king scallop larvae at environmentally realistic exposures to domoic acid. Further research on potential effects to wildlife under realistic environmental concentrations and conditions seems warranted.

The subsurface, HAB-prone dinoflagellate *A. tamarensis* complex produces saxitoxins which have been implicated in numerous fish kills and toxicity determinations (Gosselin et al. 1989; Lefebvre et al. 2004; Kudela et al. 2010; Trainer et al. 2010; Backer and Miller 2016). *A. cantenella* is the predominant PSP toxin producing species in the CA Current system, and the State of Washington has experienced numerous shellfish closures due to the presence of saxitoxin in the environment (Trainer et al. 2010; Trainer and Hardy 2015). Vasconcelos et al. (2010) reviews several studies that found effects of saxitoxin on crustacean larvae ranging from lethality among brine shrimp to sublethal effects on crab larvae. In addition to numerous study references indicating toxicity to winter flounder, red sea bream and Japanese anchovy, Gosselin et al. (1989) found heavy mortality among capelin and Atlantic herring larvae and juveniles exposed to environmentally realistic concentrations of *A. tamarensis* complex species through both vectorial poisoning and direct intoxication. Lefebvre et al. (2004) conducted a dietary uptake experiment with zebrafish larvae. They found that 24 hour exposures to high levels of saxitoxin induced paralysis had morphological impacts. Reduced growth rates were also prevalent resulting in depressed cumulative survival compared to control fish (40% vs 80%). A follow-up study with dissolved saxitoxin by Lefebvre et al. (2005) was conducted on larval Pacific herring. At levels greater than 47 µg/L, the saxitoxin caused significant reductions in sensorimotor function within one hour. Interestingly, the effect was transient, lasting only a few hours indicating there may be significant variability in species effects. However, impacts on the early life stages of Pacific herring are still likely due to an inability to avoid predators or escape the hypoxic conditions caused by significant HABs. There is a paucity of data on extracellular saxitoxin concentrations (Lefebvre et al. 2008), but the patchiness of HABs and the highly variable release of toxins when the cells lyse could produce localized high levels in the field.

Although *A. cantenella* is present along the entire outer open coast, incidents of saxitoxin effects to fish species or EFH specific to the SCB are not prevalent in the literature. Recent monitoring using a passive sampling technique (solid phase adsorption toxin tracking, or SPATT) has detected the simultaneous presence of multiple toxins along the coast, including domoic acid and saxitoxin (Smith et al. 2019). It may be that much of the detection effort is focused on the marine mammals and birds impacted by domoic acid, and any saxitoxin toxicity to federally managed fisheries or their prey species is being missed. Both Lefebvre et al. (2004) and Gosselin et al. (1989) theorized that finfish stocks could be significantly impacted if their larval and early juvenile life stages coincided with saxitoxin generating HABs. More monitoring of algal species distribution and occurrence needs to occur to determine if multiple species and/or toxins

routinely overlap in the area offshore of South San Diego and in the SCB as a whole and, if so, what impact that co-occurrence may be having.

As mentioned previously, *L. polyedrum* is another dinoflagellate that is frequently associated with red tide events in the CA Current south of Santa Cruz (Trainer et al. 2010), and blooms can occur outside of the upwelling season (Kudela et al. 2010). It can produce yessotoxins, and this large family of toxins has been identified as the major causative agent in the largest invertebrate mass mortality event recorded in coastal Northern California, in Sonoma County in 2001, which impacted red abalone, sea urchins, and crab species from Bodega Bay to Anchor Bay (De Wit et al. 2014b). Similar to all dense HABs, its effect on EFH likely comes from impacts to dissolved oxygen levels at the scale of the algal bloom resulting in fish kills (Trainer et al. 2010; Anderson et al. 2012; Backer and Miller 2016) and presumably impacts on other species that cannot escape the HAB area. Algal masses are known to rapidly deplete available dissolved oxygen in the water column due to high respiration by the algae or increased respiration by bacteria during algal decay, and this decrease can potentially cause hypoxic levels for periods of time (Booth et al. 2014; Booth 2015; Backer and Miller 2016). HAB biomass is believed to be contributing to the overall decline of dissolved oxygen levels in coastal waters (Capone and Hutchins 2013; Booth et al. 2014; Booth 2015; McLaughlin et al. 2017).

The *P. spp.* are also known to flocculate and form masses large enough to sink to the ocean floor, carrying domoic acid with them which may be ingested by benthic species and thus spreading the toxin within the benthic food web (Schnetzer et al. 2007, 2013; Trainer et al. 2010; Smith et al. 2018, 2021). Rapid transport is likely due to subduction by eddies, and there may be a significant topographic influence in the SCB (Kessouri et al. 2020a) leading to benthic hot spots. The SCB 2018 Regional Marine Monitoring Program found widespread domoic acid contamination in the sediments of the SCB with significant detections on the mid-shelf area (67% of the area) including offshore of southern San Diego County and the area adjacent to the Point Loma Wastewater Treatment Plant (Smith et al. 2021). In all, the toxin was detected in 54% of the SCB shelf habitats sampled. Sediment domoic acid concentrations ranged from 0.57 to 168.0 ng/g sediment over two years of sampling and two different, but similar, detection methods. It is unclear if these concentrations are having direct effects on benthic species in the SCB. Marine worms were found to have high levels of contamination compared to other benthic infauna. This reservoir of domoic acid poses a risk of being transferred into the food web and thereby affecting fish species managed under the MSA.

Anthropogenic nutrient sources, particularly wastewater effluent discharged at ocean outfalls, are significant at a local sub-regional spatial scale, which is ecologically relevant since it is consistent with the scales at which algal blooms develop (Howard et al. 2012, Nezlin et al. 2012). The physical oceanography in the vicinity of the PLOO discharge influences the fate and transport of the nutrients and any subsequent phytoplankton or zooplankton that utilize the nutrients to grow. Given that the SCB experiences frequent eddy patterns due to the CA Current and the Southern CA Counter Current found just offshore of the shelf area during certain times of the year (Howard et al. 2012b), nutrients from the PLOO discharge may remain in the area offshore of South San Diego for an appreciable amount of time. South San Diego has been identified as a nearshore bloom “hot spot” area with longer residence times and higher CHL- α levels (Trainer et al. 2010; Nezlin et al. 2012; Smith et al. 2021). Additional nutrients may enter

the South San Diego area from the south due to the Southern CA Counter Current (Howard et al. 2012b) and from winter runoff.

Information specific to the impacts HABs may have on EFH in the action area is lacking, though other studies of HABs have noted mechanical damage to fish gills and shading impacts to other species of phytoplankton or even sea grass beds (Anderson et al. 2012; Backer and Miller 2016). CPS impacts may be occurring when life stages of these species are incapable of escaping an area experiencing a HAB, likely due to impacts from depressed dissolved oxygen levels. It is also unknown if benthic habitat that supports Pacific Coast Groundfish species is being impacted when HABs die-off and sink within the action area. Although these conditions are transient in nature, their apparent increasing frequency and severity is cause for concern. Monitoring for toxins in the water column other than domoic acid does not seem to be occurring (or is not being published), and therefore it is unknown if algal species that produce ichthyotoxins are impacting EFH in the action area.

The City acknowledges that the PLOO discharge is contributing nutrients to coastal waters, but notes there is no direct evidence that these nutrient loads are causing HABs. However, the City recognizes further analysis and study are warranted and proposes supplemental monitoring to better understand the issue. Specifically, the City proposes to supplement weekly ammonia monitoring with a one-year effluent monitoring effort that includes monthly monitoring of the PLOO total nitrogen and phosphorus concentrations and computation of mass loads. The City would also use treatment models to assess the nutrient loads projected to occur when implementation of the Pure Water Project goes into affect. In addition, the City proposes to implement a special one-year supplemental receiving water study in which nitrogen, phosphorus and chlorophyll-*a* data are collected at selected offshore stations near the PLOO in conjunction with regular NPDES monitoring of stations near and surrounding the PLOO. This information should improve our understanding of nutrient discharges and HAB formation in the action area.

3.2.4. Dissolved Oxygen, Biological Oxygen Demand, and Total Suspended Solids

Aquatic organisms require sufficient levels of dissolved oxygen to breathe and grow. As a result, dissolved oxygen is an important measure of water quality and an indicator of a water body's ability to support aquatic life (see previous Section 3.2.3 for information on potential effects from HABs and nutrients on dissolved oxygen). Biological oxygen demand and total suspended solids both affect the level of oxygen in a receiving water, either directly or indirectly. Oxygen is depleted more rapidly with higher biological oxygen demand. Although the impact mechanisms are more indirect with respect to total suspended solids (e.g., decreased photosynthesis resulting from reduced light; increased water temperature, which holds less oxygen; etc.), elevated total suspended solids can also reduce dissolved oxygen levels. Despite interannual variability, actual mass emissions of total suspended solids have declined steadily since 1995. The permit includes effluent limits for biological oxygen demand (e.g., annual average system-wide percent removal of at least 58%) and total suspended solids (e.g., monthly average system-wide percent removal of at least 80%), which have both been met since 2008.

3.2.5. Contaminants of Emerging Concern and Persistent Organic Pollutants

As described above in Section 2.4.1.1.3 of the Environmental Baseline, the term CEC refers to several types of chemicals, including POPs, pharmaceuticals and personal care products, veterinary medicines, endocrine-disrupting chemicals, and nanomaterials. Wastewater effluent can be a major source of CECs, which can cause deleterious effects on aquatic life. For instance, organophosphate flame retardants are increasingly being used in place of PBDEs and may have similar effects as PBDEs, such as endocrine disruption and neurotoxicity that can negatively impact fish nervous systems, thyroid and liver functions, and endocrine and reproductive systems (Siddiqi et al. 2003). Crane et al. (2006) found that pharmaceuticals have the potential to adversely affect aquatic organisms, and recommended additional testing to better understand their acute and chronic effects on the natural environment.

A review of recent publications on environmental concentrations and aquatic toxicity of personal care products by Brausch and Rand (2011) noted that available information varied substantially depending upon the specific compound. According to their review, existing data indicate most personal care products are relatively non-toxic to aquatic organisms at anticipated environmental concentrations. However, many of these compounds are known endocrine disruptors that can have negative effects on fish, with some having a potential to cause estrogenic effects at relatively low concentrations (Brausch and Rand 2011). They also recommended additional studies be conducted on the potential toxicity of these substances to aquatic organisms, especially benthic invertebrates, algae, and vascular plants. In addition, Vajda et al. (2008) identified increased gonadal intersex (i.e., the presence of both male and female characteristics within the same fish), altered sex ratios, and other reproductive abnormalities in fish downstream of wastewater effluent with elevated concentrations of endocrine-disrupting chemicals, while no evidence of reproductive disruption was observed upstream of the site. It is worth noting this study was conducted within a more confined river system, and evaluating similar effects on the marine environment is more difficult due to greater dispersion and dilution rates (Reyes et al. 2012).

Reyes et al. (2012) evaluated the reproductive endocrine status of hornyhead turbot at locations near the coastal discharge sites of four large municipal WWTPs (including Point Loma) and at far-field reference locations in the region. Levels of estrogens and androgens measured in hornyhead turbot differed by location, but these differences could not be linked to ocean discharge locations for the four WWTPs. For example, higher concentrations of the active estrogen 17 β -estradiol (E2) were observed in male hornyhead turbot collected from both the PLOO and a reference site at Dana Point, while lower E2 concentrations were observed elsewhere, including sites adjacent to other WWTP discharges (Reyes et al. 2012). In addition, elevated levels of E2 did not appear to impair gonadal steroid production or its seasonality. Reyes et al. (2012) concluded that although some environment-associated differences in endocrine function were documented in hornyhead turbot in the study, there was no clear correlation to WWTP discharges.

CECs currently have no Clean Water Act regulatory standard (e.g., no established water quality standards and/or notification levels). However, the California SWRCB and Regional Water Quality Control Boards have identified monitoring strategies and sampling plans for CECs. The City (2021c) discusses POPs and other CECs and notes that effluent results for many POPs,

including PCBs, chlorinated pesticides, dioxins, and furans are at levels below detection limits and below the applicable numeric effluent limits as well as California Ocean Plan water quality objectives.

There are no routine monitoring data for other POPs and CECs to evaluate, though CECs have been detected previously in the PLOO discharge. However, special studies evaluating possible effects of CECs on fish exposed to the PLOO effluent have included one-time surveys of the discharge for representative compounds in the CEC category, including pharmaceuticals, personal care products, hormones, current use pesticides, and industrial/commercial products. These studies have produced somewhat mixed results. For instance, Vidal-Dorsch et al (2014) evaluated direct exposure of flatfish to CECs in effluent from the PLOO and Hyperion and found no significant biological responses in fish exposed to effluent compared to controls, even at concentrations 10 times those encountered in the environment near the PLOO. The study also observed that the concentrations of estrogenic compounds detected in the PLOO effluent was lower than the levels shown to cause estrogenic effects on other fish species and was a likely explanation as to why estrogenic responses were absent (Vidal-Dorsch et al. 2014).

A pathway analysis conducted by Vidal-Dorsch et al. (2013) demonstrated that the PLOO effluent exposure affected genes involved in metabolic pathways, while exposure to Hyperion effluent affected cellular processes and signal transduction. It is worth noting that full secondary treatment at Hyperion generally removed more CECs than the advanced primary treatment implemented at the PLOO, with one study reporting that the PLOO had significantly higher concentrations for 78% of the chemicals detected in effluents from Hyperion and the PLOO (Vidal-Dorsch et al. 2013, 2014). However, effluent mixtures can be complex such that even relatively low concentrations of individual chemicals can have an additive effect. For instance, Vidal-Dorsch et al. (2013) note that exposure to low levels of various individual estrogenic chemicals present in a mixture of chemicals can produce additive effects on fathead minnows. Thus, they conclude that a mixture of chemicals with similar effects pathways, rather than a single constituent, was the most probable cause for the observed responses (Vidal-Dorsch et al. 2013). In another study, Vidal-Dorsch et al. (2012) analyzed seawater collected near outfall discharges and reference stations for 56 CECs and detected fewer CECs and at lower concentrations at the reference station compared to outfall sites. Although effluent CEC concentrations were below known chronic toxicity thresholds, the study noted certain compounds can bioaccumulate in fish, that aquatic life toxicity thresholds have been developed for relatively few CECs, and suggested additional studies to better understand the effects of CECs in marine environments (Vidal-Dorsch et al. 2012).

The use of PBDEs is declining, and the 2018 Bight Regional Monitoring results showed non-detect or low levels of PBDEs in coastal sediment samples (EPA 2021). However, as mentioned previously, PBDEs are being replaced by organophosphate flame retardants that can have similar adverse effects on fish and have the potential to biomagnify in a marine food web (Siddiqi et al. 2003; Bekele et al. 2019). Organophosphate flame retardants were not included in previous PLOO monitoring efforts, but have been detected in the effluent for two other facilities in the SCB (Hyperion and TIWRP) (LASAN 2020). Given the potential adverse effects to EFH, NMFS is requesting that measurements of organophosphate flame retardants in the effluent and the

loading of receiving waters occur as described under the terms and conditions of this biological opinion.

City of San Diego (2021c) notes that although literature indicates evidence of CECs in some marine species, studies have not found negative impacts to sentinel marine monitoring species in the SCB. However, the City also acknowledges the list of CECs and POPs is large and growing, and that additional study is warranted due to the uncertainty that exists about the sources, concentrations, and effects of CECs and POPs in coastal waters. Efforts are underway to begin addressing this issue that are in addition to the monitoring required under the existing NPDES permit. For instance, as part of their Recycled Water Policy, the SWRCB is coordinating efforts to develop, validate, and accredit chemistry and bioanalytical tools for identifying and quantifying these compounds in wastewater, storm runoff, and the environment (SWRCB, 2018a, 2018b). To this end, the SWRCB convened a Science Advisory Panel, which developed a series of recommendations regarding CEC/POP monitoring and biological screening tools (Drewes et al., 2018). Consistent with these recommendations, the SWRCB is encouraging the use of existing bioassay tools in an effort to identify if bioactivity is occurring; and if so, isolate and identify the compounds that may be causing the harm. In concert with this effort, SCCWRP has initiated a comprehensive research plan (revised May 2019) for CECs that includes narrowing the field by screening a large array of CECs, diagnosing the likelihood of CEC effects, assessing CEC impacts *in situ* and providing critical information regarding source identification and tracking. This work can help determine what role, if any, wastewater outfalls may play as a source of CECs to the marine environment. The City proposes to participate in the effort by providing analysis of the PLOO effluent for a target number of CECs. The proposed approach is designed to provide data on the presence of the target compounds in the PLOO effluent, as well as add to the scientific community's research efforts. To accomplish this, the City intends to perform testing of the PLOO discharge for target CECs/POPs prior to and after the startup of Pure Water phase 1. The target compounds will be determined in consultation with the EPA, NMFS, SCCWRP and others to ensure coordination and to add value to their research efforts. NMFS believes that these efforts, combined with the studies required in the ESA terms and conditions, and ESA conservation recommendations, should substantially improve our ability to evaluate adverse CEC- and POP-induced EFH effects and the measures needed to address them.

3.2.6. Sediment Contamination

Many heavy metals and persistent organic compounds, such as pesticides and PCBs, tend to adhere to solid particles discharged from outfalls. As the particles are deposited, these compounds or their degradation products (which may be as toxic or more so than the parent compounds) can enter the EFH foodchain by bioaccumulating in benthic and pelagic organisms at much higher concentrations than in the surrounding waters (Stein et al. 1995). Due to burrowing, diffusion, and other upward transport mechanisms that move buried contaminants to the surface layers and eventually to the water column, pelagic and nektonic biota may also be exposed to contaminated sediments through mobilization into the water column. It is worth noting that low site fidelity for many target species in monitoring programs indicates that fish may be exposed to contaminants at multiple locations; thus, the location of capture is not a definitive indicator of the location of exposure (Ed Parnell et al. 2008; Burns et al. 2019).

However, the EFH Assessment states that organic enrichment of sediments due to the outfall discharge is not occurring beyond the ZID, and contaminant loading of sediments is not evident in the vicinity of the discharge.

Areas of sediment contamination are present within the action area. In particular, there is widespread contamination of DDT and PCBs at various sites throughout the SCB, much of which is a result of historical deposition and not associated with recent discharges from the PLOO. For instance, after evaluating various possible PCB sources, including wastewater discharge via ocean outfalls, natural dispersal from San Diego Bay, and surface runoff from local watersheds, Parnell et al. (2008) concluded the most probable source of PCB contamination in fish on the shelf off San Diego was the offshore disposal of dredged sediments from San Diego Bay. The EFH Assessment notes that concentrations of PCBs, PAHs, and DDT in sediments at Point Loma have been generally low. The notable exception is DDE, a byproduct of the pesticide DDT. However, DDE is found in sediments throughout southern California due to historical discharges, and levels of DDE at Point Loma are within the range of concentrations elsewhere in the SCB.

Despite these legacy contaminant issues, benthic habitat condition in the action area has improved substantially. For instance, the 2018 Bight Study concluded that sediment toxicity across the Bight was low, with a continued trend of decreasing toxicity and continued improvement in sediment quality (with the exception of marinas) (SCCWRP 2020). In addition, the City (City of San Diego 2021a) concludes that benthic habitats and associated macrofaunal communities on the continental shelf in the vicinity of the PLOO and throughout the PLOO monitoring region remain in good condition and are representative of natural communities (e.g., types, distribution, and populations of species) from similar habitats on the Southern California continental shelf. The Benthic Response Index results reveal little evidence of disturbance, and the EFH Assessment notes a balanced indigenous population of shellfish, fish, and wildlife exists immediately beyond the ZID. Other major community metrics (species richness, macrofaunal abundance, diversity, evenness, and dominance) show no evidence of impact or habitat degradation. Moreover, the results of all sediment toxicity testing conducted for the PLOO region during a 3-year pilot study (2016-2018), and during follow-up monitoring in the summer of 2019, did not indicate any evidence of toxicity at any of the monitoring sites. The existing NPDES permit contains sediment quality monitoring requirements, consistent with the California Ocean Plan, to assess physical and chemical properties of seafloor sediments, seafloor sediment toxicity to assess bioavailability and toxicity of sediment contaminants, and ecological status of the biological communities (benthos) that live in or on the seafloor sediments.

3.2.7. Impacts on Sensitive Habitats

Discharge sites may also impact sensitive habitats, such as kelp beds, if located improperly. For instance, high discharge velocities may cause scouring at the discharge point or entrain particulates and create turbidity plumes. These turbidity plumes of suspended particulates can reduce light penetration and lower the rate of photosynthesis and the primary productivity of an aquatic area while elevated turbidity persists. The contents of the suspended material can react with the dissolved oxygen in the water and result in oxygen depletion, or smother submerged aquatic vegetation sites including kelp beds and seagrass.

Kelp beds are present in the action area with the closest, the Point Loma kelp bed, located approximately 3.5 miles (5.6 km) inshore of the PLOO. Research and ongoing monitoring, including quarterly mapping of kelp beds in the area in accordance with regulations promulgated by the San Diego Regional Water Quality Control Board, conducted at the Point Loma kelp bed, has not found any impacts associated with the PLOO discharge. Instead, kelp forest community structure appears to be controlled by large-scale, low-frequency episodic changes in oceanographic conditions. The PLOO outfall discharges 4.5 miles (7.2 km) offshore at a depth of 320 ft (98 m) that was designed to prevent nearshore transport of the effluent.

Eelgrass habitat does exist within shallower regions near Point Loma, such as within the entrance channels to Mission and San Diego Bays, and east of Zuniga Jetty just offshore of Breakers Beach. However, the PLOO discharge is located more than 4.5 miles from the nearest known eelgrass beds. In addition, monitoring data rarely detect a potential plume at the most nearshore stations in the action area, which are located farther offshore and in deeper water than the eelgrass beds. Therefore, we expect any adverse effects on eelgrass from the proposed project would be minimal.

3.2.8. Cumulative Impacts

Wastewater discharges have historically been one of the largest contributors to pollutants in coastal waters, and contaminants released into the action area from the PLOO and other permitted discharges (e.g., South Bay Ocean Outfall) result in cumulative impacts on EFH. However, the EFH Assessment notes that mass emissions of most constituents in wastewater discharges have been substantially reduced coincident with increasingly stringent requirements over the last 40 years. The PLOO discharge has consistently complied with acute and chronic whole effluent toxicity tests that serve as surrogate measures to assess the cumulative effect of both regulated and unregulated compounds.

In addition to wastewater discharges, non-point sources, including natural dispersal from San Diego Bay and surface runoff from local watersheds (e.g., San Diego River and Tijuana River), can also contribute significant amounts of pollutants to the action area. However, as mentioned previously, recent ocean monitoring in the vicinity of the PLOO has found benthic habitats and associated macrofaunal communities to be in relatively good condition and representative of natural communities from similar habitats on the Southern California continental shelf. In addition, SCCWRP technical reports Volumes I (Sediment Toxicity) and II (Sediment Chemistry) for the Southern California Bight 2018 Regional Monitoring Program determined that the continental shelf areas had mostly low exposure to sediment contamination. Therefore, generally speaking, discharge prohibitions and other NPDES permit requirements, in combination with additional studies to better understand certain constituents or types of impacts (e.g., CECs/POPs, HABs), should help maintain adequate water quality in the action area.

3.3. EFH Adverse Effects Determination

Based upon the above effects analysis, NMFS has determined that the activities covered under the proposed action would adversely affect EFH for various federally managed fish species under the Pacific Coast Groundfish, CPS, and HMS FMPs due to impacts associated with the release of various contaminants into the action area. Adverse effects on EFH for species managed under the Pacific Coast Groundfish, CPS, and HMS FMPs associated with the proposed project would be primarily limited to the ZID, and to the influence of the discharge on HAB formation and prevalence in the action area.

Due to the high site fidelity of many species managed under the Pacific Coast Groundfish FMP (e.g., rockfish), they may be at risk of greater localized impacts from wastewater discharges relative to other fish species with a more dispersed, pelagic distribution, such as those managed under the CPS and HMS FMPs. However, as detailed in the above effects analysis, in general, mass emissions of most constituents continue to exhibit a declining trend, and benthic habitats and associated macrofaunal communities throughout the PLOO monitoring region are similar to natural communities on the Southern California continental shelf.

Moreover, the proposed action includes measures to avoid, minimize, or otherwise offset many of these adverse effects, including source control programs for toxic constituents, compliance with discharge permit requirements and water quality standards, outfall design to prevent nearshore transport of the effluent, and effluent discharge via a multi-port diffuser to reduce discharge velocities and pollutant concentrations at the point of discharge. In addition, where data gaps exist (e.g., nutrient inputs and HABs, CECs/POPs) special studies and/or additional monitoring have been proposed to increase the understanding of potential impacts associated with these constituents.

Therefore, as long as these measures are implemented, in addition to the measures spelled out in Section 2.9.4 (Terms and Conditions), we conclude that no additional measures are needed to avoid or minimize the adverse effects described in Section 3.2 (Adverse Effects on Essential Fish Habitat).

3.4. Supplemental Consultation

The EPA must reinitiate EFH consultation with NMFS if the proposed action is substantially revised in a way that may adversely affect EFH, or if new information becomes available that affects the basis for NMFS's EFH Conservation Recommendations [50 CFR 600.920(l)].

4. DATA QUALITY ACT DOCUMENTATION AND PRE-DISSEMINATION REVIEW

The Data Quality Act (DQA) specifies three components contributing to the quality of a document. They are utility, integrity, and objectivity. This section of the opinion addresses these DQA components, documents compliance with the DQA, and certifies that this opinion has undergone pre-dissemination review.

4.1. Utility

Utility principally refers to ensuring that the information contained in this consultation is helpful, serviceable, and beneficial to the intended users. The intended users of this opinion are the U.S. EPA and the California State Water Resources Board that jointly issues wastewater discharge permits in federal waters off the coast of California including the permit subject to this proposed action. Other interested users could include City of San Diego, other WWTPs that discharge into state and federal waters in California and elsewhere along the U.S. West Coast, as well as non-governmental organizations that monitor water quality issues in Southern California and beyond. Individual copies of this opinion were provided to the EPA. The document will be available within 2 weeks at the NOAA Library Institutional Repository [<https://repository.library.noaa.gov/welcome>]. The format and naming adhere to conventional standards for style.

4.2. Integrity

This consultation was completed on a computer system managed by NMFS in accordance with relevant information technology security policies and standards set out in Appendix III, 'Security of Automated Information Resources,' Office of Management and Budget Circular A-130; the Computer Security Act; and the Government Information Security Reform Act.

4.3. Objectivity

Information Product Category: Natural Resource Plan

Standards: This consultation and supporting documents are clear, concise, complete, and unbiased; and were developed using commonly accepted scientific research methods. They adhere to published standards including the NMFS's ESA Consultation Handbook, ESA regulations, 50 CFR 402.01 et seq., and the MSA implementing regulations regarding EFH, 50 CFR part 600.

Best Available Information: This consultation and supporting documents use the best available information, as referenced in the References section. The analyses in this opinion and EFH consultation contain more background on information sources and quality.

Referencing: All supporting materials, information, data and analyses are properly referenced, consistent with standard scientific referencing style.

Review Process: This consultation was drafted by NMFS staff with training in ESA and MSA implementation, and reviewed in accordance with West Coast Region ESA quality control and assurance processes.

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