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 Permit for the Orange County Sanitation District Reclamation Plant No. 1, Treatment Plant No. 2, Collection System, and Outfalls

NMFS Consultation Number: WCR-2021-00164

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?
Giant manta ray (*Mobula birostris*)	Threatened	No		NA	
Green sturgeon, Southern DPS (*Acipenser medirostris*)	Threatened	No		NA	
Oceanic whitetip shark (*Carcharhinus longimanus*)	Threatened	No		NA	
Scalloped hammerhead shark, Eastern Pacific DPS (*Sphyrna lewini*)	Endangered	No		NA	
Steelhead, Southern California DPS (*Oncorhynchus mykiss*)	Endangered	No		NA	
Green turtle, East Pacific DPS (*Chelonia mydas*)	Threatened	Yes	No	NA	
Leatherback turtle (*Dermochelys coriacea*)	Endangered	Yes	No	NA	
Loggerhead turtle, North Pacific Ocean DPS (*Caretta caretta*)	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?
Olive ridley turtle (*Lepidochelys olivacea*)	Threatened	Yes	No	NA	
Guadalupe fur seal (*Arctocephalus townsendii*)	Threatened	Yes	No	NA	
Blue whale (*Balaenoptera musculus*)	Endangered	Yes	No	NA	
Fin whale (*Balaenoptera physalus*)	Endangered	Yes	No	NA	
Gray whale, Western North Pacific DPS (*Eschrichtius robustus*)	Endangered	Yes	No	NA	
Humpback whale, Central American DPS (*Megaptera novaeangliae*)	Endangered	Yes	No	NA	
Humpback whale, Mexico DPS (*Megaptera novaeangliae*)	Endangered	Yes	No	NA	
North Pacific right whale (*Eubalaena japonica*)	Endangered	No		NA	
Sei whale (*Balaenoptera borealis*)	Endangered	No		NA	
Sperm whale (*Physeter macrocephalus*)	Endangered	No		NA	
Black abalone (*Haliotis cracherodii*)	Endangered	No		No	
White abalone (*Haliotis sorenseni*)	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: *June 16, 2021*

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LIST OF ACRONYMS

BE	Biological evaluation
BOD	Biological oxygen demand
BOD_5	Five-day biological oxygen demand
CA/OR/WA	California-Oregon-Washington
$CBOD_5$	Carbonaceous biological oxygen demand
CCE	California Current Ecosystem
CEC	Contaminants of emerging concern
CFR	Code of Federal Register
CHL-*a*	Chlorophyll-*a*
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
CPS	Coastal pelagic species
CTD	Conductivity-temperature-depth
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
ESA	Endangered Species Act
FIB	Fecal indicator bacteria
FMP	Fishery Management Plan
FR	Federal Register
GAP	Green Acres Project
GWRS	Groundwater Replenishment System
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
JWPCP	Joint Water Pollution Control Plant
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
NMFS	National Marine Fisheries Service

NOAA	National Oceanic and Atmospheric Administration
NOEC	No observed effect concentration
NPDES	National Pollutant Discharge Elimination System
OC San	Orange County Sanitation District
OC Water District	Orange County Water District
PAH	Polycyclic aromatic hydrocarbon
PBDE	Polybrominated diphenyl ether
PBR	Potential biological removal
PCB	Polychlorinated biphenyl
PDO	Pacific Decadal Oscillation
PFAS	Per- and polyfluoroalkyl substances
PFMC	Pacific Fishery Management Council
POP	Persistent organic pollutant
PPCP	Pharmaceutical and personal care products
PSP	Paralytic shellfish poisoning
RFMO	Regional Fishery Management Organization
RO	Reverse osmosis
RPM	Reasonable and prudent measure
SAR	Stock assessment report
SAIC	Science Applications International Corporation
SCB	Southern California Bight
SCCOOS	Southern California Coastal Ocean Observing System
SCCWRP	Southern California Coastal Water Research Project
SPLASH	Structure of Populations, Levels of Abundance and Status of Humpbacks
STAJ	Sea Turtle Association of Japan
SWRCB	State Water Resources Control Board
T&Cs	Terms and conditions
TBT	Tributyltin
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)
TIWRP	Terminal Island Water Reclamation Plant
TMDL	Total maximum daily load
TNBP	Tri-n-butyl phosphate
TPP or TPhP	Triphenyl phosphate
TPPO	Triphenylphosphine oxide
UME	Unusual mortality event
USFWS	U.S. Fish and Wildlife Service
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 USC 1531 et seq.), and implementing regulations at 50 CFR 402, as amended.

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 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 two 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 March 27, 2018, NMFS WCR received a letter from EPA regarding the need to consult on the EPA's reissuance of a National Pollutant Discharge Elimination System (NPDES) permit for the Orange County Sanitation District (OC San) Reclamation and Treatment Plants. EPA requested a list of federally listed, proposed, and candidate endangered or threatened species and designated and proposed critical habitat in the vicinity of the discharge, as well as any pertinent information to consider in assessing adverse effects from the discharge on these species and critical habitat.

On July 22, 2020, NMFS WCR received a request from EPA for early coordination and technical assistance regarding the EPA's reissuance of the NPDES permit to OC San. On July 30, 2020, the EPA shared a preliminary species list and NMFS WCR provided comments on additional species and critical habitat to consider.

On August 12, 2020, EPA sent a draft biological evaluation (BE) and essential fish habitat assessment (EFHA) to NMFS WCR for review and comment. On September 4, 2020, EPA met with NMFS WCR to provide an overview of the proposed action and timeline and to address initial questions regarding the draft BE and EFHA. On September 16, 2020, EPA and OC San met with NMFS WCR to provide a more in-depth presentation on the discharge and monitoring program. Following these meetings and presentations, EPA provided additional information and references from OC San regarding toxicity testing, monitoring data, and special studies, as well as a pre-public notice draft of the NPDES permit for review.

On October 9, 2020, NMFS WCR sent comments and questions on the draft BE and EFHA to EPA. In our comments, we recommended incorporating the information presented by OC San on September 16, 2020, into the BE and EFHA, to provide a comprehensive analysis of the potential exposure and effects of the discharge on ESA-listed resources and EFH. We also provided specific comments on the ESA-listed resources present in the action area and requested additional information about the outfalls, discharge plume, monitoring program, and proposed special studies under the new permit.

On January 12, 2021, EPA met with NMFS WCR to discuss the timeline and clarify a few questions on NMFS' comments regarding the draft BE and EFHA. On January 26, 2021, EPA provided the supporting documents, data, and references regarding discharge plume modeling and monitoring in response to our comments and questions.

On February 2, 2021, NMFS WCR received a letter from EPA requesting formal consultation under Section 7 of the ESA for the EPA's reissuance of a NPDES permit for OC San reclamation and treatment plants. EPA also requested to consult on EFH under Section 305(b)(2) of the MSA. As part of the consultation request, EPA provided the public notice draft NPDES permit and a revised BE and EFHA, including a summary of responses to NMFS WCR's comments on the draft BE and EFHA.

In the BE and EFHA (EPA 2021), EPA concluded that the proposed action is likely to adversely affect eight marine mammal species and four sea turtle species listed under the ESA. EPA also concluded and requested concurrence on their determination that the proposed action is not likely to adversely affect five fish species and two abalone species listed under the ESA. Regarding EFH, EPA concluded that the proposed action may adversely affect EFH for coastal pelagic species, highly migratory species, and Pacific coast groundfish.

We reviewed the public notice draft permit and the revised BE and EFHA submitted on February 2, 2021. 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 February 2, 2021. Subsequently, we exchanged information with EPA staff, including several conference calls (on March 18, April 22, and May 4, 2021) to discuss questions related to the consultation.

In the February 2, 2021, letter requesting consultation, EPA also requested that a draft biological opinion and EFH response be made available for their review and discussion prior to finalizing. On June 2, 2021, we transmitted a draft biological opinion and EFH response to EPA describing our analysis and conclusions regarding the effects of the proposed action on ESA-listed species, designated and proposed critical habitat, and EFH. Specifically, EPA requested receipt of a 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). On June 10, 2021, 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 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 (50 CFR 402.02). Under MSA, Federal action means any action authorized, funded, or undertaken, or proposed to be authorized, funded, or undertaken by a Federal Agency (50 CFR 600.910).

The proposed action is EPA's reissuance of a NPDES permit for the OC San reclamation and treatment plants. The purpose of the NPDES permit is to authorize the discharge of secondary treated wastewater and reverse osmosis (RO) concentrate (or brine) to the Pacific Ocean through Discharge Point 001 (the "120-inch outfall"). The NPDES permit would also authorize the discharge of secondary treated wastewater and RO concentrate to the Pacific Ocean through Discharge Point 002 (the "78-inch outfall") and Discharge Point 003 (Santa Ana River Overflow Weirs) in the event of an emergency or during planned essential maintenance or capital improvement projects on Discharge Point 001. The NPDES permit would be valid for a period of five years. In the following sections, we describe the OC San reclamation and treatment plant discharge operations, effluent limits, monitoring program, and special studies under the proposed permit.

We considered, under the ESA, whether or not the proposed action would cause any other activities that would have consequences on listed species or their critical habitat and determined that it would not.

1.3.1. OC San Operations

OC San provides treatment and disposal of domestic, commercial, and industrial wastewater for central and northern Orange County, California, and secondary treated effluent to the Orange County Water District (OC Water District) for the Green Acres Project (GAP) and the Groundwater Replenishment System (GWRS). Reclamation Plant No. 1 (Plant No. 1) is located in Fountain Valley adjacent to the Santa Ana River and Treatment Plant No. 2 (Plant No. 2) is located near the mouth of the Santa Ana River in Huntington Beach.

Both plants have been providing secondary treatment since 2012. Secondary treated effluent at Plant No. 1 is sent to the GWRS and/or GAP to be reclaimed using microfiltration, reverse osmosis (RO), and ultraviolet disinfection processes. Secondary treated effluent from Plant No. 2 is blended with other waste streams (e.g., RO concentrate and backwash) from the GWRS, the GAP, and both plants to form the final effluent. This final effluent is mainly discharged to the Pacific Ocean through the 120" outfall (Discharge Point 001) located on the San Pedro Shelf.

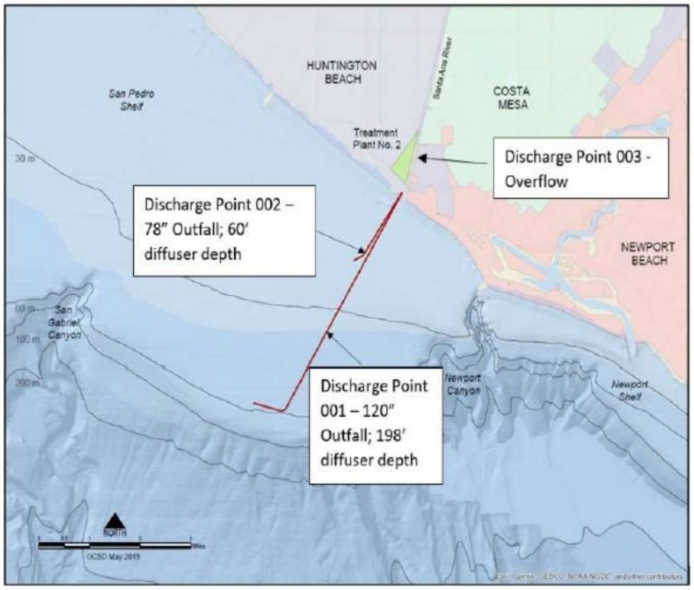

Figure 1. Location of Discharge Point 001, 002, and 003 (Figure 2 from EPA 2021).

Discharge Point 001 (the 120" ocean outfall) is the primary discharge point (Figure 1). This outfall is located on the San Pedro Shelf off Huntington Beach and has been in operation since 1971. The outfall is buried under sand until about 30 feet depth, where it emerges and extends out to a depth of 197 feet (60m). The outfall extends a total distance offshore of approximately 4.5 miles. The outfall is L-shaped, with the last mile running perpendicular to the rest of the outfall pipe. The last mile is a multi-port diffuser and follows the 197-ft depth contour off the mainland shelf. The diffuser section includes 500 diffuser ports that discharge horizontally on both sides of the diffuser. The 120" outfall pipe lies on soft-sand bottom, with the only rocky or hard substrate in the vicinity being the outfall pipe and the ballast rock and riprap used to stabilize the pipe. The 120" outfall has a capacity of 480 million gallons per day (MGD) at high tide. The 2019 California Ocean Plan (available at the State Water Resources Control Board (SWRCB) website) estimates a minimum probable initial dilution for this outfall as 181:1.

Discharge Point 002 (the 78" diameter ocean outfall) is only used during an emergency or during planned essential maintenance or capital improvement projects on the 120" outfall (Discharge Point 001) (Figure 1). This 78" outfall was placed in operation in 1954. It extends 1.5 miles offshore and to a depth of 65 feet (about 20m), traversing across soft-sand bottom. The diffuser section is 970 ft long with 122 round effluent ports and 8 rectangular ports and terminates in a flap gate. The 78" outfall has a discharge capacity of 230 MGD. The California Ocean Plan estimates a minimum probable initial dilution of 37:1. Discharge Point 002 has only been used once over the last ten years, during a six-week period in 2012 during a planned capital improvement project to inspect, assess, and rehabilitate the 120" outfall.

Discharge Point 003 consists of two extreme emergency discharge points (Santa Ana River Overflow Weirs) that discharge into the tidal prism in the Santa Ana River and ultimately into the Pacific Ocean (Figure 1). Discharge Point 003 is only to be used during extreme emergencies, such as tsunamis, earthquakes, floods, and acts of war or terrorism. One overflow point was installed in 1989 at the Ocean Outfall Booster Station facility at Plant No. 2 and consists of a 50 ft long overflow weir with two 72-inch pipes. The second overflow point was installed in 2007 at the termination structure upstream of the Effluent Pump Station Annex and consists of a 50 ft long overflow weir with two 66-inch pipes. The capacity of the first overflow weir is 475 MGD and the capacity of the second is 130 MGD, for a total capacity of 605 MGD (actual capacity dependent on the river's water level). Discharge Point 003 has not been used over the past 10 years.

The proposed permit prohibits the discharge of wastewater from Discharge Points 002 and 003, except in the event of an emergency or during planned essential maintenance or capital improvement projects on Discharge Point 001. An emergency is defined as "a circumstance that precludes discharging all wastewater to the 120" diameter ocean outfall despite proper operation and maintenance of the Sanitation District's facilities." Even in the event of an emergency, the proposed permit only allows discharge of disinfected secondary treated wastewater from Discharge Points 002 and 003.

Full secondary treatment of wastewater began in 2012. Prior to this, OC San added chlorine to the influent and primary effluent to address high concentrations of organic matter. Beginning in 2002, OC San used hypochlorite bleach on the final effluent followed by de-chlorination with sodium bisulfite to reduce fecal indicator bacteria. In 2012, OC San stopped using chlorine on the influent and primary effluent and reduced the use of hypochlorite bleach and sodium bisulfite by more than 90%. In 2015, OC San stopped the use of hypochlorite bleach and sodium bisulfite.

The average annual discharge to the Pacific Ocean has decreased over time, due to increased water reclamation diversions and reduced influent flows. Prior to 2008, the average annual discharge ranged from 175 to 262 MGD, with 3-10 MGD of treated wastewater diverted to OC Water District and to the GAP. In 2007-2008, OC San began diverting 35 MGD of secondary effluent to the GWRS for treatment (to standards for reclaimed water or for indirect potable reuse). The average volume of effluent diverted for water reclamation has increased over time, to 130 MGD in 2017-2018. At the same time, the average annual ocean discharge volume has decreased, to 117 MGD in 2018-2019, and is expected to continue to decrease over the life of the proposed permit. The main reason for this decrease is the OC San and OC Water District's joint GWRS Final Expansion project. Upon completion of the GWRS Final Expansion project in 2023, OC San expects to divert up to 175 MGD of effluent to the GWRS and decrease ocean discharge volumes by half. Ocean discharge would consist of a mix of secondary treated wastewater and GWRS RO reject. Table 1 summarizes the expected changes in discharge volume and composition after completion of the GWRS Final Expansion project in 2023.

Table 1. Summary of expected changes to discharge volumes and composition after completion of the GWRS Final Expansion project in 2023 (from EPA 2021).

Outfall	Year	Flow (MGD)	Estimated TDS[1] (mg/L)	Estimated Salinity (‰)
120" outfall	In 2019	100 (average daily)	3,000	1.7
120" outfall	In 2025	55-73 (average dry weather) 344-389 (peak wet weather)	<4,000	3
78" outfall	In 2012	140-145 (average daily)	3,000	1.7
78" outfall	After 2023	Varies (up to 230)	<4,000	3

[1]TDS = total dissolved solids; a measure of salinity.

1.3.2. Permitted Effluent Limits

The proposed permitted effluent limits and wastewater discharge requirements for the OC San 120" outfall and 78" outfall are described in the draft proposed NPDES permit submitted to us by the EPA on February 2, 2021. 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 proposed permit also includes monitoring of performance goals and mass emission benchmarks, which are not considered enforceable effluent limitations or standards, but are instead based on the last five years of actual performance data for the OC San plants and used as an indication of treatment efficiency at the plants. Permitted effluent limits, performance goals, and mass emission benchmarks may be measured over varying time scales, such as average weekly, monthly, or annual values, as well daily or instantaneous maximum values. The proposed permitted effluent limits, performance goals, and mass emission benchmarks are largely derived from or reflect the objectives that are laid out it in the California Ocean Plan, along with site-specific considerations and the performance of OC San's discharge under previous NPDES permits.

In the BE/EFHA (EPA 2021), EPA evaluated effluent quality based on the 16 pollutants that were consistently quantifiable or among the highest in terms of concentration or mass. These 16 pollutants include 11 metals (antimony, arsenic, cadmium, chromium, copper, cyanide, lead, nickel, selenium, silver, and zinc), total suspended solids, biological oxygen demand (BOD), chlorine, ammonia, and nutrients. The proposed permit includes limitations for these 16 pollutants, as well as additional pollutants that can be found in the effluent, including persistent organic pollutants (POPs) such as benzidine, hexachlorobenzene, toxaphene, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and tetrachlorodibenzo-p-dioxin (TCDD) equivalents. These additional pollutants are also monitored but are consistently not detected or the measured concentrations are consistently below the limits of quantification for their respective analytical methods.

Minimum dilution ratios are used to calculate the water-quality based effluent limitations (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 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 Discharge 001 (120” outfall), the minimum dilution ratio used to calculate effluent limitations is 180:1 (180 parts seawater to one part effluent). For Discharge 002 (78” outfall), the minimum dilution ratio used is 37:1.

As discussed above, OC San expects to increase water reclamation and decrease ocean discharge volumes over the next five years under the proposed permit. OC San used models to evaluate how this decrease in discharge volume may affect initial dilution. Model results estimate an increase in initial dilution as discharge volumes decrease, with little to no effect on the vertical or horizontal extent of the area where initial dilution is expected to occur. The proposed permit will require special studies to evaluate how increased water reclamation affects initial dilution and the concentrations of pollutants in the effluent (see Section 1.3.4 Special Studies).

1.3.3. Monitoring Program

The proposed permit prescribes a detailed monitoring program to demonstrate compliance with the effluent limitations, discharge specifications, and other requirements in the proposed permit (detailed in Attachment E of the proposed permit). The monitoring program covers a 288 square mile area off the coast of Orange County and encompasses monitoring stations for water quality, benthic, and trawl sampling (Figure 2). Under the proposed permit, OC San is required to collect and analyze samples of influent, effluent, receiving water, sediment, fish, and invertebrates, as well as participate in regional monitoring, to assess the effects of the discharge on the receiving waters. OC San’s monitoring program consists of three interconnected components:

(1) Core monitoring: water quality, benthic, and trawl stations;
(2) Bight Regional Monitoring; and
(3) Strategic process studies (part of the Special Studies; see Section 1.3.4)

1.3.3.1. Core Monitoring

The core monitoring program consists of the following elements: influent and effluent monitoring stations; receiving water monitoring stations (inshore and offshore); benthic infauna and sediment chemistry sampling; trawl monitoring stations; and local bioaccumulation sampling.

The proposed permit requires monitoring of influent and effluent at varying intervals depending on the constituent being monitored, ranging from daily, weekly, monthly, quarterly, semiannually, to annually. During planned discharges through Discharge Point 002, the proposed permit requires the same effluent monitoring as is required for Discharge Point 001. During emergency discharge through Discharge Point 002 or 003, the proposed permit requires minimum daily sampling until the emergency discharge ceases. This daily monitoring during emergency discharge must include at least bacteria indicators, BOD_5, carbonaceous biological oxygen demand (5-day; $CBOD_5$), oil and grease, total suspended solids, settleable solids, turbidity, pH, total residual chlorine, and ammonia.

Under the proposed permit, changes to the monitoring program include:

- an increase in the monitoring frequency for nutrients (including nitrate nitrogen) in the effluent and receiving water from quarterly to monthly;

- a new sediment monitoring station design to provide rapid assessment of conditions as well as long-term trends analysis;
- revisions to the target fish species for trawl monitoring and the analysis of fish tissues for bioaccumulation analysis; and
- the addition of a new effluent monitoring station for discharge through Discharge Point 002 during periods of essential maintenance or capital improvement projects.

Receiving water quality monitoring is conducted once per month at a grid of 28 stations centered on Discharge Point 001 and covering the coastline from Huntington Beach to Newport Beach (Figure 2), with additional quarterly sampling at the eight inshore stations (within 3 miles of shore) for nitrate nitrogen, ammonia, and fecal indicator bacteria (FIB; enterocci plus total and fecal coliform bacteria). Water column profiling is conducted at all stations and at depths ranging from 3 to 246 feet using a conductivity-temperature-depth (CTD) profiling system. This monitoring provides the information necessary to demonstrate compliance with the water quality standards. Compliance will be evaluated based on statistical comparisons between water quality profiles in the reference and plume-affected zones.

Benthic infauna and sediment chemistry sampling is conducted at least annually to regularly assess trends in sediment contamination and biological response along a fixed grid of sites within the influence of the discharge. As part of this sampling, whole sediment toxicity monitoring is conducted annually using a 10-day amphipod (*Eohaustorius estuaries*) survival test. Under the proposed permit, the benthic and sediment monitoring design will be revised from 29 semi-annual and 39 annual monitoring stations to 11 quarterly, 11 annual, and 35 one-off (once per five years) stations. These changes are intended to provide a more rapid assessment of sediment conditions and the benthic infaunal community at quarterly stations immediately surrounding the outfall diffuser and long-term trends analysis at the one-off stations.

Trawl sampling occurs at least annually at an array of sites to examine the health and temporal trends in community structure for demersal fishes (flatfish) and epibenthic macroinvertebrates in the vicinity of the outfall. Bioaccumulation sampling will also occur annually to determine if fish tissue contamination levels in the vicinity of the outfall are changing over time, focusing on flatfish and rockfish. The proposed permit includes several changes to fish sampling. The target fish species for trawl sampling will include any species of flatfish rather than specific species (English sole, *Parophrys vetulus*, and hornyhead turbot, *Pleuronichthys verticalis*), due to difficulties in collecting the required number of fish per species under the previous permit. Muscle tissue will continue to be analyzed for rockfish, but not for flatfish. For flatfish, analyses will focus on liver tissue chemistry and histopathology. Liver histopathology monitoring will also increase in frequency from once per permit term to annually, to allow for more rapid detection of effects from the GWRS final expansion. Finally, liver and muscle chemistry analyses will be conducted on composite samples to increase the ability to detect target compounds.

1.3.3.2.Bight Regional Monitoring

Under the proposed permit, OC San is required to participate in the five comprehensive regional monitoring activities conducted in the Southern California Bight (SCB): the SCB Regional

Monitoring Program, SCB Regional Water Quality Program, Central Regional Kelp Survey, Orange County Regional Shoreline REC-1 Cooperative Monitoring Program, and Ocean Acidification and Hypoxia Mooring. We describe two of 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 SCB Regional Water Quality Program (previously known as the Central Bight Water Quality Cooperative Program or Central Bight Regional Water Quality Monitoring Program) is a coordinated quarterly receiving water quality monitoring program conducted by OC San, the County Sanitation Districts of Los Angeles, the City of Los Angeles, the City of San Diego, and the City of Oxnard. Under the proposed permit, OC San will reduce the number of water quality monitoring stations from 66 to 60 and reconfigure the array to extend monitoring further downcoast to Dana Point.

1.3.3.3. Harmful Algal Bloom (HAB) Monitoring

Recent studies show that wastewater discharges and upwelling are the two most important contributors of nutrients in the SCB (Howard et al. 2014). The planned increase in water reclamation/recycling and increase in RO reject flow has the potential to increase nutrient concentrations in the effluent. To address this, the proposed permit will require weekly ammonia monitoring in both the influent and effluent, annual total nitrogen monitoring, and increase the monitoring frequency for nitrogen and phosphorus in the effluent from quarterly to monthly. The proposed permit will also require monthly nitrate nitrogen and ammonia monitoring for the receiving waters and additional quarterly nitrate nitrogen and ammonia monitoring at the eight inshore stations (within 3 miles of shore). These changes will provide data to characterize effluent quality, assess the effect of increased water reclamation and RO reject flow on nutrient outputs, and investigate the relationship between nutrient loadings and HABs. Under the proposed permit, OC San will also continue to participate in the Bight Regional Monitoring Program to analyze the relationship between nutrients discharged through wastewater outfalls, upwelling, and HABs.

1.3.3.4. Whole Effluent Toxicity Testing

The proposed permit requires monthly chronic toxicity testing and quarterly acute toxicity testing using whole effluent for both Discharge Point 001 and 002. Toxicity is determined relative to a control or reference sample using a given test method. Whole effluent toxicity tests employ 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, EPA considers standard test species to represent the sensitive range of all ecosystems analyzed. For acute toxicity testing, standard test species are topsmelt

(*Atherinops affinis*), mysid (*Americamysis bahia*), and inland silverside (*Menidia beryllina*). For chronic toxicity tests, standard test species are topsmelt, inland silverside, red abalone (*Haliotis rufescens*), purple sea urchin (*Strongylocentrotus purpuratus*), sand dollar (*Dendraster excentricus*), and giant kelp (*Macrocystis pyrifera*).

EPA has noted that the results of whole effluent tests in the past using five marine/estuarine short-term chronic test methods also 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 is 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

The proposed permit includes Strategic Process Studies and additional special studies to supplement OC San's monitoring program (see Attachment E of the proposed permit for more details). These studies will specifically evaluate the effects of the GWRS final expansion, which is scheduled to be completed by 2023 and expected to result in decreased discharge volumes and higher effluent concentrations due to increased RO reject flow.

The proposed permit requires OC San to conduct Strategic Process Studies to document the effectiveness of its source control and wastewater treatment operations in protecting the coastal ocean. These Strategic Process Studies are designed to address unanswered questions raised by the core monitoring program and/or to focus on issues of interest. The proposed permit includes five Strategic Process Studies to enhance our knowledge of emerging issues associated with the discharge before and after the GWRS final expansion:

(1) ROMS-BEC ocean outfall modeling to assess the spatial and temporal extent of the discharged effluent before and after implementation of the GWRS final expansion.
(2) Microplastics characterization to evaluate the quantity and type of microplastics throughout the treatment system and to develop methods and analyses to inform the transport and fate of microplastics in the treatment process and the receiving environment, to inform exposure levels for humans and wildlife.
(3) In-vitro cell bioassay monitoring to assess contaminants of emerging concern (CEC) in the receiving water. Bioanalytical screening (in-vitro cell bioassays) will be used to screen for a broad array of CECs that exert toxicity through common modes of action (e.g., estrogenicity or cancer causing). Seawater, sediment, and fish tissue samples will be screened for three types of receptor activity. An initial assessment will be conducted and will provide the foundation for follow-up investigations to evaluate the effects of the GWRS final expansion on the effluent and receiving water quality.
(4) Sediment linear alkylbenzene analysis to evaluate whether other contaminants in the sediment are associated with the effluent discharge and to evaluate future changes due to the GWRS final expansion. Linear alkylbenzenes are organic contaminants that can be used to track wastewater particles in the offshore environment.
(5) Marine biota (i.e., meiofauna) monitoring to evaluate potential effects of the GWRS final expansion on receiving water quality and marine aquatic life. Meiofauna are animals

ranging from 63 to 500 µm in size and known to be more sensitive to anthropogenic effects than macrofauna.

The proposed permit also includes three additional special studies to evaluate the effects of the GWRS final expansion on the receiving environment:

(1) CEC monitoring: the proposed permit will require OC San to continue to investigate CECs in the discharge and/or receiving waters. Previous CEC monitoring (in 2014-2018) included evaluation of the following contaminants in the final effluent: 15 pharmaceuticals and personal care products, seven hormones, seven industrial endocrine disrupting compounds, and nine polybrominated diphenyl ethers (PBDEs; flame retardants). The proposed permit adds the following CECs to the list for monitoring: the next generation of phosphate-based flame retardants (chlorinated organophosphate flame retardants such as TDCPP, TCEP, and TCPP), six emerging pesticides, and 12 per- and polyfluoroalkyl substances (PFAS).
(2) Updates to the discharge plume and dilution model to consider the effects of the GWRS final expansion (e.g., higher salinity and lower discharge flow rate) on discharge plume behavior in the zone of initial dilution.
(3) Outfall and diffuser system inspection: OC San will be required to externally inspect the 120” and 78” outfalls every 2.5 years to assess their condition and that they can continue to operate safely. The inspection will include a scoping study to analyze the potential effects of low flows on diffuser functionality (e.g., diffuser hydraulics and plugging). We note that inspection of the 120” outfall will not necessarily require discharge through the 78” outfall.

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 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 non-discretionary reasonable and prudent measures and terms and conditions to minimize such impacts.

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 does not include an adverse modification analysis because the proposed action is not likely to adversely affect any designated critical habitat within the action area.

The 2019 regulations define effects of the action using the term "consequences" (50 CFR 402.02). As explained in the preamble to the regulations (84 FR 44977), that definition 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 to 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 would 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" as described in 50 CFR 402.02.

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).

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 the blue whale 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. 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. 2020). 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. 2020). 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 between 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. 2020). 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. Populations 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 2020). It is difficult to estimate the numbers of blue whales possibly killed and injured by fishing gear, because large whales that may 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 (0.4/year) from vessel strikes is less than the calculated potential biological removal (PBR; 1.23) 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. 2020). 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 2013-2017 is 1.44 whales annually (Carretta et al. 2020). Considering the effects of estimated vessel strikes, plus the effects of entanglements, human impacts exceed the calculated PBR of 1.23 for this stock. Observed and assigned levels of serious injury and mortality due to commercial fisheries (1.44) exceed the stock's PBR (1.23); thus, commercial fishery take levels are not approaching zero mortality and serious injury rate. 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 2020) 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.

[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).

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 2017).

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. 2020). 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 CA/OR/WA stock.

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).

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 to 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. 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, the total quantified documented incidental mortality and serious injury (2.5/yr) due to fisheries (0.67/yr) and estimated vessel strikes (43/yr) is less than the calculated PBR of 81 (Carretta et al. 2020). Total fishery mortality is less than 10% of PBR and, therefore, may be approaching zero mortality and serious injury rate (Carretta et al. 2020). However, in recent years, there have been additional 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.

Based on the most recent information on humpback whales off the U.S. west coast (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 that was used by (Wade 2017) or in the most recent CA/OR/WA SAR (Carretta et al. 2020). In March 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 2021b). Table 2 summarizes the estimated abundance and proportion of each DPS off the U.S. west coast.

Based on NMFS WCR’s March 2021 memo (NMFS 2021b), 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 39% of the humpback whales present in the action area would be Central America DPS, and 61% 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.

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 = 1,877)**	**Probability that a humpback would be from Mexico DPS (N = 6,725)**	**Probability that a humpback would be from Hawaii (non-listed) DPS**
CA/OR*	39%	61%	0%
WA**	9%	28%	63%

*Probabilities are based on the assumption that both the listed DPSs have increased 6% per year since the SPLASH studies (NMFS 2021b). 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 verses a Mexico DPS (Calambokidis et al. 2017).

**Source: Wade (2017); Table 3b). Note that the majority of humpbacks that may be found off WA likely are moving north of the U.S. border and feeding primarily off southern British Columbia. In addition, we are currently applying the same proportions of the two listed DPSs to both coastal and inland waters of WA until further analysis is completed.

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 Revillagigedos 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 (Carretta et al. 2020). 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 March 2021 memo, we assume that the population has increased by 6% annually over the last 15 years. Using Wade (2017)'s population estimate of 2,806 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 6,725 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 (Carretta et al. 2020). 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 March 2021 memo, we assume that the population has increased by 6% annually over the last 15 years. Using Wade (2017)'s population estimate of 783 whales based on information from 2004-2006 and the assumed 6% annual increase, the current abundance estimate for the Central America DPS is 1,877 animals.

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, the estimated annual mortality and serious injury of the CA/OR/WA stock of humpback whales due to commercial fishery entanglements (17.3yr), non-fishery entanglements (0.2/yr), recreational crab pot fisheries (0.35/yr), tribal fisheries (0.2/yr), serious injuries assigned to unidentified whale entanglements (2.1/yr), plus estimated ship strikes (22/yr), equals 42.1 animals. This estimate is greater than the PBR allocation of 16.7 animals for U.S. waters (Carretta et al. 2020). 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 (17.3/yr) is greater than 10% of the PBR; therefore, total fishery mortality and serious injury is not approaching the zero mortality and serious injury rate (Carretta et al. 2020).

2.2.1.4. Gray Whale, Western North Pacific Population

Western North Pacific (WNP) 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. 2020). The most recent minimum estimate of endangered WNP gray whale abundance is 271 individuals (Carretta et al. 2020). 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. 2020). 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. 2020).

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. (2020), 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 Aurioles-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. 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 34,187 to 43,954 seals is based on pup count data collected in 2013 (García-Aguilar et al. 2018). 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. 2020). 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, 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., it is presumed 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. 2020).

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. 2020), 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. Hashimoto et al. (2017) used photo identification to record at least 62 different individuals present in the San Gabriel River between 2008 and 2015. Recent satellite tracking studies indicate that green turtles generally spend the majority of their time in coastal estuaries, but that at least some individuals use the San Pedro Shelf extensively (Hanna et al. 2020). 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 populations, including the Long Beach populations (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. 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.

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 Sulawasi 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 collect and/or 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. A current global population estimate is not available at this time, but we provide details on known populations below.

In the Pacific, leatherback populations are declining at all major Pacific basin nesting beaches, particularly over the last two decades (Spotila et al. 1996, 2000; NMFS and USFWS 2007b). In the eastern Pacific, nesting counts indicate that the population has continued to decline since the mid 1990's, leading some researchers to conclude that Pacific leatherbacks are on the verge of extirpation (Spotila et al. 1996; Spotila et al. 2000). Estimates of the number of nesting females/year in Mexico and for Costa Rica were reported to be approximately 200 animals or less for each country per year (NMFS and USFWS 2013). More recent estimates show a more positive increasing trend on the nesting beaches in Mexico, where an estimated 280 females may have nested along the Pacific coast of Mexico during 2010-2012 (Lopez et al. 2012). However, a more disturbing decline has been reported at Las Baulas, Costa Rica, with less than 30 females nesting in recent years (G. Schillinger, The Leatherback Trust, personal communication, 2016).

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; this region is also known as the Bird's Head Peninsula, where approximately 75 percent of regional nesting occurs (Hitipeuw et al. 2007). The Bird's Head region consists of four main beaches, three that make up the Jamursba-Medi beach complex, and a fourth, which is Wermon beach (Dutton et al. 2007). A decade ago, the nesting population was comprised of an estimated 2,700–4,500 breeding females (Dutton et al. 2007; Hitipeuw et al. 2007). Although there is generally insufficient long-term data to calculate population trends, in all of these areas, the number of nesting females is substantially lower than historical records (Nel 2012). 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 2020). 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 2020).

The most recently available information on the number of nesting females in the Bird's Head region reflects a significant decline. Tapilatu et al. (2013) estimated that the annual number of nests at Jamursba-Medi has declined 78.2% over the past 27 years (5.5% annual rate of decline), from 14,522 in 1984 to 1,532 in 2011. The beach at Wermon has been consistently monitored since 2002 and has declined 62.8% from 2,944 nests in 2002 to 1,292 nests in 2011 (11.6% annual rate of decline). Collectively, Tapilatu et al. (2013) estimated that since 1984, these primary western Pacific beaches have experienced a long-term decline in nesting of 5.9% per year, with an estimated 489 females nesting in 2011. Based on that information, the total number of adult females in the Bird's Head region was estimated to be 1,949 based on summer nests (April-September) (Tapilatu et al. 2013; Van Houtan 2014). This represents about 75% of the nesting activity in the Western Pacific; therefore, NMFS estimated that there were approximately 2,600 nesting females in this population (NMFS 2014).

Since 2012, monitoring effort at Jamursba-Medi and Wermon beaches has been somewhat variable and the overall nesting trend has continued to decline by 5% (NMFS 2019b). The trend in nesting activity oscillates, but appears to have increased slightly in recent years, though this increase is not affecting the long term trend and more years of data are needed to understand this increase means for the population (Jones et al. 2018; NMFS 2019b). Jones et al. (2018) estimated the current adult portion of the population is 1,851 (~1,390 females). Using the best available data for the West Pacific leatherback population (Fitry Pakiding, University of Papua, pers. comm. 2020), Martin et al. (2020) tested several models and found that the data best fit a Bayesian steady-state model, which 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% credible interval of 666 to 942 females as a snapshot of current abundance in 2017. Applying the conservative estimate of 75 percent to the Martin et al. (2020) estimate of 790 nesting females in Jamursba Medi and Wermon, the total number of nesting females in the West Pacific population would be 1,054 females with an overall 95% credible interval of 888 to 1,256 females.

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 Jamursba-Medi and Wermon portion of the Western Pacific leatherback population, along with the probabilities of this subpopulation reaching critical abundance thresholds within a 100 year projection period, and time in years (mean, median, and 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 0.949 (95% confidence interval 0.849 to 1.061), which suggest that most trajectories of this population can be expected to decrease in the coming years (NMFS 2019b). More recently, Martin et al. (2020) used a Bayesian state-space population growth model to estimate a declining trend for leatherbacks (−6.1% annually; 95% CI: −5.6% to −6.4%). Although human interactions are a major source of mortality for this declining population, there are indications that natural fluctuations in environmental and oceanic conditions could be significant influences on survival rates across various life stages or on reproductive rates (Van Houtan 2011; NMFS 2012b; Tomillo et al. 2012).

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 2020).

Leatherbacks are vulnerable to bycatch in a variety 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 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 6,984 (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, which 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 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 2007c). 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 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 2007c). 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 2007c).

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 off the Orange County coastline covering the San Pedro Shelf, San Gabriel Canyon, and Newport Canyon and includes the 288 square mile area of ocean encompassing the OC San outfalls and core water quality, benthic, and trawl stations (Figure 2). The 120" outfall traverses the San Pedro Shelf, terminating at about 197 ft depth on the Shelf between the Newport and San Gabriel submarine canyons. Bottom depths on the San Pedro Shelf gradually increase from the shoreline to approximately 262 ft and then rapidly increase down to the open basin. The San Pedro Shelf is primarily composed of soft bottom sediments (sand with silt and clay). The Newport Shelf is located southeast of the San Pedro Shelf and is much narrower, with a more abrupt transition from coastline to open basin.

The action area encompasses the area where effects are anticipated, based on detections of the OC San wastewater discharge plume and our understanding of the plume distribution and extent. The plume distribution and extent are influenced by oceanographic conditions and processes, which result in variable seawater movement within the action area. Alongshore flows (upcoast/downcoast) predominate, with minor across-shelf transport (OC San 1997, 1998, 2004, 2011; SAIC 2001, 2009, 2011, cited in EPA 2021), but the specific direction of flow varies with depth and can reverse over time periods of days to weeks (SAIC 2011). Oceanographic processes like upwelling, coastal eddies, and tidal flows and currents also influence the mixing and transport of the effluent with coastal receiving waters and sediments.

Plume studies conducted by OC San indicate that the discharge plume is primarily transported alongshore following shelf topography, both north and south of the outfall depending on currents, with some indications that it also approaches but does not reach the shoreline (Noble and Xu 2004; Noble et al. 2009; SAIC 2009, 2011; cited in EPA 2021). Uchiyama et al. (2014)'s modeling of the discharge also captured this movement; daily snapshots predicted a highly variable pattern of dispersal that can include the plume folding back onto itself at times. The plume is estimated to extend a minimum of 3.1 miles (5 km) and a maximum of 7.8 miles (12.5 km) alongshore, with a plume age of less than half to almost four days (OC San 2001; cited in EPA 2021). The plume is typically detected at a depth of 98 to 131 feet, but may rise into the upper 33 ft less than 2% of the time (Tetra Tech 2002, 2008; cited in EPA 2021). A particle settling and modeling study estimated that the discharge affects an approximately 89 square mile (230 km^2) area around the 120” outfall (EPA 2021). Uchiyama et al. (2014)’s modeling also predicted that the plume can move north along the countercurrent to the Palos Verdes headland, where it would combine with the discharge from the Los Angeles County Joint Water Pollution Control Plant (JWPCP) on its way into Santa Monica Bay. NMFS recognizes that there is still significant uncertainty regarding movement of the OC San's discharge plume, as well as transport of the algal biomass generated by the nutrients in the discharge, that are too great to anticipate effects in Santa Monica Bay at this time. We expect that monitoring and modeling efforts currently underway in the SCB will help to clarify these uncertainties and to more precisely define the action area in future consultations with EPA regarding NPDES permit reissuance to OC San.

Another important component of the action area is the zone of initial dilution (ZID). The ZID is the area surrounding the diffuser where the effluent undergoes initial mixing with the receiving water and where water quality criteria are allowed to be exceeded while this initial mixing occurs. When effluent is discharged from the 120” outfall, it first undergoes an initial dilution process where the effluent rapidly mixes with ambient seawater until a point of neutral buoyancy is reached, either trapping the effluent below the surface or reaching a boundary (e.g., the ocean surface or bottom). This initial dilution process is rapid and energetic, occurring over timescales of seconds to minutes, and is complete when the diluted effluent ceases to rise in the water column and first begins to spread horizontally. This plume of mixed effluent and ambient seawater moves away from the discharge point and becomes more diluted with increasing distance from the outfall.

The ZID is defined by critical conditions, that is, the conditions under which the initial dilution will be lowest and the physical mixing zone the largest. Critical conditions are identified by evaluating plume characteristics and initial dilution over a range of conditions for the effluent and ambient receiving waters to determine, for example, the highest effluent flow, minimum and maximum ambient currents, and density structure of the effluent and receiving water that result in the lowest initial dilution.

The ZID can be delineated as a spatial area or established by computing a dilution factor. In the proposed permit, the ZID (or mixing zone) is specified by a dilution factor of 180 (180 parts seawater to one part effluent). This dilution factor is used to calculate end-of-pipe effluent limits; that is, the pollutant concentration that is allowed in the effluent so that at the completion of initial dilution, the concentration will meet the water quality objective as specified in the permit. A dilution factor is used because the spatial extent of the ZID can fluctuate depending on currents and stratification (Tetra Tech 2008; cited in EPA 2021). For example, the length of the ZID was estimated to range from 31.5 to 1,220.5 ft (9.6 to 372 m).

The OC San's planned increase in water reclamation/recycling is expected to result in reduced discharge volumes and increased effluent salinity, temperature, and concentration. These changes have the potential to affect initial dilution as well as the plume distribution and extent. For example, increased salinity would make the effluent slightly less buoyant, but this may be offset by the increased temperature, which would make the effluent more buoyant. Model results from a recent dilution study (CDM Smith and Brown and Caldwell 2016; cited in EPA 2021) indicate that the reduced discharge volumes would increase initial dilution and have minimal to no effect on the vertical range or length of mixing. In addition, the plume remained deeply submerged in the model runs. The permit will require special studies to update the discharge plume and dilution model to incorporate the effects of these expected changes in discharge volume, concentration, salinity, and temperature.

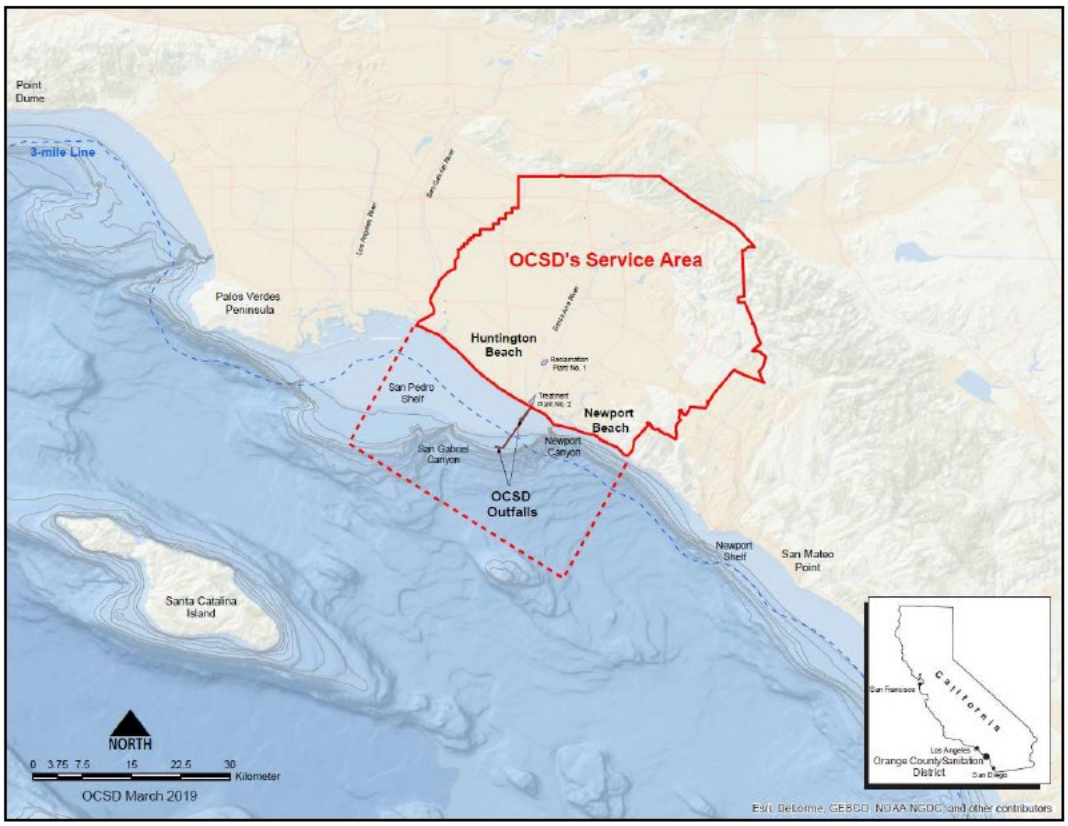

Figure 2. Action area off the Orange County coast, including the 288 square mile area of ocean encompassing the OC San outfalls and core water quality, benthic, and trawl stations (marked by the red dotted line).

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 402.02).

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., the San Pedro Shelf, extending through the neighboring San Gabriel and Newport Canyon area), even though the action area is a relatively small and confined area compared to the large ranges of most of these highly mobile species. Given the large human population and high level of human activity in and around the coastal waters off Los Angeles and Orange County, 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 in 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' 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, but did not identify any.

2.4.1. Habitat and Environment Health

The San Pedro Shelf is one of the broadest mainland continental shelf segments between Monterey, California, and the United States - Mexico border. The nearshore coastal waters of the action area receive wastes from a variety of human-related sources such as treated wastewater discharges, dredged material disposal, oil and gas activities, boat/vessel discharges, urban and agricultural runoff, and atmospheric fallout (EPA 2021). Discharges from the Los Angeles, San Gabriel, and Santa Ana Rivers are responsible for substantial surface water contaminant inputs to the SCB (Schafer and Gossett 1988; SCCWRP 1992; Schiff and Tiefenthaler 2001).

Ocean conditions within the action area are affected by both regional- and local-scale currents. Large regional climatic and current conditions, such as El Niño and the California Current, influence the water characteristics and the direction of water flow along the Orange County coastline (Hood 1993). Locally, the predominant low-frequency current flows in the monitoring area are alongshore (i.e., either upcoast or downcoast) with minor across-shelf (i.e., toward the beach) transport (OC San 2001; SAIC 2011). The specific direction of the flows varies with depth and is subject to reversals over time periods of days to weeks (SAIC 2011).

Seven facilities discharge into the action area and are sources of pollutants in the offshore environment. These facilities include:

- Four wastewater treatment plants (WWTPs): Orange County Sanitation District Sewage Treatment Plant, Terminal Island Water Reclamation Plant (TIWRP), Los Angeles County Joint Water Pollution Control Plant (JWPCP), and South Orange County Wastewater Authority.
- One electricity generating station: AES Huntington Beach, LLC.
- Two oil refineries: Nuevo Energy Company's Platform Esther and Platform Eva.

As described in the BE/EFHA, the OC San facilities receive wastewater flows from 3 million residents and 688 permitted businesses in an area encompassing approximately 480 square miles in central and northern Orange County. The main outfall (the 120" outfall or Discharge Point 001) has been in operation since 1971. Discharge of wastewater by OC San to other than the 120" outfall is prohibited except in the event of an emergency or during scheduled essential maintenance or capital improvement on the 120" outfall. OC San began operating at full secondary treatment in 2012. OC San has also operated an extensive source control pretreatment program to monitor and minimize industrial and non-industrial sources of pollutants in the influent itself. Both operation of the source control pretreatment program and transition to full secondary treatment have resulted in improved effluent quality. For example, the concentrations of most metals in the influent have declined since 2008. Regarding sediment quality, the 2018 Bight Regional Monitoring Program results indicate that multiple sources contribute to pollutants found in the sediment. For example, total DDT (dichlorodiphenyltrichloroethane) sediment concentrations were greatest near Palos Verdes and Los Angeles Harbor due to historical discharges from the JWPCP ocean outfall (EPA 2021). From the late 1950s to early 1970s, JWPCP is estimated to have discharged over 1,700 tons of DDT, largely due to wastewater discharged by the former Montrose Chemical Corporation DDT manufacturing plant (EPA Superfund website). Persistent pollutants like total PAHs, PBDEs, and pyrethroids were highest in embayments likely due to land-based runoff, whereas metals like copper were highest in embayments as well as in offshore areas (EPA 2021).

2.4.1.1. Water Quality in the Action Area

As described above in section 1.3.2 of the Proposed Action, OC San identified 16 pollutants of concern for the action area, based on those that were consistently quantifiable (above detection and reporting limits) or among the highest in terms of concentration or mass in the effluent. Additional pollutants are also monitored (including POPs such as PCBs, PAHs, and PBDEs) but are consistently not detected or are detected at levels below their limits of quantification.

In general, contaminants enter marine waters and sediments from numerous point sources (including WWTPs) and non-point sources such as from atmospheric transport and deposition, ocean current transport, and terrestrial runoff (Iwata et al. 1993; Hartwell 2004, 2008; EPA 2017, 2021). Contaminant levels are typically concentrated near populated areas of high human activity and industrialization. For example, San Pedro Bay near/offshore zone is listed as a section 303(d) impaired water body under the Clean Water Act, largely due to sediment toxicity in the area. The Dominguez Channel and Greater Los Angeles and Long Beach Harbor Waters Toxic Pollutants Total Maximum Daily Load (Harbor Toxics TMDL) covers all of San Pedro Bay and includes more than 70 pollutant-water body combinations. Pollutants specified include chlordane, PCBs, and DDT. A fish consumption advisory exists within the action area that extends from South of

Seal Beach Pier to San Mateo Point (https://oehha.ca.gov/advisories/south-seal-beach-pier-san-mateo-point).

Marine mammals and sea turtles 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. Addition 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. In the BE/EFHA, EPA describes how the pretreatment program implemented since 1977 has successfully reduced the average daily amount of metals entering OC San's system by 84% and metals discharged to the marine environment by 97% (EPA 2021). Due to both industrial and non-industrial source control efforts as part of OC San's extensive pretreatment program and improved secondary treatment at the plant, concentrations for most metals in OC San's influent have declined since 2008. For example, cadmium, chromium, silver, and lead have been reduced by over 99%; copper by 98%; nickel by 93%; and zinc by 96% (EPA 2021).

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 2017). In the BE/EFHA, EPA describes that ammonia concentrations have remained relatively stable since 2008 (Figure 5 *in* EPA 2021), and that average daily and monthly values of ammonia for 2008-2019 are well below the California Ocean Plan limits when a 180:1 dilution factor is applied according to OC San's permit (EPA 2021). In the BE/EFHA, EPA reports that the receiving water monitoring data shows ammonia concentrations to be well below the concentrations required by the ammonia water quality objectives.

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 (Bonefeld-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. Data from OC San's fish bioaccumulation monitoring in the action area indicate that PCB and DDT levels in fish tissues (muscle and liver) are similar within outfall and non-outfall areas (EPA 2021). Mean concentrations of PCB and DDT in muscle tissue were consistently below state and federal advisory limits for human consumption. Mean concentrations were also consistently higher in liver tissues than in muscle tissues. Tagging and tracking of target fish species showed a low level of site fidelity, indicating that the fish may be exposed to contaminants at multiple locations and that the location of capture is not a definitive indicator of exposure (Burns et al. 2019; cited in EPA 2021). Consequently, the Palos Verdes Shelf, located west of the action area, is a major source of DDT and PCBs to the action area and to the surrounding area as well (Los Angeles County Sanitation Districts (LACSD) 2020). This contamination of the shelf resulted from wastewater discharge from the Montrose Chemical Corporation manufacturing plant. The long-term significant discharges of DDTs from this manufacturing plant (Eganhouse et al. 2000; Bay et al. 2003) and the historically heavy agricultural use in the area lead to these higher levels of DDTs in California creating what has been called the "California signature." In general, levels of DDTs are higher than PCB concentrations in sediments and marine biota from central and southern California (Jarvis et al. 2007; Kimbrough et al. 2008; Blasius and Goodmanlowe 2008).

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 Voutsa 2020). Additive flame-retardants can readily dissociate from the products they are added to and discharge into the environment. From 2014 to 2018, OC San did not detect PBDEs in their final effluent (EPA 2021). Preliminary Bight 2018 data also 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 Voutsa 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. Data on TCEP, TCPP, and TDCPP in OC San's effluent are not available at this time; these pollutants were not previously included in OC San's monitoring program (we note the proposed permit includes a special study to monitor TCEP, TCPP, TDCPP, and other CECs in the final effluent). However, information is available for other areas, including just 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 2017). 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 (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 and 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 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 are found at relatively low levels in the basin sediments compared to other coastal sediments. The proposed permit includes performance goals and mass emission benchmarks, as well as quarterly 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.,17α-ethynylestradiol, which also is a PCPP, 17ß-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;
- Nanomaterials such as carbon nanotubes or nano-scale particulate titanium dioxide, of which little is known about either their environmental fate or effects.

Since 2014, OC San has annually monitored the final effluent for 15 pharmaceuticals and PCPPs, seven hormones, seven industrial EDCs, and nine PBDE flame retardants (EPA 2021). The monitoring results show that the secondary treatment process removed few of the CECs. Frequently detected CECs included several PPCPs (i.e., acetaminophen, DEET, carbamazepine, gemfibrozil, ibuprofen, and trimethroprim) and several industrial EDCs (i.e., 4-nonylphenol, nonylphenol monoethoxylate, and 4-tert-octylphenol). Mean concentrations varied widely among individual CECs, ranging from not detected to greater than one microgram per liter among the hormones, from not detected to greater than 6 micrograms per liter among the PPCPs, and from not detected to greater than 18 micrograms per liter among the industrial EDCs. No PBDE flame retardants were detected (i.e., below the minimum detection levels), consistent with the 2018 Bight Regional Monitoring results that showed 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 it 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).

2.4.1.2. Harmful Algal Blooms

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 Hyperion and OC San (Howard et al. 2014). OC San alone is permitted to discharge up to 182 MGD of secondary treated effluent from Treatment Plant No. 1 and up to 150 MGD of secondary treatment from Treatment Plant No. 2 during dry weather flows. In recent years, the average daily discharge has been approximately 100 MGD and is expected to decrease due to additional water reclamation and recycling over the upcoming five-year NPDES permit cycle.

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 which graze upon them (Trainer et al. 2010). There are many known species in the California Current that may develop into HAB levels, but the most prevalent in the vicinity of the action area seem to be two diatom groups, the *Pseudo-nitzschia delicatissima* group and the *P. seriata* group, and dinoflagellates such as *Prorocentrum* spp., *Ceratium* spp., and *Lingulodinium polyedrum*, based on monitoring data generated by the Southern California Coastal Ocean Observing System (SCCOOS) at Newport Pier for years 2016-2021 (https://sccoos.org/harmful-algal-bloom/). Additional species of dinoflagellates, including *Alexandrium* spp., *Cochlodinium* spp., *Dinophysis* spp., *Gymnodinium* spp., and *Akashiwo saguinea,* have been detected at Newport 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 is responsible for well-documented toxic events to marine mammals and birds in the SCB and 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 saguinea* 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 in the last 10-15 years (Howard et al. 2012; Nezlin et al. 2012; Booth 2015). Anderson et al. (2012) notes that there are multiple reasons for this increasing bloom trend, including: 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. 2012, 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 and resulting literature 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. 2012, 2017; Booth 2015; Pondella et al. 2016). The largest four WWTPs in the SCB (including the OC San facility) accounted for 90% of the total WWTP discharges at the time of these studies (Howard et al. 2014).

Howard et al. (2014) determined that in the San Pedro Shelf region of the SCB, effluent discharges annually provided approximately half of the total nitrogen load as compared to upwelling contributions but are also the primary source of nitrogen inputs for a significant portion of the year outside of the upwelling and rainy seasons. As noted above (see Section 2.4.1 Habitat and Environment Health), there are four WWTPs that discharge into the San Pedro Shelf region, including two large WWTPs (OC San and the JWPCP) and two small WWTPs (TIWRP and the South Orange County Wastewater Authority – Aliso Creek Ocean Outfall). Howard et al. (2014) included all four WWTPs to calculate the total nitrogen contributions from effluent discharges to the San Pedro Shelf region. Thus, OC San's discharge accounts for only a portion of this total nitrogen contribution (see Section 2.5.3.1 Effect of OC San's Discharge on HAB Occurrence for details on the proportion assigned to OC San).

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. 2012, 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 to the thermocline or to the surface 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 OC San) 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 San Pedro Shelf and at the WWTP outfalls. This is consistent with results from CHL-*a* fluorescence surveys conducted from 2004 to 2018 within the action area, which indicate that blooms occurred during all months and were most prevalent near the outfall and ZID and inshore of the outfall (Seubert et al. 2013; cited in EPA 2021).

The physical oceanography in the vicinity of the OC San discharge influences the fate and transport of 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 times of the year (Howard et al. 2012), nutrients from the OC San facility may remain on the San Pedro Shelf for an appreciable amount of time. The San Pedro Shelf has been identified as a hot spot area 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 Pedro Shelf region from the south due to the Southern CA Counter Current (Howard et al. 2012) and from winter runoff.

HAB species are known to persist in the subsurface zone and then be advected into the shallow surface waters 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 OC San, 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 Pedro Shelf region when stratification is weak or shallow. These subsurface populations of HAB species can then be uplifted into the surface waters and are a probable 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 OC San's discharge plume shows the distribution to be primarily alongshore to both the north and south of the outfall, as well as approaching but not reaching the shoreline (EPA 2021). Ammonia concentrations from the discharge were primarily detected upcoast and offshore from the outfall (EPA 2021), but becomes more difficult to detect as ammonium 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. 2012; 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 discharge 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 of many 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. 2012). During wet years, urea entering the San Pedro Shelf region from the land (e.g., the Santa Ana River) 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 OC San outfall are advected into the surface waters 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 is likely due to subduction by eddies and there may be a significant topographic influence in the SCB (Kessouri et al. 2020) 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), including on the San Pedro Shelf area adjacent to the OC San outfall (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 to 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.

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 status. 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 Voutsa 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 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 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). 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). Less is known about olive ridley sea turtles.

2.4.2.2. Strandings

Strandings of ESA-listed marine mammals and turtles have been documented within 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: 3 fin whales, 3 humpback whales, 10 gray whales (it is unknown if any of these gray whales belong to the ESA-listed WNP gray whale population), one blue whale, and 12 Guadalupe fur seals. The cause of many of these strandings is unknown. Four whales (one fin, one humpback, one gray, and one blue whale) appeared to strand due to vessel strikes and 11 of 12 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: 36 green turtles, 3 loggerheads, 2 leatherbacks, 2 olive ridleys, and one unidentified sea turtle. The cause of many of these strandings is unknown. Fourteen of the green turtles interacted with recreational and/or commercial fishing gear, eight were struck by vessels, three were entrained in local utility systems, and one was harmed by marine debris. One loggerhead and two olive ridleys appear to have been struck by a vessel, and one leatherback and one loggerhead were entangled/harmed by marine debris.

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. 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 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 OC San:

- 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 OC San'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 this 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.

2.5.1. Exposure and Response to the Toxicity of OC San's Effluent

2.5.1.1. Species Occurrence and Exposure

To evaluate the presence of ESA-listed species within 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.

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), observed frequently transiting or foraging in areas that can occur very close to shore, including 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 Los Angeles and Orange County coastal areas 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 engage in foraging activity in association with prey sources that are known to occur in the area, including forage fish such as sardines and anchovies, and krill. Foraging is expected to occur 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 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 Los Angeles and Orange County 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 blue, humpback, fin, and gray whales have occurred in or very near the action area in recent years. As a result, we conclude all of these 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-lived lifetimes of extensive migrations that include the SCB. 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-lived lifetimes of extensive migrations or residence in the SCB. 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 well. 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. While we do not have any information that suggests any individuals from these 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 their relatively long-lived 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 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 utilizing Southern California waters more regularly or for extended foraging activities.

Green turtle strandings in the area are much more frequent than the other sea turtle species. Existence of a resident foraging population of juveniles and adults in nearshore and estuarine areas associated with the action area (Long Beach and Seal Beach) has recently been established through multiple avenues of study (Lawson et al. 2011; Crear et al. 2016; Hanna et al. 2020). Fidelity to foraging sites by green turtles has been well described, including foraging sites in Southern California within or near the action area (e.g., Crear et al. 2016; Hanna et al. 2020). At a minimum, we do 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-lived lifetimes of extensive migrations or residence in the SCB, and that the duration of exposure could last up to many days, weeks, or even months at a time or more.

2.5.1.1.3. Overlap with effects of the discharge

EPA predicts the effluent discharge plume from OC San to extend alongshore the Los Angeles and Orange County coastline on the San Pedro Shelf and neighboring canyons following shelf topography (EPA 2021). The plume is typically detected at depths of 100 feet (30 m) or more, but occasionally rises into the upper 33 ft (10 m) (EPA 2021). 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 primary discharge at the 120" outfall, the ZID is estimated to extend 31.5 to 1,220.5 ft (10 to 372 m). No ZID estimates have been provided for the occasional discharge that may occur from the 78" outfall, but we assume that this volume would also be relatively very small compared to the action area. 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.

2.5.1.2. Constituents of OC San's Discharge

As described in the BE/EFHA and in Section 1.3.2 (Permitted Effluent Limits), EPA evaluated 16 pollutants that are known to be present in quantifiable amounts in OC San's effluent. These pollutants include metals such as cadmium, copper, nickel, lead, silver, zinc. Elements and compounds such as nitrogen (ammonia), chlorine, and other nutrients are also discharged. Other constituents of the discharge that are monitored include POPs such as benzidine, hexachlorobenzene, toxaphene, PAHs, PCBs, and TCDD equivalents. Other constituents of the discharge that are suspected or known to be present in wastewater discharge include: POPs such as PBDEs, organophosphate esters, and 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 OC San'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 OC San 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; City of

Los Angeles Environmental Monitoring Division 2015; EPA 2017). 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. Available data for the action area shows phytoplankton blooms occurring in all months over the past 15 years; blooms were most prevalent near the outfall and ZID and inshore (EPA 2021). Infaunal and epibenthic macroinvertebrate communities did not show any outfall-related trends in community measures (e.g., abundance, species richness and diversity, Swartz's 75% Dominance Index) when comparing within-ZID to non-ZID stations (EPA 2021).

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 OC San'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 OC San'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 OC San'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.

2.5.2. 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; Fonnum 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; Bonefeld-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 San Francisco Bay 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 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 in this area (SCCWRP 2020 cited in EPA 2021; LACSD 2020). Historical discharges of PCBs also persist in sediments due to their long degradation time (LACSD 2020). The proposed permit includes average monthly effluent limits for PCBs, performance measures and mass emission benchmarks for PCBs and DDTs, and influent, effluent, and sediment monitoring requirements for PCBs and DDTs (influent and effluent sampling twice per year and sediment sampling at quarterly, annual, and once-per-five year sampling stations). Effluent monitoring shows that both PCBs and DDTs are detected, but consistently at concentrations below the limits of quantification (EPA 2021). Sediment concentrations of total DDT showed no outfall-related trends and were consistently below regional value (based on Bight 2008 and Bight 2013 sediment chemistry surveys) (EPA 2021). Sediment concentrations of total PCB were generally higher at the outfall, but were comparable to regional values, ranging from less than 5 to about 25 µg/kg between 2008 - 2019 (EPA 2021). Mean concentrations of DDTs and PCBs in fish tissues were similar among outfall and non-outfall reference sites from 2008-2018 (EPA 2021). The concentration of PCBs in muscle tissues spiked in 2013-2014 for English sole and in 2018-2019 for sportfish (rockfish) caught near the outfall (EPA 2021); however, the location of capture may not accurately represent the location of exposure (Burns et al. 2019; cited in EPA 2021). The potential for exposure to the legacy PCBs and DDT continue 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.
OC San's discharge is not currently identified as a significant source of PBDEs (flame retardants) as no PBDEs were detected in the effluent from 2014 to 2018 (EPA 2021). 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. Of particular interest 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 exposure of ESA-listed species to these pollutants due to the proposed

action, it is important to monitor the concentrations of these flame retardants in the effluent and understand how these pollutants from wastewater effluent move through the food web.

As an exercise to understand and isolate the effects of organophosphate flame retardant loading from OC San's discharge, we estimate the amount of organophosphate flame retardant that is expected to be discharged for the 5 years covered by the proposed NPDES permit. Organophosphate flame retardant loads from OC San are currently unknown. As described in the proposed action, to help fill this data gap, EPA and the CA State Water Resources Control Board agreed to add a special study to examine the discharge of phosphate-based flame retardants (chlorinated organophosphate flame retardants such as TDCPP, TCEP, and TCPP) in the effluent to evaluate the loading of the action area. In the meantime, to estimate the level of organophosphate flame retardant in OC San's wastewater effluent, 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 in a 2019 special study (LASAN 2020). Average mass loading of the organophospate flame retardants TCEP, TCPP, and TDCPP combined (mostly TCPP) in Hyperion's effluent was approximately five pounds per day (Figure 3) (LASAN 2020). This mass loading occurred during a period where the average discharge at Hyperion was approximately 230 MGD (LASAN 2020).

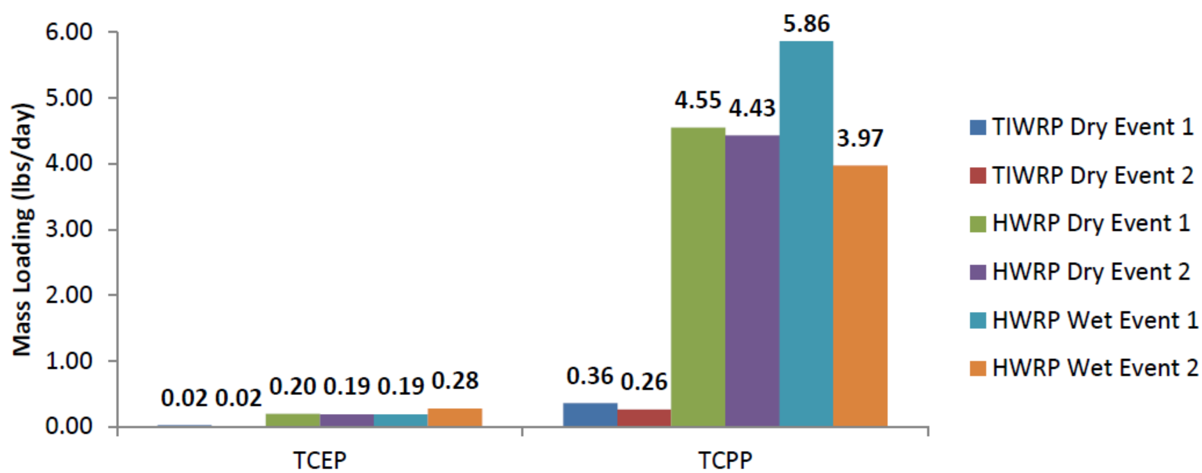

Figure 3. Mass loading calculations of organophosphate flame retardants found in Hyperion and Terminal Island Water Reclamation Plant effluents (from LASAN 2020).

Although OC San's 120" outfall has a capacity of 480 million MGD, the average annual ocean discharge volume has decreased over time to 117 MGD in recent years, and is expected to continue to decrease over the life of the proposed permit. Because OC San 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 OC San are likely to be relatively equivalent to what has been measured at Hyperion, without any results from OC San to examine. 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 OC San's effluent, using the average flow of 117 MGD and the peak hydraulic capacity of 480 MGD. As a result, we estimate a range of approximately 2.5 to 10.4 pounds (1.1 – 4.7 kg) per day of organophosphate flame retardants (TCEP, TCPP, and TDCPP combined) may be loaded into the action area as a result of OC San's wastewater discharge. This equates to an estimated 912.5 to 3,796 pounds (413.9 – 1721.8 kg) of organophosphate flame retardants discharged per year. For the total five years of this proposed action, this equates to approximately 4,562.5 to 18,980 pounds (2,069.5 – 8609.2 kg) for the permit cycle. It is important to note that OC San's actual wastewater flows have been substantially less than maximum flows, due primarily to increased water reclamation diversions and reduced influent flows, and that further decreases resulting from the GWRS Final Expansion project are expected (EPA 2021).

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 permits.

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, where 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 and is 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, calapnoid 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 which are then consumed by pelagic organisms, exposing the pelagic food web. This is likely an exposure route for POPs in OC San's effluent to ESA-listed marine mammals and sea turtles, in addition to deposition in sediments and the benthic food web.

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 et al. 1962). Currently, there are not 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 effects to different species from POP exposure.

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 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 hunting 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 and 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 to the cholinergic or neurotransmitter system (Talsness 2008). Therefore, marine mammal calves and pups are likely more susceptible to adverse health effects than adult whales and pinnipeds only exposed as adults 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 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 in the fish population (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). 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 concern for the harbor seal population.

Less is known about early exposure POPs to 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).

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 health risks are probably elevated as a result of interactions between toxic chemicals, and wildlife is rarely exposed to single compounds, the majority of studies have examined the effects of isolated chemicals. It has only been in more 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 circulating thyroid hormone concentrations (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, it is reasonable to 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

The OC San effluent contains potentially harmful contaminants that have been well established to adversely affect laboratory and wildlife species. PCBs and DDT have been measured in the action area, including in sediments near the outfall, and in the effluent, though at concentrations below the limits of quantification. The threat of TBT may be uncertain, but it will be subject to monitoring in the effluent. The potential presence of PBDE is of concern as well, although recent sampling has indicated this constituent may be decreasing. Recent information indicates that increasing use of organophosphate flame retardants is leading to an increased presence and threat from potential exposure to these constituents. Based on surrogate data from a similar adjacent WWTP, we estimate approximately 10.4 pounds (4.7 kg) per year of organophosphate flame retardants may be loaded into the action area as a result of OC San's wastewater discharge. Over the course of the five years of this proposed action, this equates to approximately 52 pounds (23.5 kg) for the total permit cycle. Once in the aquatic system, these constituents become bioavailable to food webs for uptake by ESA-listed species.

We recognize the prospect that concentrations of potentially harmful contaminants in the effluent may change over the next 5-year permit period given the planned increase in water recycling associated with the GWRS Final Expansion project. EPA expects the reduced discharge volume and increased (potentially) concentration (due to increased brine discharge) to have minimal effects on effluent quality, initial dilution, and plume distribution/extent (EPA 2021). However, EPA's proposed permit includes several special studies to evaluate the effects of these changes on the discharge, including levels of potentially harmful contaminants identified in this analysis.

ESA-listed marine mammals and sea turtles are affected by the proposed action indirectly by consuming prey that has accumulated POPs from OC San's effluent, which expedites the potential or timing for adverse health effects in ESA-listed marine mammals and sea turtles feeding in the action area to occur. 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 risk 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 to an individual's reproductive, endocrine, and immune systems. 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.

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 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 may also be present in OC San's discharge. However, there is limited knowledge and available information describing the levels of most of these contaminants in wastewater discharge and the extent of potential harmful effects. As described in Section 1.3.4 (Special Studies), EPA is requiring OC San to conduct a special study that includes developing monitoring programs for some of the CECs that may be harmful to ESA-listed species, including organophosphate flame retardants.

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), promoting HABs that potentially pose a threat to ESA-listed species.

2.5.3.1. Effect of OC San's Discharge on HAB Occurrence

The OC San discharge may have the effect of fertilizing or kick-starting the spring time 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; Seegers et al. 2015) and by providing nitrogen to the upper water column 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 through the euphotic zone to the thermocline or to the surface when stratification is weak (Reifel et al. 2013; Seegers et al. 2015). These conditions can commonly 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 along the San Pedro shelf near the OC San outfall and elsewhere in the SCB with limited exchange of water.

The physical oceanography in the vicinity of the OC San discharge influences the fate and transport of the nutrients and any subsequent phytoplankton or zooplankton that utilize the nutrients to grow. Circulation patterns over the San Pedro Shelf are influenced by the equatorward California Current that transports cold Subarctic water south and the poleward Southern California Countercurrent that transports warm southern water north through the Shelf into Santa Monica Bay and the Santa Barbara Channel (Howard et al. 2012). Local current flows are generally alongshore (upcoast or downcoast) with minor across-shelf transport (OC San 1997, 1998, 2004, 2011; SAIC 2001, 2009, 2011, cited in EPA 2021). Oceanographic processes such as upwelling and coastal eddies influence the mixing and transport of OC San's wastewater discharge on the San Pedro Shelf (OC San 2021). Nezlin et al. (2012) identified the San Pedro Shelf as a hot spot area with longer residence time of its water and higher CHL-α levels. Additional nutrients may enter the Bay from the south due to the Southern CA Counter Current (Howard et al. 2012) and from river flows and urban stormwater runoff (Hood 1993; Grant et al. 2001; Warrick et al. 2007; cited in OC San 2021).

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. 2012, 2014). 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. 2012; 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. Theses 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 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 of many years in San Pedro Bay (Nezlin et al. 2012).

Howard et al. (2014) estimated that nutrient loading of the San Pedro subregion through waste water effluent increased total nitrogen in the San Pedro subregion by about 26,455 pounds (12,000 kg) of nitrogen per km^2 per year. Given the area of the San Pedro Shelf subregion (1641 km^2), this amounts to about 43.4 million pounds (19.7 million kg) of nitrogen over the course of a year. As described in Section 2.4.1.2 (Harmful Algal Blooms), this amount of nitrogen is roughly equivalent to half the nitrogen brought into the San Pedro Shelf region from coastal upwelling. OC San and JWPCP are both large WWTP operating within the San Pedro subregion, along with some other small facilities. As a relative comparison, recent estimates of average effluent discharge at JWPCP have been approximately 260 MGD (CWB 2017, 2020), which is approximately twice as large as recent effluent discharges from OC San (~117 MGD). Knowing that these two facilities constitute most of all of the nitrogen discharge by volume in the area, we conservatively estimate that OC San may discharge up to one-third (33%) of the total nitrogen discharged into the San Pedro subregion, with up to two-thirds (67%) of the total nitrogen discharged associated with JWPCP. As a result, we estimate that OC San may increase total nitrogen in the San Pedro Shelf region by about 8,818 pounds (4,000 kg) of nitrogen per km^2 per year. This amounts to 14.3 million pounds (6.5 million kg) of nitrogen over the course of a year from OC San's discharge.

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

The potential for exposure of ESA-listed species to biotoxins such as those 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 risk of exposure of ESA-listed species and/or any prey species that may rely on phytoplankton (e.g., northern anchovies, pacific sardines, Pacific mackerel) to any HABs in the area is inherently present. As described above, it is likely OC San'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 could occur in and around the action area during the five years of the proposed permit. Although it is uncertain what degree OC San's effluent would play in any HAB, NMFS anticipates the OC San effluent would provide conditions that help encourage a HAB during this five year proposed action and in future actions. Therefore, the foraging ESA-listed marine mammals and sea turtles nearby (both inshore and offshore species) could be at increased risk of exposure to biotoxins.

Four classes of marine algal toxins have been associated with marine mammal mortality and morbidity events. These include saxitoxin, brevetoxin, ciguatoxin, and domoic acid (Van Dolah et al. 2003). Between 1978 and 2006, there were 57 of these mortality events detected nationally by the NOAA Fisheries Stranding Network. 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 4). Of the 71 UMEs, 18% have been caused by biotoxins from HABs (Figure 4), 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. The San Pedro Channel has been identified as a hot spot for domoic acid, with HABs dominated by *Pseudo-nitzschia* spp. (Schnetzer et al. 2007, 2013). The dinoflagellates *Procentrum* and *Akashiwo sanguinea*, which can produce saxitoxin, are also known to be present near OC San (https://sccoos.org/harmful-algal-bloom/). Phytoplankton assemblages 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 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).

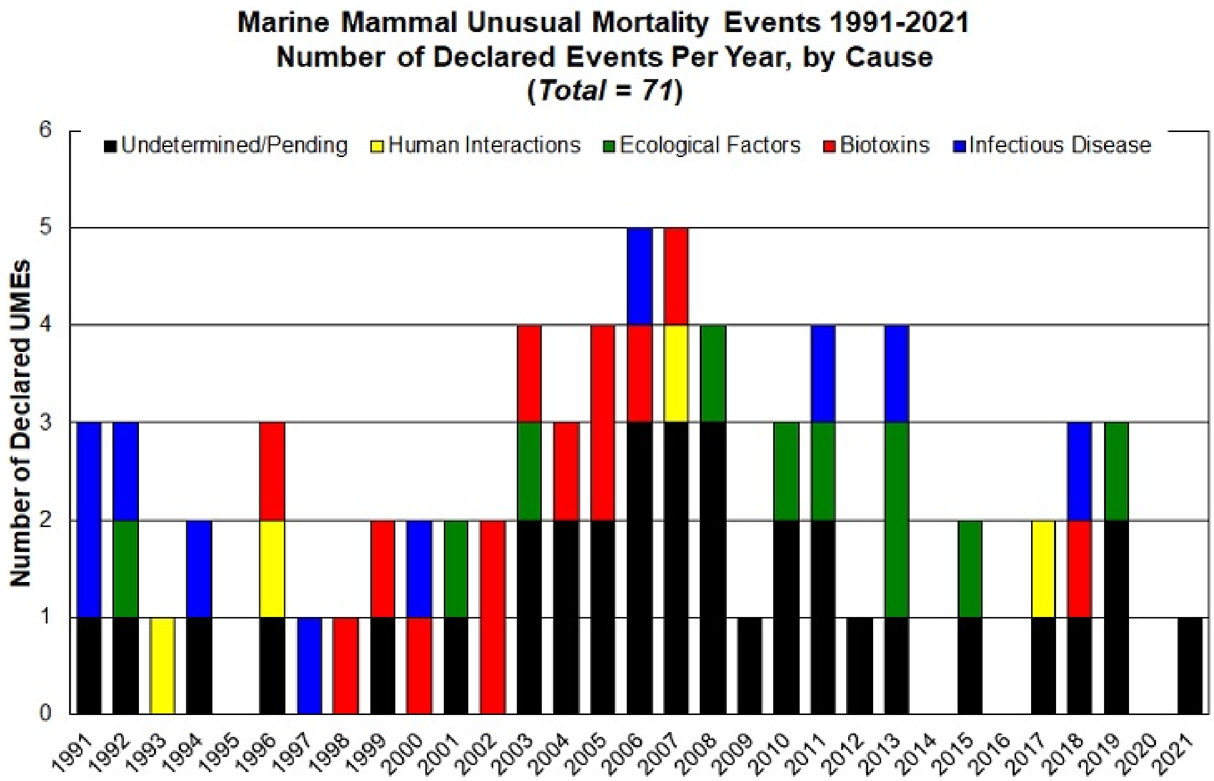

Figure 4. Number of unusual mortality events between 1991 and 2021 per year and by cause (figure reprinted from http://www.nmfs.noaa.gov/pr/health/mmume/events.html).

Clearance of these biotoxins from the blood is rapid. 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, which greatly 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 it 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 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 generally 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 capenis*). 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 be capable of effecting 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 the potential for domoic acid exposure cannot be ruled out. In their study, they tested for domoic acid in plasma and feces, but suggested urine and stomach contents are likely better samples for evaluating exposure (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 seizuring California sea lions. These data reveal sea turtles can be exposed to biotoxins, but it is currently unclear if the effects are similar to that found in birds and marine mammals.

2.5.3.3. 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 OC San can potentially increase the frequency and/or extent of HABs. At this time, we cannot predict the precise extent that OC San'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 created by OC San's discharge. What is clear is that HABs pose a significant health risk for ESA-listed marine mammals and sea turtles, that increasing the probability of HAB occurrence further increases the likelihood of adverse effects from HABs, including impaired health (injury) and mortality, and that OC San's discharge increases the possibility of this occurrence. 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 risks of adverse effects that HABs are known to present.

2.5.4. Risks to Marine Mammal and Sea Turtle Populations

In summary, OC San's discharge of effluent poses a risk to ESA-listed marine mammals and sea turtles via exposure of individuals to pollutants in the effluent and plume and/or to the increased frequency or extent of HABs promoted by the effluent. The concentrations of metals and most other potentially toxic constituents in the discharge effluent plume are expected 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 for ESA-listed marine mammals and sea turtles.

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 expedite diminished 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 mortality and other health issues in marine mammals along the California coast. The potential increase in frequency and/or extent of HABs due to the discharge effluent poses an increased risk of mortality 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, and better understand the association of these events with the discharge effluent in order to more completely assess the potential exposure of ESA-listed marine mammals and sea turtles to these blooms as a result of OC San's continued discharge.

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 that may occur in the action area. We cannot discount these effects as extremely unlikely to occur or dismiss them as insignificant.

However, 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 given the available information. Long-term effects for individuals, including diminished reproductive capacity and lower survival rates, could result from continued accumulation of potentially harmful contaminants, which is further accelerated by the proposed action. More acute effects, such as physical impairments, reduced foraging, disorientation, and even death are possible effects of exposure to HABs, which may be increased by the proposed action. These affects 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 OC San'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 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.

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 in the environmental baseline (Section 2.4).

As described in the environmental baseline, six additional facilities discharge into the action area and are sources of pollutants in the offshore environment: three wastewater treatment facilities (Los Angeles County JWPCP, TIWRP, and South Orange County Wastewater Authority), one electricity generating station (AES Huntington Beach, LLC), and two oil and gas drilling/production platforms (Platform Esther and Platform Eva). Because these facilities discharge into State waters, the State has delegated authority from EPA to issue the NPDES permits for these facilities. Thus, the California Regional Water Quality Control Boards issue the

NPDES permits for these facilities. The continued discharge of waste and wastewater from these neighboring, non-federally permitted facilities could affect water and sediment quality within the action area. For example, the Los Angeles County JWPCP's discharge plume may enter the action area, adding additional pollutants, though at diluted concentrations given the distance from OC San's outfalls (29 km) (LACSD 2020).

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. Three river systems drain into the San Pedro Shelf: the Los Angeles River, San Gabriel River, and Santa Ana River. The Santa Ana River watershed, which covers over 9,320 km^2 and is home to approximately six million people, drains directly into the action area. Models indicate stormwater runoff plumes can spread throughout the San Pedro Shelf and persist for several days (Holt et al. 2017; cited in EPA 2021).

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 to ESA-listed species and designated critical habitat within the action area. Upcoming activities in the action area could include development of aquaculture facilities and decommissioning of existing oil and gas extraction infrastructure, which will also require future ESA section 7 consultation. Oil spills and the introduction of other pathogens and parasites could occur within the time frame of the permit and could affect ESA-listed species and designated critical habitat 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 of oil, the amount, local conditions, and the location. 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 our assessment of the risk posed to species as a result of implementing the proposed action. 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 (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 OC San reclamation and treatment plants. The purpose of the NPDES permit is to authorize the discharge of secondary treated wastewater and reverse osmosis (RO) concentrate (or brine) to the Pacific Ocean through Discharge Point 001 (the "120-inch outfall"). The NPDES permit would also authorize the discharge of secondary treated wastewater and RO concentrate to the Pacific Ocean through Discharge Point 002 (the "78-inch outfall") and Discharge Point 003 (Santa Ana River Overflow Weirs) in the event of an emergency or during planned essential maintenance or capital improvement projects on Discharge Point 001. 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 OC San'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 OC San's discharge may contain numerous other contaminants that could potentially harm ESA-listed species, but that the available information 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 within ESA-listed marine mammals and sea turtles feeding in the action area, increasing the potential for and rate at which adverse health effects to these species can occur. The occurrence and magnitude of exposure and adverse effects that we expect to result from the discharge of potentially harmful contaminants is uncertain, in part because levels of some POPs and other potentially harmful contaminants in the effluent have not been extensively monitored, and because of the variable potential exposure and response of individuals to the proposed action.

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 OC San'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 help initiate efforts by

EPA and the OC San 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 OC San'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 OC San's discharge. The proposed permit includes revised monitoring requirements for the influent, effluent, and receiving waters to better understand the nitrogen dynamics of OC San's discharge and the nitrogen loading that results from OC San's discharge into the action area. These revised monitoring requirements will improve our understanding of the proposed action's contribution to nutrient loading and HABs in the action area. We also expect the resulting monitoring data will help initiate efforts by EPA and OC San 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.

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 threats to the health of ESA-listed marine mammals and sea turtles that may visit the action area. Similarly, we expect the contributions of OC San's discharge to the overall health of the action area, and to the health of ESA-listed marine mammals and sea turtles, to persist as threats to the area as a whole and to ESA-listed species at an individual level. 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 the probability 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. 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 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 OC San'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 OC San'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 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 potential effects of the proposed action to reduce the likelihood of survival and recovery of blue whales, based on: (a) our current understanding of potential effects, as well as uncertainties regarding their magnitude and extent, in light of the status of the species in concert with other anticipated impacts; (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. 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 OC San'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 OC San'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 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 potential effects of the proposed action to reduce the likelihood of survival and recovery of fin whales, based on: (a) our current understanding of potential effects, as well as uncertainties regarding their magnitude and extent, in light of the status of the species in concert with other anticipated impacts; (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. 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 6,725 individuals, and it is most likely (~61% chance) that any individual present in the action area belongs to the Mexican DPS. 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 OC San'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 OC San'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 the Mexican DPS humpback whales to actions throughout their range, including specifically their presence and abundance in the SCB.

We do not expect the potential effects of 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 potential effects, as well as uncertainties regarding their magnitude and extent, in light of the status of the species in concert with other anticipated impacts; (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. The Central America DPS is estimated to consist of 1,877 individuals. While it is slightly less likely that any given individual that may be present in the action area will be a Central America DPS whale (~39% chance), 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 OC San'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 OC San'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 potential effects of 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 potential effects, as well as uncertainties regarding their magnitude and extent, in light of the status of the species in concert with other anticipated impacts; (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 Population

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. The WNP population of gray whales is very small (~290 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 OC San'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 OC San'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 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 potential effects of the proposed action to reduce the likelihood of survival and recovery of WNP gray whales, based on: (a) our current understanding of potential effects, as well as uncertainties regarding their magnitude and extent, in light of the status of the species in concert with other anticipated impacts; (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 Seal

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. The Guadalupe fur seal population is estimated to be at least 31,091 individuals, 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 OC San'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 OC San'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 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 potential effects of the proposed action to reduce the likelihood of survival and recovery of Guadalupe fur seals, based on: (a) our current understanding of potential effects, as well as uncertainties regarding their magnitude and extent, in light of the status of the species in concert with other anticipated impacts; (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. 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 10,000 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 OC San'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 OC San'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 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 potential effects of the proposed action to reduce the likelihood of survival and recovery of green sea turtles, based on: (a) our current understanding of potential effects, as well as uncertainties regarding their magnitude and extent, in light of the status of the species in concert with other anticipated impacts; (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. 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 and at high risk of going extinction. 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 OC San'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 OC San'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 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 potential effects of the proposed action to reduce the likelihood of survival and recovery of leatherback sea turtles, based on: (a) our current understanding of potential effects, as well as uncertainties regarding their magnitude and extent, in light of the status of the species in concert with other anticipated impacts; (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. 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 over 9,000, 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 OC San'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 OC San'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 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 potential effects of the proposed action to reduce the likelihood of survival and recovery of loggerhead sea turtles, based on: (a) our current understanding of potential effects, as well as uncertainties regarding their magnitude and extent, in light of the status of the species in concert with other anticipated impacts; (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. 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 OC San'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 OC San'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 potential effects of the proposed action to reduce the likelihood of survival and recovery of olive ridley sea turtles, based on: (a) our current understanding of potential effects, as well as uncertainties regarding their magnitude and extent, in light of the status of the species in concert with other anticipated impacts; (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 that may be affected by the proposed action, the environmental baseline within the action area, the effects of the proposed action, the effects of other activities caused by the proposed action, and cumulative effects, it is NMFS' biological opinion that the proposed action is not likely to jeopardize the continued existence of the following ESA-listed species: blue whales, fin whales, Mexico DPS and Central America DPS humpback whales, WNP population of gray whales, Guadalupe fur seals, East Pacific DPS green sea turtles, leatherback sea turtles, North Pacific Ocean DPS loggerhead sea turtles, and olive ridley sea turtles. 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). "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 and sea turtles 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 will be at increased risks of increased body burdens associated with the proposed action, although we expect that adverse effects will 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 will 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 OC San. 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 OC San. As we described in Section 2.5 (Effects of the Action), the levels of organophosphate flame retardants that are discharged by OC San have not been documented historically. Using available information from a similar adjacent WWTP, we estimated that OC San could discharge up to approximately 3,796 pounds (1721.8 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 18,980 pounds (8609.2 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 OC San by EPA, we expect OC San 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 EPA.

We also anticipate that all individual ESA-listed marine mammals and sea turtles residing or feeding in the action area would face increased risks of exposure to HABs, and subsequent risks of sublethal and lethal health effects resulting from those exposures. We expect all ESA-listed individuals that may enter or reside in the action area are at risk of exposure to increased HABs as a result of the proposed action, although we expect that adverse effects will generally be limited to relatively few individuals (adults or juveniles) of these populations.

At this time, we cannot predict the precise extent that OC San'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 OC San'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 nutrients, specifically nitrogen, that are being released into the action area. We elect to use the extent of total nitrogen 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 have therefore quantified the potential incidental take of the proposed action in terms of total nitrogen that we expect to be discharged by OC San. As we described in Section 2.5 (Effects of the Action), using information from Howard et al. (2014), we estimated that OC San's discharge increased total nitrogen in the action area by about 8,818 pounds (4,000 kg) of nitrogen per km^2 per year. Given the size of the action area, this equates to about 14.3 million pounds (6.5 million kg) of nitrogen over the course of a year being released into the area as a result of OC San's discharge.

As part of the proposed action, EPA requires OC San to monitor the influent, effluent, and receiving waters for parameters that include the several forms of nitrogen (e.g., ammonia, nitrate nitrogen, nitrite nitrogen). That data can be used to develop estimates of nitrogen loading resulting from OC San's discharge consistent with what has been done previously by Howard et al. (2014). Through this required monitoring under the proposed permit, which includes analysis of the nitrogen species being treated and discharged by OC San, we expect OC San to be able to monitor nitrogen levels in the discharge and estimate the total amount/loading of nitrogen discharge 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 EPA.

2.9.2. Effect of the Take

In this biological opinion, we have 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. No critical habitat has been designated or proposed for these species in the action area.

2.9.3. Reasonable and Prudent Measures

"Reasonable and prudent measures" (RPM) are nondiscretionary 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 OC San's discharge consistent with the surrogates described in Section 2.9.1 of this biological opinion, through the requirements placed upon OC San.

2.9.4. Terms and Conditions

The terms and conditions described below are non-discretionary, and the EPA or any applicant must comply with them in order to implement the RPMs (50 CFR 402.14). 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 OC San implement the CEC Monitoring Special Study required by the permit, 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) to measure levels of organophosphate flame retardants and other POPs in the effluent and the loading of receiving waters.

1b. EPA shall require that OC San collect the necessary data to support the ongoing monitoring of all nitrogen forms from OC San's discharge and nitrate and ammonia nitrogen from the receiving water. As part of the monitoring required by EPA, we expect OC San to be able to monitor nitrogen levels in the discharge and the action area and estimate the total level of nitrogen loading of the action area relative to the total amount/loading of nitrogen discharge to the action area that has been assumed and described above, through the requirements placed upon OC San.

1c. As part of the Strategic Process Studies and/or additional special studies that will specifically evaluate the effects of the GWRS final expansion, EPA shall require OC San to evaluate the potential for denitrification at the OC San Wastewater Treatment Plant.

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 OC San into the action area per year; the estimated total level of nitrogen (pounds or kg) discharged by OC San into the action area per year; and the results of OC San's evaluation of the potential for denitrification at the OC San Wastewater Treatment Plant. EPA may require OC San to directly submit the report to NMFS, provided that 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 OC San's estimated annual discharge of organophosphate flame retardants and/or estimated annual total level of nitrogen exceeds the total amounts/levels that have been assumed and described above.

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 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 OC San's discharge on the frequency and extent of HABs in the action area.

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 OC San's discharge. This could include:

   a) Generation of nitrogen form, timing, and mass balance data from upwelling and stormwater runoff events in the San Pedro Shelf area and the SCB to couple with the required generation of nitrogen data from OC San's discharge.

   b) Assess what HAB species are in the San Pedro Shelf area; 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 OC San or through the five-year Regional Bight Monitoring Program that examines multiple WWTPs within the area.

   c) Incident-specific monitoring of phytoplankton communities in the San Pedro Shelf area before, during, and after planned discharges from the 78" outfall, to evaluate the presence, composition, and extent of blooms related to the discharge in the nearshore area.

   d) Results of additional data collection, monitoring and/or evaluation can be provided to NMFS in a report or reports, submitted on a schedule to be determined.

2.11. Reinitiation of Consultation

This concludes formal consultation for re-issuance of a permit by EPA to the Orange County Sanitation District for wastewater discharge by OC San Reclamation Plan No. 1 (Fountain

Valley), Treatment Plant No. 2 (Huntington Beach), Collection System, and Outfalls under the NPDES.

As 50 CFR 402.16 states, 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 if: (1) The amount or extent of incidental taking specified in the ITS is exceeded, (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion, (3) 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 (4) a new species is listed or critical habitat designated that may be affected by the 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 OC San, specifically the total loading of organophosphate flame retardants. We estimated that OC San discharges approximately 3,796 pounds (1721.8 kg) of organophosphate flame retardants (TCEP, TCPP, and TDCPP combined) into the action area each year. If OC San's discharge of these organophosphate flame retardants per year is determined to be greater than this estimate (through the special studies required by 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 OC San, specifically nitrogen. We estimate that OC San discharges about 8,818 pounds (4,000 kg) of nitrogen per km^2, or about 14.3 million pounds (6.5 million kg) of nitrogen, into the action area per year. If OC San's discharge of nitrogen per year is determined to be greater than this estimate (through the monitoring required by 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 OC San'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, white abalone, or black abalone. We also do not anticipate the proposed action to adversely affect designated critical habitat for black abalone.

In our effects analysis, we identified three potential stressors to result from OC San'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, and critical habitat 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, and only 14 North Pacific right whales have been sighted off California since 1950 (NMFS 2017). As a result, we do not anticipate exposure of these whales to the stressors of the proposed action.

2.12.1.2. Sei whales

Similarly, sei whales have not been sighted during dedicated marine mammal surveys by NMFS in Southern California since 1991 (Carretta et al. 2020). Therefore, we do not anticipate exposure of these whales to the stressors of 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. 2020). 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 relatively rare; no sperm whales have been sighted during dedicated marine mammal surveys by NMFS in Southern California since 1991 (Carretta et al. 2020). Although the action area does include some areas of deep water canyons, occurrence of sperm whales foraging in the action area would be very rare and may involve prey that are not as 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 anticipate exposure of these whales to the stressors of the proposed action.

2.12.1.4. Conclusion for other ESA-listed marine mammals

Based on all of the above, we conclude that North Pacific right whales, sei whales, and sperm whales are not likely to be adversely affected by OC San's permitted discharge of wastewater. We do not anticipate exposure of these whales to the stressors of the proposed action. Therefore, effects of the proposed action on these three whale species is extremely unlikely to occur and are considered discountable.

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 temperature waters and are commonly found offshore and in productive coastal areas, though 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% 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. A few giant manta rays have been observed as bycatch in U.S. fisheries off California prior to 2010; none have been observed since 2010. 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.

Based on all of the above, we conclude that the proposed action is not likely to adversely affect giant manta rays. The risks of adverse effects from the proposed action are discountable, because the potential for the species to be exposed to the proposed action is extremely unlikely.

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 m, spending most of their time in areas between 20 to 80 m in depth (Erickson and Hightower 2007; Huff et al. 2011). While in marine waters, they may be feeding or simply migrating between estuaries.

Green sturgeon consist of two DPSs (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 in the California halibut set net fishery off San Pedro/Rancho Palos Verdes in 1993 (unpublished data from Rand Rasmussen, 18 July 2006). We are not aware of any additional records of green sturgeon within the vicinity of the action area. The available information indicates green sturgeon could occur in the action area, but 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 have been 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 occurrence in the action area and of exposure to OC San'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. As a result, we conclude that the effects of the proposed action on Southern DPS green sturgeon are discountable, because the likelihood that green sturgeon occur in the action area is extremely low based on the species distribution and habitat use. Therefore, the proposed action is not likely to adversely affect Southern DPS green sturgeon.

2.12.4. Oceanic whitetip shark

The oceanic whitetip shark was listed 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 m (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 not recorded any observed encounters with oceanic whitetip sharks (Young et al. 2018). For example, the California/Oregon 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).

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. Therefore, we do not anticipate exposure of oceanic whitetip sharks to the stressors of the proposed action. We conclude that the proposed action is not likely to adversely affect oceanic whitetip sharks. The risks of adverse effects from the proposed action are discountable, because the potential for the species to be exposed to the proposed action is extremely unlikely.

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 m (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, which 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 the presence of scalloped hammerheads in the action area is possible, the possibility of such an occurrence during the course of the proposed action 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.

Based on all of the above, we conclude that the proposed action is not likely to adversely affect the Eastern Pacific DPS of scalloped hammerhead sharks. The risks of adverse effects from the proposed action are discountable, because the potential for the species to be exposed to the proposed action is extremely unlikely.

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.

Within the action area, we expect the presence of individual adult and juvenile steelhead migrating to/from known steelhead watersheds (Los Angeles River, San Gabriel River, and Santa Ana River; Figure 5) tributary to the San Pedro Shelf. Based on our understanding, their presence in the action area is expected to be intermittent and short in duration (hours to a few days), occurring when they migrate to or from the freshwater environment. 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 (Pearcy et al. 1990).

Steelhead are not expected to encounter the principal areas of discharge (i.e., Discharge Point 001 and 002 and the ZID). Discharge Point 001 is the primary discharge point and extends approximately 4.5 miles offshore at about 197 ft (60m depth). Because this 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. Discharge Point 002 is only used during an emergency or during planned essential maintenance or capital improvement projects on the 120" outfall (Discharge Point 001). As a result, Discharge Point 002 is used infrequently and for short-duration. For this reason, and because Discharge Point 002 discharges at a depth of 65 feet (about 20m), we expect juvenile and adult steelhead are unlikely to be exposed to wastewater effluent from this discharge point.

Any exposure of steelhead to the wastewater effluent from Discharge Point 001 or 002 would be limited to low concentrations and short duration and would not be expected to result in lethal or sub-lethal effects. The discharge plume is typically detected at a depth of 98 to 131 feet, with detections in the upper 33 ft less than 2% of the time (Tetra Tech 2002, 2008; cited in EPA 2021). Thus, steelhead would have limited exposure to the discharge plume, given the location of steelhead in the very upper part of the water column. 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 not expected to facilitate accumulation of these constituents sufficient to cause 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 weights and 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 dissolve 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.

Overall, neither lethal nor sub-lethal effects to steelhead are expected to occur as a result of the proposed action. Based on this analysis, we conclude that the effects of the proposed action on steelhead are expected to be discountable or insignificant, and the proposed action is not likely to adversely affect endangered southern California steelhead.

Figure

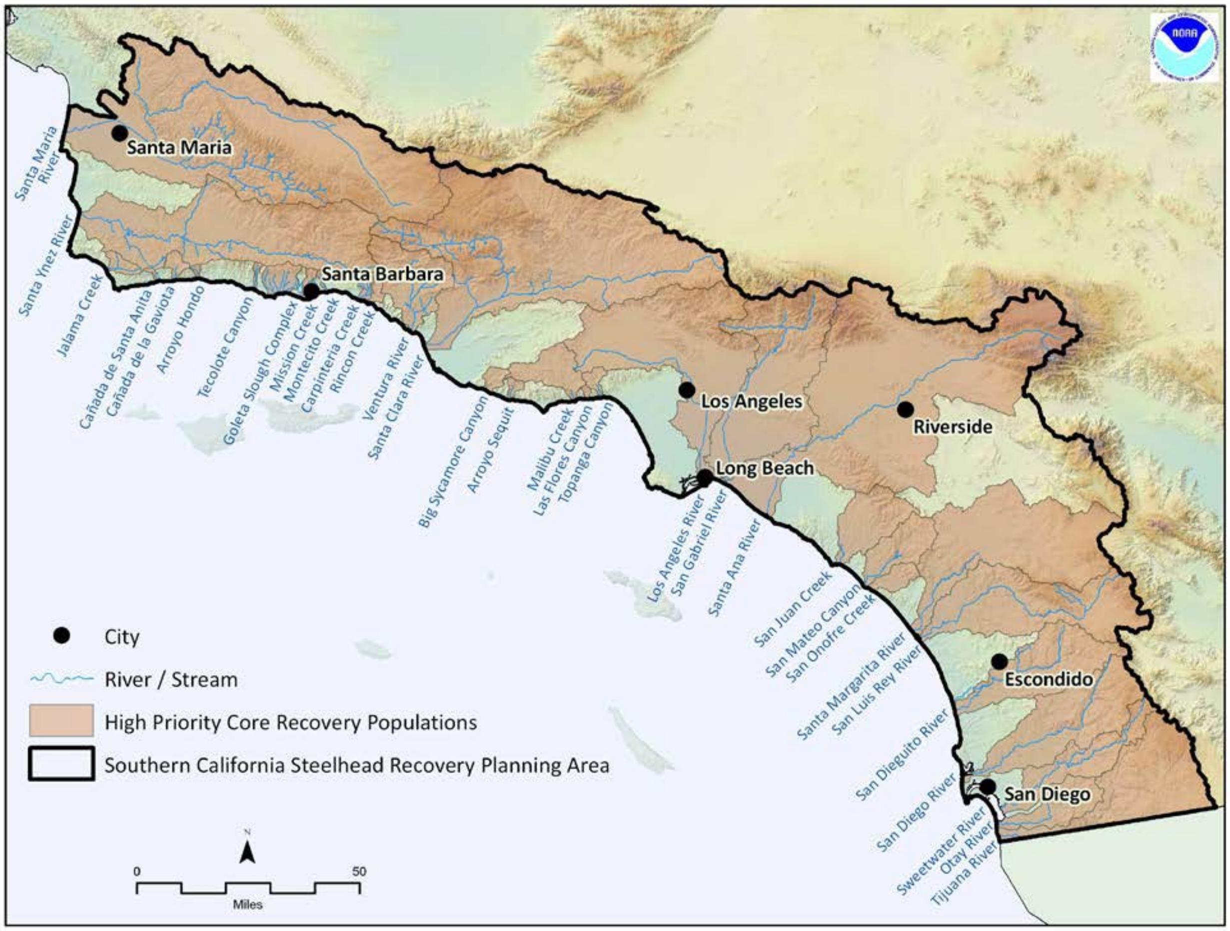

Figure 5. Southern California Steelhead Recovery Planning Area; maps shows the rivers and streams within the range of Southern California Steelhead (Figure 1-1 from Southern California Steelhead Recovery Plan (NMFS 2012a)).

2.12.7. 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).

NMFS 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. The fragmented populations that remain in the wild are likely unable to reproduce successfully or at levels needed for recovery (NMFS 2021a). White abalone are critically endangered, but much progress has been made toward recovery since 2001. 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, with additional outplanting efforts planned over the next five years

(NMFS 2021a). Expanded field monitoring off southern California and Mexico also supports improved assessments of the species' status in the wild (NMFS 2021a).

Comprehensive surveys for white abalone have not yet been conducted throughout the San Pedro Shelf, and thus limited information is available on white abalone presence within the action area. The action area primarily consists of soft bottom habitat, with limited rocky habitat within 60 m depths, except for Horseshoe Kelp/Reef, located about 15-20 km (~ 9-12 miles) to the northwest of Discharge Point 001 (Figure 6). Since 2010, increased survey efforts have resulted in new observations of white abalone along the mainland southern California coast (Neuman et al. 2015), including one white abalone found in Horseshoe Kelp/Reef. This individual was subsequently collected in 2016 to serve as broodstock for the captive breeding program, as authorized under ESA Permit No. 14344-2R, issued to the University of California, Davis – Bodega Marine Laboratory. We are not aware of any additional observations of white abalone within the action area in recent years.

The documented observation of at least one white abalone confirms that Horseshoe Kelp/Reef contains suitable habitat. White abalone are not likely to occur in the action area outside of Horseshoe Kelp/Reef, based on the lack of rocky habitat within suitable depths and lack kelp throughout the rest of the action area.

Additional white abalone may occur in Horseshoe Kelp/Reef and be exposed to OC San's discharge. Modeling of the discharge plume indicates that the plume can move north toward Horseshoe Kelp/Reef and beyond to Palos Verdes (Uchiyama et al. 2014). Monitoring of the discharge plume confirms that dispersal is primarily alongshore to both the north and south of the outfall (EPA 2021). Monitoring also indicates that the plume becomes increasingly diluted with distance from the outfall. The percent detection of the plume decreases to about 20% or less within about 10km (~6 miles) of the outfall, based on plume detections at the receiving water quality monitoring stations (a grid encompassing an approximately 6 km by 12 km area around the outfall) (EPA 2021). The plume is likely to become even more diluted by the time it reaches Horseshoe Kelp/Reef, which is at least 15-20 km (~9-12 miles) from the outfall. Thus, any white abalone in Horseshoe Kelp/Reef would be exposed to very diluted concentrations of the effluent. Based on this, we expect the exposure of white abalone to the plume to be insignificant and unlikely to result in adverse effects.

White abalone feed on macroalgae, including drift kelp. Kelp may take up pollutants from OC San's discharge and serve as a conduit for uptake of pollutants by white abalone. This is very unlikely, however, given the general lack of rocky bottom habitat and kelp within the action area, particularly within the vicinity of the outfall. Kelp and other macroalgae on Horseshoe Kelp/Reef may be exposed to pollutants in the discharge plume. However, as described above, Horseshoe Kelp/Reef would be exposed to very diluted concentrations of the discharge plume. Based on this, we expect the exposure of kelp and other macroalgae to the plume to be insignificant, and thus the potential uptake of pollutants by white abalone through ingestion of kelp and other macroalgae to be insignificant and unlikely to result in adverse effects.

As described in Section 2.5.3 (Harmful Algal Blooms) of the Effects Analysis, OC San's discharge could contribute to and promote HABs within the action area. Monitoring indicates that blooms are most prevalent near the outfall, ZID, and inshore (EPA 2021). Given the distance of Horseshoe Kelp/Reef from the outfall and ZID (about 15-20 km or 9-12 miles) and from shore (about 13km or 8 miles), we expect any white abalone in Horseshoe Kelp/Reef to have limited exposure to HABs resulting from or promoted by OC San's discharge. Based on this, we expect the exposure of white abalone to HABs resulting from the discharge to be insignificant and unlikely to result in adverse effects.

Based on all of the above, we conclude that the proposed action is not likely to adversely affect white abalone.

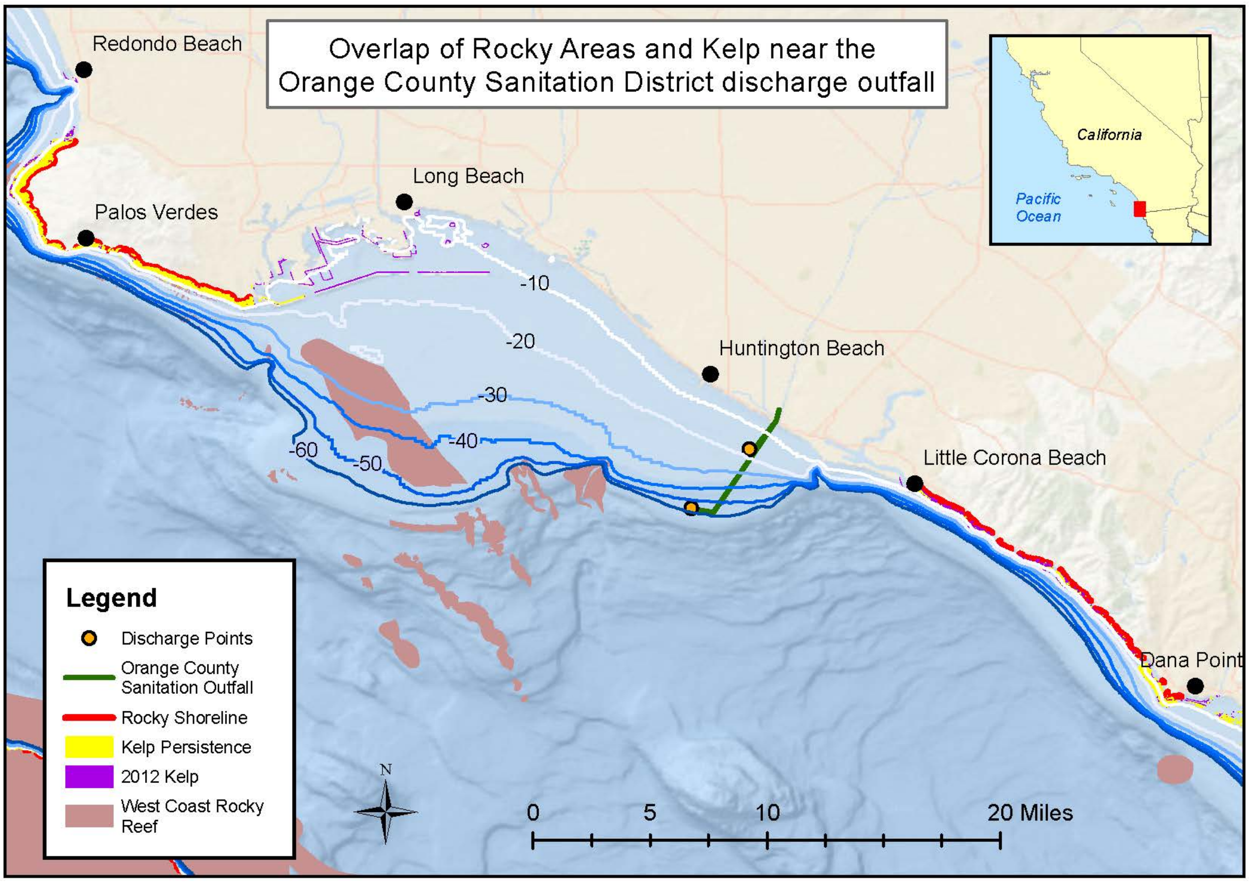

Figure 6. Map showing the presence of rocky shoreline habitat, rocky subtidal habitats, and kelp within the San Pedro Shelf and vicinity.

2.12.8. Black abalone

Black abalone range from Point Arena, California, to Bahia 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 2018). Overall, black abalone remain in danger of extinction.

Although the San Pedro Shelf is within the known range of black abalone, the likelihood of black abalone occurring within the action area is very low, given the limited amount of suitable habitat (i.e., rocky intertidal and shallow subtidal habitat) within the action area. Suitable habitat occurs on the north and south periphery of the action area (Figure 6). Black abalone have been observed on rocky intertidal reefs to the north on the Palos Verdes Peninsula as recently as 2015 (Eckdahl 2015). Rocky intertidal and shallow subtidal habitat also occurs to the south (i.e., Newport) adjacent to the action area, but black abalone have not been observed in this area in recent surveys (Chung and Peay 2020).

Monitoring of the discharge plume shows that dispersal is primarily alongshore to both the north and south of the outfall (EPA 2021) and modeling suggests the plume can move toward Palos Verdes (Uchiyama et al. 2014). These monitoring and modeling results indicate potential exposure of suitable habitat and any black abalone present in that habitat to the plume. However, we expect that exposure to be limited to very diluted concentrations of the discharge plume, based on the distance from the outfall. Suitable habitat to the north at Palos Verdes is about 32 km (~ 20 miles) from the outfall; suitable habitat to the south on the Newport coast is about 13 km (~ 8 miles) from the outfall. Monitoring data indicates that the plume becomes increasingly diluted with distance from the outfall, with the percent detection of the plume decreasing to about 20% or less within about 10km (~6 miles) of the outfall (EPA 2021). Based on this, we expect the exposure of black abalone to the plume to be insignificant and unlikely to result in adverse effects.

Like white abalone, black abalone also feed on macroalgae, including drift kelp. Kelp along the rocky shore at the Palos Verdes Peninsula and on the Newport coast may be exposed to the plume and could take up pollutants from this exposure. However, given the distance of these kelp beds from the outfall, we expect exposure to be limited to very diluted concentrations of the discharge plume. As described above for white abalone (Section 2.12.7), there is a general lack of rocky bottom habitat and kelp within the action area, particularly within the vicinity of the outfall. Kelp and other macroalgae on Horseshoe Kelp/Reef may be exposed to pollutants in the discharge plume, but at very diluted concentrations given the distance of this reef from the outfall. Based on this, we expect the exposure of kelp and other macroalgae to the plume to be insignificant, and thus the potential uptake of pollutants by black abalone through ingestion of kelp and other macroalgae to be insignificant and unlikely to result in adverse effects.

As described in Section 2.5.3 (Harmful Algal Blooms) of the Effects Analysis, OC San's discharge could contribute to and promote HABs within the action area. Monitoring indicates that blooms are most prevalent near the outfall, ZID, and inshore (EPA 2021). Given the location of suitable black abalone habitat at the north and south periphery of the action area and the distance from the outfall and ZID, we expect any black abalone in these habitats to have limited exposure to HABs resulting from or promoted by OC San's discharge. Based on this, we expect the exposure of black abalone to HABs resulting from the discharge to be insignificant and unlikely to result in adverse effects.

Based on all of the above, we conclude that the proposed action is not likely to adversely affect black abalone.

2.12.9. Black abalone critical habitat

NMFS designated critical habitat for black abalone in 2011 (76 FR 66806). The designation encompasses rocky intertidal and subtidal habitat (from the mean higher high water line to a depth of -6m relative to the mean lower low water line) within five segments of the California coast between Del Mar Landing Ecological Reserve to the Palos Verdes Peninsula, as well as on the offshore islands. Essential habitat features include rocky substrate (e.g., rocky benches formed from consolidated rock or large boulders that provide complex crevice habitat); food resources (e.g., macroalgae); juvenile settlement habitat (rocky substrates with crustose coralline algae and crevices or cryptic biogenic structures); suitable water quality (e.g., temperature, salinity, pH) for normal survival, settlement, growth, and behavior; and suitable nearshore circulation patterns to support successful fertilization and larval settlement within appropriate habitat.

Threats to black abalone critical habitat include coastal development and in-water construction projects (e.g., coastal armoring, pier construction or repair); activities that can increase sedimentation (e.g., sand replenishment, beach nourishment, side-casting); oil or chemical spills and response activities; and vessel grounding and response activities. Operations that involve withdrawing water from and/or discharging water to marine coastal waters may also affect black abalone critical habitat by increasing local water temperatures (e.g., discharge of heated effluent), introducing elevated levels of metals or other contaminants into the water, or altering nearshore circulation patterns.

The rocky intertidal and shallow subtidal habitats surrounding the Palos Verdes Peninsula (from the Palos Verdes/Torrance border to Los Angeles Harbor in southwestern Los Angeles County) are designated as critical habitat for black abalone. Modeling indicates that the plume can move toward Palos Verdes (Uchiyama et al. 2014). Based on this, designated critical habitat for black abalone on the Palos Verdes Peninsula may be affected by the proposed action.

Past long-term monitoring data indicate that Palos Verdes supported dense black abalone populations. Populations have declined severely due to disease, but critical habitat remains in fair to excellent condition. In particular, the area continues to provide good to high quality rocky substrate and food resources and fair to good settlement habitat for black abalone (NMFS 2011). Studies indicate past effects of wastewater discharges on black abalone critical habitat at Palos Verdes. Wastewater discharges likely contributed to the decline of kelp beds along the Palos Verdes Peninsula in the 1940s through 1960s, by increasing siltation, reducing light levels, and reducing water quality (Wilson et al. 1980). In the mid-1970s, kelp beds along the Palos Verdes Peninsula began to recover, due to a variety of factors including improvements in wastewater treatment and kelp restoration efforts (Wilson et al. 1980).

OC San is one of four WWTPs within the area surrounding the Palos Verdes Peninsula, the others being Hyperion, JWPCP, and TIWRP. Although OC San's discharge plume can reach Palos Verdes (Uchiyama et al. 2014), monitoring data shows that the plume becomes increasingly diluted with distance from the outfall (EPA 2021). Black abalone critical habitat on the Palos Verdes Peninsula is located about 32 km (~ 20 miles) from OC San's outfalls (Figure 6), in shallow nearshore waters where plume concentrations are expected to be relatively low. We expect the exposure of black abalone critical habitat to be limited to very diluted concentrations of the discharge plume, reducing the likelihood that the discharge will have substantial effects on essential habitat features (rocky substrate, food resources, juveniles settlement habitat, suitable water quality, suitable nearshore circulation patterns).

Overall, the discharge may affect critical habitat, but at an insignificant level that is not likely to reduce the quality of the habitat and its conservation value for black abalone. As a result, we conclude that the proposed action is not likely to adversely affect black abalone critical habitat.

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 2005), coastal pelagic species (CPS) (PFMC 1998), and highly migratory species (HMS) (PFMC 2007) 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, Coastal Pelagic Species, and Highly Migratory Species 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 (HAPC) for various federally managed fish species within the Pacific Coast Groundfish FMP. HAPC are described in the regulations as subsets of EFH which are rare, particularly susceptible to human-induced degradation, especially ecologically important, or located in an environmentally stressed area. Designated HAPC are not afforded any additional regulatory protection under MSA; however, federal projects with potential adverse impacts to HAPC will be 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. These discharges can impair functions of finfish, shellfish, and related organisms, such as 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 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), EPA identified 16 pollutants that are known to be present in quantifiable amounts in OC San's effluent. The EFH Assessment provided by EPA evaluated the effects of these pollutants, which include 11 metals (antimony, arsenic, cadmium, chromium, copper, cyanide, lead, nickel, selenium, silver, and zinc), total suspended solids, BOD, chlorine, ammonia, and nutrients. CECs were also analyzed in the EFH Assessment. In the following sections, we evaluate the adverse effects to EFH from these pollutants, including 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 directly from the EFH Assessment provided by EPA.

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. The concentrations of most metals in the plant influent have declined significantly since the 1977, largely due to OC San's source control pretreatment program. Arsenic, copper, cyanide, nickel, selenium, and zinc were most frequently detected in the final effluent from 2012-2019. However, concentrations of all detected metals in the effluent, after applying the initial dilution factor as prescribed by the 2019 California Ocean Plan, are below water quality standards. Moreover, the proposed permit includes performance goals for metals to ensure that treatment (i.e., removal efficiency) is maintained. Specifically, OC San must investigate the cause if a performance goal is exceeded twice consecutively. If a performance goal is exceeded in three successive monitoring periods, OC San must submit a written report on the nature of the exceedance, the results of the investigation, and the cause of the exceedance.

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 which results in 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. As a result, the EFH Assessment concludes that organisms temporarily entrained in or passing through the region surrounding the diffuser where initial dilution occurs, referred to as the ZID, are not present long enough to be exposed to chronic or lethal toxicity effects. As noted previously, the ZID under critical conditions for Discharge Point 001 (120" outfall) was estimated to extend 31.5 to 1220.5 feet (9.6 to 372 m) in length (EPA 2021).

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. OC San has been using topsmelt and purple sea urchin for acute and chronic toxicity tests, respectively. Because 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 proposed permit also requires monthly chronic toxicity testing and quarterly acute toxicity testing. Results are reported as either "Pass" or "Fail" following the Test of Significant Toxicity (EPA 2010). OC San's effluent received a "Pass" for all chronic and acute toxicity tests performed from 2012 through 2019 for discharge through Discharge Point 001 (EPA 2021) (Attachment F of the proposed permit).

3.2.3. Nutrients and HABs

As described above in Section 2.4.1.2 (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 and are the most frequently noted HAB species in the action area. Domoic acid is a water soluble neutrotoxin that accumulates in shellfish and planktivorous fish such as anchovy and sardine (Lefebvre et al. 2007, 2012; Smith et al. 2018). Although the effects to piscivorous birds and marine mammals are well documented and wide spread (Schnetzer et al. 2013; Smith et al. 2018), impacts to other species are less certain. Behavioral or schooling impacts to 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. Effects to zebrafish egg hatching and development were shown when domoic acid was microinjected into their eggs (Vasconcelos et al. 2010), but studies reporting effects to fish egg or larval development under realistic exposure scenarios were not readily found in a 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. However, 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. tamarense* 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 to 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 to capelin and Atlantic herring larvae and juveniles exposed to environmentally realistic concentrations of *A. tamarense* complex species through both vectorial poisoning and direct intoxication. Lefebvre et al. (2004) conducted a dietary uptake experiment with zebrafish larvae and found that 24 hour exposures to saxitoxin induced paralysis, and high levels 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 affect was transient to a few hours indicating there may be significant variability in effects to different species. However, impacts to 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 cell lyse could produce 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 for species whose 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 on the San Pedro Shelf and in the SCB as a whole, and what impact may be occurring to fish species.

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. 2014). Similar to all dense HABs, its effect to 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 to other species which 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 be to 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. 2020) 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) including on the San Pedro Shelf area adjacent to the OC San facility (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 to 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 fish species managed under the MSA.

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). The physical oceanography in the vicinity of the OC San 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 times of the year (Howard et al. 2012), nutrients from the OC San discharge may remain on the San Pedro Shelf for an appreciable amount of time. The San Pedro Shelf has been identified as a 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 San Pedro Shelf from the south due to the Southern CA Counter Current (Howard et al. 2012) and from winter runoff.

Information specific to the impacts on EFH by HABs in the action area is lacking. Impacts to CPS 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 ichthytoxins are impacting EFH 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. 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. The permit includes effluent limits for biological oxygen demand and total suspended solids, which have been met since full secondary treatment was implemented in 2012.

3.2.5. Contaminants of Emerging Concern

As described above in Section 2.4.1.1.3 of the Environmental Baseline, the term CEC refers to several types of chemicals, including persistent organic pollutants, 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 in 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 in 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 to 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 in 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 OC San) 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, levels of the active estrogen, 17β-estradiol (E2) were lower in both male and female hornyhead turbot collected from offshore of Orange County compared to other sample locations (Reyes et al. 2012). Although males sampled in other locations showed elevated levels of E2, these E2 levels 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. As described in Section 2.4.1.1.3 (Contaminants of Emerging Concern) of the Environmental Baseline, OC San has conducted CEC monitoring since 2014, annually monitoring the final effluent for pharmaceuticals, PCPPs, hormones, industrial EDCs, and PBDE flame retardants (EPA 2021). Frequently detected CECs included several PPCPs and several industrial EDCs, with mean concentrations varying widely among individual CECs. No PBDE flame retardants were detected (i.e., below the minimum detection levels), consistent with the 2018 Bight Regional Monitoring results showing non detect or low levels of PBDEs in coastal sediment samples (EPA 2021). Organophosphate flame retardants were not included in OC San's previous CEC monitoring, but will be included as part of the CEC monitoring required under the proposed permit. This is important given that organophosphate flame retardants have already been detected in the effluent for two adjacent facilities (Hyperion and TIWRP) (LASAN 2020).

OC San's receiving water monitoring program indicates recovery of the infaunal community in recent years since the implementation of full secondary treatment in 2012 and reduction/cessation of chlorine usage for disinfection in 2012-2015 (EPA 2021). Epibenthic macroinvertebrate and fish community health also shows minimal impact to these species, as no outfall-related trends in community measures were observed (EPA 2021). Community measures were within regional ranges, despite being highly variable across stations and between surveys (likely influenced by other factors, such as natural oceanographic events) (EPA 2021). The proposed permit will require continued and expanded CEC monitoring, as well as in-vitro cell bioassay monitoring to assess CECs in the receiving water. These special studies will provide useful information related to CECs in the action area and SCB.

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 equally or more toxic 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.

Areas of sediment contamination are present within the action area, much of which is a result of historical deposition and not associated with recent discharges from OC San. In particular, there is widespread contamination of DDT and PCBs in the action area and adjacent areas (e.g., Palos Verdes). A TMDL for DDT and PCBs in San Pedro Bay was developed to address this legacy contaminant issues (Harbor Toxics TMDL). Bioaccumulation of DDT and PCB in several species, including flatfish and rockfish managed under the Pacific Coast Groundfish FMP, has been analyzed and documented. In general, mean concentrations of DDT and PCB were higher in the liver tissue than in the muscle tissue for flatfish species (English sole and hornyhead turbot) (EPA 2021). Mean concentrations were similar across species and between outfall and non-outfall stations (EPA 2021). DDT and PCB concentrations in all fish samples were consistently below the status and federal advisory limits, despite a spike in PCBs in English sole in 2013-2014 and in rockfish in 2018-2019 (EPA 2021). Low site fidelity in the target species 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 (Burns et al. 2019).

Despite these legacy contaminant issues, benthic communities in the action area have 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). OC San's sediment monitoring indicates similar trends. In amphipod survival tests involving exposure to whole sediment samples from control and outfall-depth stations, there was no statistical difference in survival in all but one test from 2008-2019 (EPA 2021). The proposed permit will require quarterly, annual, and once per five year sediment monitoring for, among others, metals, pesticides, DDTs, PCBs, and chlorinated hydrocarbons (e.g., aldrin, dieldrin, endrin, chlordane, heptachlor, heptachlor epoxide, endosulfan), which should inform our understanding of these compounds as a potential source of sediment contamination.

3.2.7. Impacts to 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 eelgrass.

Kelp beds in the action area are primarily limited to shoreline areas at the north and south periphery of the San Pedro Shelf and potentially also occur at Horseshoe Kelp/Reef. None of these areas are in close proximity to the discharge point. The 120" outfall (Discharge Point 001) also discharges at a depth of 197 ft (60 m) and was designed to prevent nearshore transport of the effluent. Eelgrass habitat is not expected to be present within the action area.

3.2.8. Cumulative Impacts

Contaminants released into the action from six other permitted discharges result in cumulative impacts to EFH, including three other WWTPs (Los Angeles County JWPCP, TIWRP, and South Orange County Wastewater Authority). Stormwater runoff can also be a significant source of pollutants, as three river systems drain into the San Pedro Shelf (Los Angeles River, San Gabriel River, and Santa Ana River). Low flow diversions and treatment facilities have been effective at reducing bacteria and influent levels. When combined with other stormwater management practices, these diversions and facilities will improve water quality within the action area. In addition, increased water recycling/reclamation is expected to reduce discharge volumes from the WWTPs. Generally speaking, reduced flow, discharge prohibitions, and other NPDES permit requirements help improve water quality in the action area, although the potential for increased concentration of contaminants that are not removed during treatment or recycling processes remains a concern in some situations.

3.3. Essential Fish Habitat Conservation Recommendations

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 to 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.

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, localized impacts from discharge via Discharge Point 001 (120" outfall) have decreased, both in spatial extent and severity, over the past few years as a result of implementing full secondary treatment and a decrease in effluent volume. Increases in invertebrate and fish species abundance and diversity suggest the conditions around the outfall are progressing toward background conditions.

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., toxicity effects anticipated from increased water recycling, flame retardant and hormone concentrations in the effluent and loadings to the action area), special studies 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

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' 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 OC San, 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 two weeks at the NOAA Library Institutional Repository [https://repository.library.noaa.gov/welcome]. The format and naming adheres 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 ESA Consultation Handbook, ESA regulations, 50 CFR 402.01 et seq., and the MSA implementing regulations regarding EFH, 50 CFR 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 WCR ESA quality control and assurance processes.

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