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

Issuance of U.S. appropriations under the U.S.–Mexico–Canada Agreement (USMCA) Implementation Act

NMFS Consultation Number: WCRO-2022-02064

Action Agencies: U.S. Environmental Protection Agency Region 9 and U.S. International Boundary and Water Commission

Affected Species and NMFS' Determinations:

ESA-Listed Species	Status	Is Action Likely to Adversely Affect Species?	Is Action Likely to Jeopardize the Species?	Is Action Likely to Adversely Affect Critical Habitat?	Is Action Likely to Destroy or Adversely Modify Critical Habitat?
Green turtle, East Pacific DPS (*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	
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	

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?
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	
White abalone (*Haliotis sorenseni*)	Endangered	Yes	No	NA	
Giant manta ray (*Mobula birostris*)	Threatened	No		NA	
Green sturgeon, Southern DPS (*Acipenser medirostris*)	Threatened	No		NA	
Gulf grouper *(Mycteroperca jordani)*	Endangered	No		NA	
Oceanic whitetip shark (*Carcharhinus longimanus*)	Threatened	No		NA	
Scalloped hammerhead shark, Eastern Pacific DPS (*Sphyrna lewini*)	Endangered	No		NA	

ESA-Listed Species	Status	Is Action Likely to Adversely Affect Species?	Is Action Likely to Jeopardize the Species?	Is Action Likely to Adversely Affect Critical Habitat?	Is Action Likely to Destroy or Adversely Modify Critical Habitat?
Steelhead, Southern California DPS (*Oncorhynchus mykiss*)	Endangered	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		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: December 19, 2022

TABLE OF CONTENTS

LIST OF ACRONYMS v
1. Introduction 1
1.1. Background 1
1.2. Consultation History 1
1.3. Proposed Federal Action 3
1.3.1. Expanded ITP Capacity 4
1.3.2. Installation of Wastewater Conveyance System to Direct Flows to ITP 5
1.3.3. Tijuana Sewer Repairs 5
1.3.4. Advanced Primary Treatment Plant (APTP) Construction and Operation 5
1.3.5. Projected Changes to Untreated Wastewater Flows and SBOO Effluent Discharge 6
2. Endangered Species Act: Biological Opinion And Incidental Take Statement 8
2.1. Analytical Approach 9
2.2. Rangewide Status of the Species 9
2.2.1. Marine Mammals 12
2.2.2. Sea Turtles 24
2.2.3. White Abalone 37
2.3. Action Area 38
2.4. Environmental Baseline 42
2.4.1. Habitat and Environmental Health 43
2.4.2. Marine Mammals and Sea Turtles 55
2.4.3. Abalone 61
2.4.4. Other Federal Actions 62
2.5. Effects of the Action 62
2.5.1. Effects from Expanded ITP and Construction and Operation of APTP 64
2.5.2. Exposure and Response to the Toxicity of the SBOO's Effluent 66
2.5.3. Accumulation of Potentially Harmful Contaminants 75
2.5.4. Harmful Algal Blooms 83
2.5.5. Risks to Populations 91
2.6. Cumulative Effects 93
2.7. Integration and Synthesis 94
2.7.1. Marine Mammals and Sea Turtles 95
2.7.2. Abalone 102
2.8. Conclusion 103

2.9. Incidental Take Statement ... 104
2.9.1. Amount or Extent of Take ... 104
2.9.2. Effect of the Take ... 106
2.9.3. Reasonable and Prudent Measures ... 106
2.9.4. Terms and Conditions ... 107
2.10. Conservation Recommendations ... 109
2.11. Reinitiation of Consultation ... 111
2.12. "Not Likely to Adversely Affect" Determinations ... 112
2.12.1. Other ESA listed marine mammals ... 113
2.12.2. Giant manta ray ... 114
2.12.3. Green sturgeon, Southern DPS ... 115
2.12.4. Gulf grouper ... 116
2.12.5. Oceanic whitetip shark ... 116
2.12.6. Scalloped hammerhead shark, Eastern Pacific DPS ... 117
2.12.7. Steelhead, Southern California DPS ... 118
2.12.8. Black abalone ... 120
3. Magnuson–Stevens Fishery Conservation and Management Act Essential Fish Habitat Response ... 121
3.1. Essential Fish Habitat Affected by the Project ... 122
3.2. Adverse Effects on Essential Fish Habitat ... 122
3.2.1. Metals ... 122
3.2.2. Toxicity ... 123
3.2.3. Nutrients and HABs ... 124
3.2.4. Dissolved Oxygen, Biological Oxygen Demand, and Total Suspended Solids ... 126
3.2.5. Contaminants of Emerging Concern and Persistent Organic Pollutants ... 127
3.2.6. Sediment Contamination ... 128
3.2.7. Effects of In-water Construction Activities ... 129
3.2.8. Impacts on Sensitive Habitats ... 130
3.2.9. Cumulative Impacts ... 131
3.3. EFH Adverse Effects Determination ... 132
3.4. Supplemental Consultation ... 133
4. Data Quality Act Documentation and Pre-Dissemination Review ... 133
4.1. Utility ... 133
4.2. Integrity ... 133

4.3. Objectivity 134

5. References 134

Table of Tables

Table 1. 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. 17

Table of Figures

Figure 1. Action area encompassing areas where benthic monitoring stations are located in the vicinity of the coasts of La Jolla and Baja California. 39
Figure 2. Distribution of stations meeting potential plume criteria (pink) and those used as reference stations (green) near the PLOO during quarterly surveys in 2018 (left half of pie) and 2019 (right half of pie). 40
Figure 3. Distribution of stations meeting potential plume criteria (pink) and those used as reference stations (green) near the SBOO during quarterly surveys in 2018 (left half of pie) and 2019 (right half of pie). 41
Figure 4. Colored map showing potential white abalone habitat within the action area, primarily consisting of subtidal rocky reefs in nearshore waters (within 20-40 m depth) off La Jolla, the Point Loma peninsula, and Imperial Beach. 70
Figure 5. 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). 87
Figure 6. 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)). 120
Figure 7 Map of maximum observed eelgrass extent within the action area 131

LIST OF ACRONYMS

APTP	Advanced Primary Treatment Plant
BA	Biological Assessment
BIA	Biologically Important Areas
BOD	Biological oxygen demand
CA/OR/WA	California-Oregon-Washington
CCE	California Current Ecosystem
CEC	Contaminants of Emerging Concern
CFR	Code of Federal Register
CHL-*a*	Chlorophyll-*a*
CPS	Coastal pelagic species
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
EID	Environmental Information Document
ENP	Eastern North Pacific
EPA	Environmental Protection Agency
ERL	Effects Range-Low
ESA	Endangered Species Act
FMP	Fishery Management Plan
FR	Federal Register
HAB	Harmful algal bloom
HAPC	Habitat area of particular concern
HMS	Highly migratory species
IPPP	Isopropylated triphenyl phosphate
ITP	South Bay International Wastewater Treatment Plant
ITS	Incidental Take Statement
IUCN	International Union for Conservation of Nature
LACSD	Los Angeles County Sanitation Districts
LASAN	City of Los Angeles Sanitation and Environment
MGD	Million gallons per day
MMPA	Marine Mammal Protection Act
MSA	Magnuson-Stevens Fishery Conservation and Management Act
mtDNA	Mitochondrial DNA
NA	Not Applicable
ND	Not Detected
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOEC	No observed effect concentration

NPDES	National Pollutant Discharge Elimination System
OEHHA	Office of Environmental Health Hazard Assessment
PAH	Polycyclic aromatic hydrocarbon
PB-CILA	Planta de Bombeo-Comisión International de Limites y Aguas
PBDE	Polybrominated diphenyl ether
PBR	Potential biological removal
PCB	Polychlorinated biphenyl
PDO	Pacific Decadal Oscillation
PFMC	Pacific Fishery Management Council
POP	Persistent organic pollutant
PPCP	Pharmaceutical and personal care products
PSP	Paralytic Shellfish Poisoning
PSRG	Pacific Scientific Review Group
RPM	Reasonable and prudent measure
SABTP	San Antonio de los Buenos Wastewater Treatment Plan
SAR	Stock assessment report
SBOO	South Bay Ocean Outfall
SBWRP	South Bay Water Reclamation Plan
SCB	Southern California Bight
SCCOOS	Southern California Coastal Ocean Observing System
SCCWRP	Southern California Coastal Water Research Project
SDRWQCB	San Diego Regional Water Quality Control Board
SPLASH	Structure of Populations, Levels of Abundance and Status of Humpbacks
SWRCB	State Water Resources Control Board
T&Cs	Terms and conditions
T2IPPP	Tris(2-isopropylphenyl) phosphate
TBT	Tributyltin
TBOEP	Tris(2-butoxyethyl) phosphate
TCDD	Tetrachlorodibenzo-p-dioxin
TCEP	Tris(chloroethyl) phosphate
TCIPP	Tris(chloroisopropyl) phosphate
TCPP	Tris (chloropropyl)phosphate
TDCPP	Tris (1,3-dichloro-2-propyl) phosphate
TDS	Total dissolved solids (a measure of salinity)
TEP	triethyl phosphate
TIWRP	Terminal Island Water Reclamation Plant
TJRE	Tijuana River Estuary
TNBP	Tri-n-butyl phosphate
TPP or TPhP	Triphenyl phosphate
TPPO	Triphenylphosphine oxide
TSS	Total Suspended Solids
UME	Unusual mortality event
USFWS	U.S. Fish and Wildlife Service
USIBWC	U.S. International Boundary and Water Commission
WCR	West Coast Region
WNP	Western North Pacific

WWF	World Wildlife Fund
WWTP	Wastewater treatment plant
ZID	Zone of initial dilution

1. INTRODUCTION

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

1.1. Background

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

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

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

1.2. Consultation History

On July 5, 2022, the U.S. District Court for the Northern District of California issued an order vacating the 2019 regulations that were revised or added to 50 FR part 402 in 2019 ("2019 Regulations," see 84 FR 44976, August 27, 2019) without making a finding on the merits. On September 21, 2022, the U.S. Court of Appeals for the Ninth Circuit granted a temporary stay of the district court's July 5 order. As a result, the 2019 regulations are once again in effect, and we are applying the 2019 regulations here. For purposes of this consultation, we considered whether the substantive analysis and conclusions articulated in the biological opinion and incidental take statement would be any different under the pre-2019 regulations. We have determined that our analysis and conclusions would not be any different.

On February 26, 2021, members of the U.S. Environmental Protection Agency (EPA)-led NEPA planning team provided a joint presentation to the National Oceanic and Atmospheric Administration (NOAA), National Marine Fisheries Service (NMFS)'s WCR that included information on the planning effort underway as part of the USMCA Mitigation of Contaminated Transboundary Flows Project. At this point in the project cycle, the EPA-led team had developed 10 project alternatives that were under consideration in an Environmental Information Document (EID).

On July 7, 2021, once the draft EID was near completion, the NEPA planning team gave a second presentation to NMFS that provided an update on three tentative project alternatives that would be brought forward to the Programmatic Environmental Impact Statement (PEIS).

On August 4, 2021, the NEPA planning team provided a technical memorandum to NMFS. The technical memorandum intended to further facilitate early discussions between the EPA and NMFS in relation to marine wildlife listed under the Endangered Species Act (ESA) and Essential Fish Habitat (EFH) that may be affected by the project. The technical memorandum described the action area that could be affected based on the proposed suite of project options under consideration for evaluation in the NEPA process. The memorandum also contained a list of species that the EPA had determined could occur within this action area, although the memorandum did not determine if those species were likely to be adversely affected by any of the projects under consideration at that time. In addition to the species list, a table of key references which includes information on FR publication dates, dates of 5-year status reviews, recovery plans, and stock assessments of the ESA-listed species was also provided. These references were compiled in order to inform the basis of a comprehensive summary of life history information and current management status under the ESA in the Biological Assessment (BA). Lastly, the technical memorandum included a discussion of potential EFH in the action area identified during the development of the EID. EPA requested feedback from NMFS on the species list, EFH resources considered, and references table in the technical memorandum.

On August 25, 2021, NMFS provided an email response with comments relating to the technical memorandum. In addition to a correction on the name of the Western North Pacific (WNP) Distinct Population Segment (DPS) of gray whale, NMFS provided information on two candidate species not included in the technical memorandum—specifically, the shortfin mako shark and the sunflower sea star, noting that both of these species may occur in the action area and so should be included in the species list. NMFS also advised that impacts to species protected under the Marine Mammal Protection Act (MMPA) should also be considered. These include all marine mammals, which are managed according to MMPA stocks. While several species listed under the ESA are also protected under the MMPA, the respective management units may differ. Subsequently, marine mammals protected under the MMPA, including ESA-listed and candidate marine mammals, are considered in the PEIS. Lastly, NMFS pointed to an updated status review for Guadalupe fur seal and a new NOAA website hosting information on Biologically Important Areas (BIAs) for cetaceans. Some of these BIAs occur close to, or may overlap, the action area described in the memo and should be considered in any future assessment.

EPA completed the Alternative Analysis which identified the Alternatives 1 and 2 considered in the Draft PEIS. The consultation with NMFS includes only proposed actions associated with Alternative 1 (i.e., the Core Projects). ESA compliance for Supplemental Projects would be conducted at the time of the subsequent tiered NEPA analyses for those projects.

On May 5, 2022, EPA submitted to NMFS a preliminary draft combined BA and EFH Assessment report and solicited feedback regarding whether EPA should move forward with requesting official review pursuant to ESA Section 7 and the Magnuson-Stevens Fishery Conservation and Management Act (MSA). On May 25, 2022, EPA submitted to NMFS a draft

BA and EFH Assessment report for review with a request to initiate informal ESA and EFH consultation (see Appendix E of the Draft PEIS). On May 27, 2022, NMFS provided comments on the May 5, 2022 preliminary draft BA and EFH Assessment report and requested that EPA make appropriate revisions before submitting the BA and EFH Assessment to initiate consultation pursuant to ESA Section 7 and the MSA. Since receiving the comments from NMFS on the preliminary draft, EPA decided to separate the BA and EFH Assessments into their own distinct reports, rather than a combined report. The updated BA incorporates revisions intended to address NMFS's ESA Section 7 consultation-specific comments on the May 5, 2022 preliminary draft BA and reflects changes in the effects determinations for several ESA-listed species.

On July 22, 2022, NMFS received a letter requesting to initiate formal consultation pursuant to section 7 of the ESA along with a biological assessment (BA) of the USMCA mitigation of contaminated transboundary flows project. Subsequently on August 10, 2022, NMFS received a request for EFH consultation along with an accompanying EFH assessment.

Between the months of August and September, NMFS and EPA exchanged emails regarding clarifications of the following topics: details of the decommissioning activity and ROV surveys, and contaminant concentrations from the proposed action. On Oct 13, 2022, EPA requested a draft biological opinion be made available for their review and discussion prior to finalizing. On October 24, 2022, we transmitted a copy of a portion of the draft biological opinion. On Oct 26, 2022 and Nov 17, 2022, EPA and NMFS met to discuss the draft biological opinion. On Nov 22, 2022, NMFS provided revisions in response to comments from EPA, and we mutually agreed to include U.S. International Boundary and Water Commission (USIBWC) as an action agency for this proposed action. On December 15, 2022, EPA provided a letter to NMFS confirming that USIBWC is part of this proposed action.

1.3. Proposed Federal Action

Under the ESA, "action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by Federal agencies (see 50 CFR 402.02). Under the MSA, "Federal action" means any action authorized, funded, or undertaken, or proposed to be authorized, funded, or undertaken by a Federal agency (see 50 CFR 600.910).]

The proposed action is the issuance of U.S. appropriations under the U.S.–Mexico–Canada Agreement (USMCA) Implementation Act to the EPA and USIBWC to design and construct water infrastructure projects to address impacts from transboundary flows in the Tijuana River watershed and adjacent coastal areas. The Tijuana River originates in Mexico and flows northwest into the U.S. before discharging to the Pacific Ocean via the Tijuana River Estuary (ERG and Tenera Environmental 2022). These transboundary flows bring untreated wastewater, trash, and sediment through the Tijuana River watershed into the U.S., resulting in water quality impacts and human health concerns (ERG and Tenera Environmental 2022). For example, the transboundary flows have been linked to beach closures along the San Diego County coast. These transboundary flows and their effects have been occurring since at least the 1930s (ERG and Tenera Environmental 2022). The purpose of the USMCA Implementation Act appropriations (passed by Congress in January 2020) is to support wastewater and stormwater treatment projects in the Tijuana River area to reduce these transboundary flows of untreated wastewater, trash, and sediment into the Pacific Ocean.

The proposed action consists of four Core Projects:

1) Expand the South Bay International Wastewater Treatment Plant (ITP) from its current capacity of 25 MGD to 60 MGD;
2) Install a wastewater conveyance system from Matadero Canyon and Los Laureles Canyon in Mexico to convey dry-weather flows to the expanded ITP for treatment;
3) Rehabilitate or replace targeted sewer collectors in Tijuana that currently leak into the Tijuana River; and
4) Construct and operate a 35 MGD Advanced Primary Treatment Plant (APTP) for advanced primary treatment of diverted water from the existing Planta de Bombeo-Comisión International de Limites y Aguas (PB-CILA) diversion in Mexico.

EPA and USIBWC expect these above four Core Projects will reduce the transboundary flow of untreated wastewater from Mexico into U.S. waters off San Diego County. EPA and USIBWC also expects these Core Projects to increase the discharge of treated wastewater through the South Bay Ocean Outfall (SBOO) off San Diego County. In the following sections, we describe the four Core Projects as well as the outcome of conducting these four core projects on the existing wastewater facilities and operations.

1.3.1. Expansion of the ITP Capacity

The ITP is a U.S.-based facility owned by the USIBWC and operated by a contract operator (Veolia). The ITP treats wastewater from Tijuana with a primary and secondary treatment system. The ITP discharges effluent to the Pacific Ocean via the SBOO under a National Pollutant Discharge Elimination System (NPDES) Permit (#CA 0108928).

EPA and USIBWC propose to expand the ITP from its current capacity of 25 MGD to 60 MGD, to provide treatment for wastewater that would otherwise be discharged to the Pacific Ocean untreated. Currently, wastewater from the Tijuana region is collected and treated at three wastewater treatment plans (WWTPs) in Mexico. One of these WWTPs, the San Antonio de los Buenos Wastewater Treatment Plan (SABTP), does not effectively treat the wastewater and discharges approximately 28.2 MGD of untreated wastewater (primarily raw sewage) to the Pacific Ocean via the San Antonio de los Buenos (SAB) Creek (ERG and Tenera Environmental 2022). This expansion is expected to be completed no later than 2027.

With the proposed expansion, the ITP would be able to receive and treat additional wastewater that would otherwise be sent to the SABTP. This would reduce nearshore pollution along the southern San Diego coast by providing primary and secondary treatment of this untreated wastewater prior to being discharged via the SBOO. The proposed expansion would include enough capacity to accommodate additional wastewater flows produced by the future population of Tijuana, based on 2050 projections (ERG and Tenera Environmental 2022). The proposed expansion would also result in increased discharge flows through the SBOO, described in more detail in Section 1.3.5.

1.3.2. Installation of Wastewater Conveyance System to Direct Flows to ITP

EPA and USIBWC propose to install a wastewater conveyance system from Matadero Canyon and Los Laureles Canyon in Mexico to convey wastewater to the expanded ITP for treatment. This wastewater conveyance system would replace the existing system, which conveys wastewater from the canyons to the SABTP, where untreated wastewater is typically discharged to the Pacific Ocean. The new conveyance system would have a capacity of 12.7 MGD, an increase from the existing system's capacity of 6.3 MGD, to allow for flow increases over time. The purpose of the proposed wastewater conveyance system is to reduce the amount of dry-weather wastewater flows that are currently discharged with little to no treatment to the Pacific Ocean via the SAB Creek. The proposed system may also reduce the volume and frequency of dry-weather wastewater flows into the canyons. Construction activities, including components in Mexico, are projected to take approximately two years to complete following mobilization, but the specific schedule for starting and completing construction is not known at this time (ERG and Tenera Environmental 2022).

1.3.3. Tijuana Sewer Repairs

EPA and USIBWC propose to rehabilitate or replace targeted sewer collectors in the Tijuana metropolitan area, to reduce the amount of untreated wastewater in the Tijuana River to 5 MGD. Currently, sewage from the damaged sewer system leaks into the Tijuana River and eventually discharges to the Pacific Ocean. EPA and USIBWC state that the proposed sewer repairs would improve downstream water quality in the Tijuana River Valley and Estuary. First, the repairs would convey more wastewater to the ITP for treatment. Second, the repairs would reduce overall river flow volumes and thus reduce the frequency of dry-weather transboundary flows caused by river flow rates that exceed the PB-CILA diversion capacity. Construction activities, including components in Mexico, are projected to take approximately two years to complete following mobilization, but the specific schedule for starting and completing construction is not known at this time (ERG and Tenera Environmental 2022).

1.3.4. Advanced Primary Treatment Plant (APTP) Construction and Operation

EPA and USIBWC propose to construct and operate a 35-MGD APTP for advanced primary treatment of diverted water from the existing PB-CILA diversion. Associated construction activities include rehabilitating and extending an existing force main from PB-CILA to the new APTP, installing supporting facilities, and site modifications. However, these construction activities are not further discussed because land-based construction are not applicable to our analysis as they are not expected to have any effect on NMFS trust resources. Construction activities are projected to take approximately two years to complete following mobilization, but the specific schedule for starting and completing construction is not known at this time (ERG and Tenera Environmental 2022).

The purpose of this APTP is to increase capacity in the U.S. for treating diverted river water from Mexico that would otherwise be pumped to the SABTP and discharged to the Pacific Ocean. The APTP would operate independently of the existing ITP, but it would also discharge to the Pacific Ocean via the SBOO. The APTP would be designed to accommodate future expansion to 60-MGD and secondary treatment. However, this future expansion is not part of the proposed action.

1.3.5. Projected Changes to Untreated Wastewater Flows and SBOO Effluent Discharge

The purpose of the proposed action is to reduce the flow of untreated wastewater, trash, and sediments from Tijuana into the U.S. and Pacific Ocean off San Diego County. EPA and USIBWC estimate that implementation of the Core Projects will substantially reduce the discharge of untreated wastewater to the Pacific Ocean via the SAB Creek, sediment loads from untreated wastewater or river water, and biochemical oxygen demand (BOD_{5}) and nutrient loads into the Pacific Ocean, by up to 92-93% initially (when the Core Projects are complete but the ITP and APTP are not yet operating at full capacity) and by up to 88-89% in year 2050 when the ITP and APTP will be operating at full capacity to accommodate for estimated population growth in Tijuana (ERG and Tenera Environmental 2022).

EPA and USIBWC also estimate that implementation of the Core Projects will more than double the discharge of effluent via the SBOO. The SBOO is the pipe structure used to discharge treated effluent from the ITP and a second facility, the South Bay Water Reclamation Plan (SBWRP). The City of San Diego maintains and operates the SBOO, whereas the USIBWC maintains and operates the inland portion (i.e., the South Bay Land Outfall).

The proposed action would increase the discharge of effluent via the SBOO to an annual average of 76.4 MGD, due to expansion of the ITP's capacity from 25 MGD to 60 MGD and the construction and operation of the new APTP. Initial construction of the APTP will allow for 16.4 MGD of treatment with the facility designed to be increased to 35 MGD if needed. Currently, average discharge flows are approximately 29 to 31 MGD, based on data through 2020, with about 25 to 27 MGD coming from the ITP and 4 MGD from the SBWRP which is not part of the federal action (ERG and Tenera Environmental 2022; City of San Diego 2022a). From 2016 through 2019, the maximum discharge flow was 44 MGD (37 MGD from the ITP and 7 MGD from the SBWRP; ERG and Tenera Environmental 2022). Under the proposed action, EPA and USIBWC estimate an initial increase in the average discharge flow to about 61.4 MGD after the completion of ITP expansion by no later than 2027. The average discharge flow will gradually increase to approximately 76.4 MGD by 2050, as the ITP reaches its full expanded capacity and APTP is in operation in response to projected population growth in Tijuana. This increase in effluent discharge under the proposed action is within the SBOO's design capacity of 174 MGD average daily flow and 233 MGD maximum daily flow. The proposed action would not affect the SBWRP and its discharge through the SBOO, which is expected increase to over 8 MGD by 2050.

Given that the proposed action includes development of the capacity at the ITP and APTP to accommodate wastewater treatment and discharge needs for the estimated population growth in Tijuana out to the year 2050, we define the timeline of the proposed action as extending out to the horizon of operation of ITP and APTP at this capacity in 2050.

1.3.6 Recommissioning the SBOO to Accommodate Expanded ITP and Future APTP Operation

The SBOO is located off Imperial Beach just north of the border between the U.S. and Mexico. It was constructed in the 1990s and began discharging effluent from the ITP in 1999 and from the SBWRP in 2002. The SBOO extends approximately 3.5 miles (5.6 km) offshore to a depth of about 89 ft (27 m) (City of San Diego 2022a).The nearshore portion of the main barrel is buried underneath the sea floor out to approximately 2.5 miles (4.1 km) offshore where it transitions to a surface-laid portion that runs for an additional 0.9 miles (1.4 km) offshore. The SBOO terminates in a wye (Y-shaped) diffuser consisting of two legs, each 1,981 ft (604 m) long, with one run to the south and one to the northwest. The surface-laid portion of the main barrel and the wye diffuser legs are completely covered in ballast rock (ERG and Tenera Environmental 2022).

The SBOO was originally designed to discharge wastewater through 165 diffuser ports and risers, with one riser at the center of the wye and 82 risers along each diffuser leg. Due to consistently low flow rates, all of the risers along the northwest leg and most of the risers along the south leg are closed. Currently, there are only 18 open risers – a single open riser at the center of the wye and 17 open risers along the south leg, most of which are clustered at the south end of the south leg (ERG and Tenera Environmental 2022).

Under the proposed action, up to 55 diffuser risers on the south leg may be recommissioned (opened) in the near future (by no later than 2027) to accommodate for the increase in effluent discharge with the expanded ITP and construction and operation of the APTP. The recommissioning or opening of additional diffuser risers will likely be conducted in phases over several years (one before the expanded ITP becomes operation and second before the APTP becomes operational), and each phase is anticipated to take place over a period of several days or weeks. As part of construction activities to recommission or open the diffuser risers, divers will remove flanged covers on risers and replace these with port units which will result in removing a small area of habitat and species on and around the diffuser heads, affecting an approximately 6 x 6 ft (1.8 x 1.8 m) area of artificial reef habitat per diffuser riser. EPA and USIBWC expects natural ecological succession processes to gradually replace the lost habitat over time.

Vessels will be used to transport the divers and equipment to the site. If anchoring is required (anticipated needing one or two), the vessel will deploy anchors on sandy habitat to avoid damaging the wye diffuser and associated structures. It is likely that the anchor lines will remain under tension, but details will ultimately depend on configuration and operation choices of the specialized recommissioning contractor that will do the work. Alternatively, a permanent mooring may be used.

To minimize potential risk for ship strikes, at least one crew (most likely the vessel operator) maintains a constant watch of the ocean surface in front and adjacent to the vessel at all times. If marine mammals and sea turtles are observed distant to the vessel, vessel operators will adjust their course as necessary to ensure they do not disturb the natural behavior of these animals. If marine mammals are in close proximity, they will:

- Slow down and operate at a no-wake speed.

- Stay out of the path of the animal's direction of travel.
- Not put their vessel between whales, especially mothers and calves
- Not chase or harass animals, and will not approach the animals head-on, from directly behind them, or from the side. If animals are following a trajectory closely parallel to the direction of vessel travel, they will gradually steer the vessel to be parallel to the animals from the side and stay at least 100 yards away (i.e., the length of a football field).

During removal of flanged covers on risers, some noise will be produced; however, only relatively low-noise methods (e.g., hand tools) will be used, and louder activities (e.g., cutting and hammering) will be avoided.

Vessel activities bring a small risk of grounding or oil spill. Vessels are likely to carry hydraulic fluids and fuel that would be toxic to marine life if spilled. Most marine vessel groundings or spills are the result of mechanical failures or pilot negligence. However, it is assumed that construction vessel operators will follow best practice by maintaining their vessels to a high standard. Furthermore, vessel operators will maintain industry standard health, safety, and environmental standards that apply specifically to the intended construction operations. This is likely to include the storage and maintenance of spill kits appropriate for dealing with small vessel-based spills such as sand buckets, absorbent pads and cloths, and other emergency containment devices to stop small spills of hydraulic fluids and other polluting fluids from entering the water if they are accidentally spilled on deck. Vessels will be maintained to a standard that eliminates the likelihood of diesel or hydraulic oil spills during normal operation. In the case of a catastrophic loss of engine power that may result in a grounding, vessel captains must have procedures in place to raise Coast Guard support rapidly.

2. Endangered Species Act: Biological Opinion And Incidental Take Statement

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

We also determined the proposed action is not likely to adversely affect (NLAA) the following ESA-listed species: North Pacific right whales, sei whales, sperm whales, giant manta rays, Southern Distinct Population Segment (DPS) green sturgeon, Gulf groupers, oceanic whitetip sharks, Eastern Pacific DPS scalloped hammerhead sharks, Southern California DPS steelhead, and black abalone. These determinations are discussed in the in the "Not Likely to Adversely

Affect" Determinations section (Section 2.12). This opinion does not consider effects on critical habitat because none is designated in the action area.

2.1. Analytical Approach

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

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

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

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

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

2.2. Rangewide Status of the Species

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

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

Multiple studies have detected changes in the abundance, quality, and distribution of whale prey species in association with climate shifts, particularly with ocean warming. The nature and extent of effects have varied across study areas and species; however, in many cases, ocean warming has led to negative effects on these prey species. For instance, in the California Current Ecosystem (CCE), an anomalous warming of the upper ocean and weak upwelling event occurred from 2013–2016, often referred to as the "blob" or the "warm blob." During this period, sharp decreases in euphausiid biomass were observed, as evidenced by declines in both abundance and body length (Peterson et al. 2017; Harvey et al. 2017). Brodeur et al. (2019) Brodeur et al. (2019) compared samples collected in the Northern California Current region during years of cool (2011, 2012), warm (2000, 2002), and intermediate (2015, 2016) conditions and found that body condition of northern anchovy, Pacific herring, and Pacific sardine were better in cool years compared to warm years, and significantly so for anchovy and herring. During the anomalous warm blob event, sardine spawned earlier and appeared farther north within the Northern California Current than in previous years (Auth et al. 2018).

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

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

Climate change impacts that could affect abalone and its habitat include ocean acidification and elevated water temperatures. Ocean acidification could result in water quality conditions that reduce larval survival and shell growth and increase shell abnormalities (Crim et al. 2011). Studies show that effects of ocean acidification are highly species-specific due to differences between species in physiology, adaptability, and exposure to natural variation in ocean pH. There is a large degree of variability and uncertainty in climate change predictions, the timeframe over which changes may occur, and how the species and their habitat may respond. For example, abalone may be able to adapt to ocean acidification because they already experience natural variability in ocean pH, including low pH levels (Feely et al. 2004, 2008, 2009; Hauri et al. 2009). Increasing ocean water temperatures may occur due to global warming, and short-term and longer-term oceanographic conditions (e.g., ENSO or PDO events) may have varying effects on abalone. For example, warmer water temperatures may reduce food availability and quality by reducing macroalgal growth (Hobday et al. 2001; Tegner et al. 2001), and increase susceptibility to withering syndrome (Ben-Horin et al. 2013). At the same time, warmer water temperatures may benefit larval survival if temperatures move toward the optimum temperatures (Leighton 1972). Studies are underway to evaluate the effects of ocean acidification and increasing water temperatures on abalone, and to assess how other factors (e.g., presence of the disease vectors) may affect these interactions.

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

2.2.1. Marine Mammals

2.2.1.1 Blue Whale

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

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

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

Population Status and Trends: Though still depleted compared to historical abundance, blue whale abundance appears to be increasing in most if not all regions during the past several decades, although the data for most areas are sparse and uncertain (Calambokidis and Barlow 2004; Branch et al. 2007). Although there is insufficient data available to assess the present status in most parts of the North Pacific, the feeding stock of blue whales off the U.S. west coast has been estimated by line-transect and mark-recapture methods. Generally, the highest abundance estimates from line-transect surveys occurred in the mid-1990s, when ocean conditions were colder than present-day (Carretta et al. 2022). 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. 2022). Evidence for a northward shift in blue whale distribution includes increasing numbers of blue whales found in Oregon and Washington waters during 1996-2014 line-transect surveys (Barlow 2016) and satellite tracks of blue whales in Gulf of Alaska and Canadian waters between 1994 and 2007 (Bailey et al. 2009). Calambokidis and Barlow (2020) estimated blue whale abundance for the U.S. west coast at 1,898 whales, based on updated photographic ID data through 2018 using mark-recapture methods. Becker et al. (2020) estimated blue whale abundance at 670 whales, using habitat-based species distribution models from line-transect data collected from 1991 to 2018. The mark-recapture estimate (1,898) is considered the best estimate of abundance for 2018 due to its higher precision and because estimates based on line-transect data reflect only animal densities within the study area at the time surveys are conducted (Carretta et al. 2022). To put this in context, NMFS (2020) established recovery criteria for the ENP blue whale management unit as at least 2,000 animals for downlisting and 2,500 for de-listing. Population trends must also be stable or increasing for downlisting.[1]

Threats: Blue whales experienced intensive whaling throughout the 20th century, and the threat of directed hunting remains. Other threats that may be affecting blue whales with at least a potential for population-level consequences or are significant enough to contribute to the species' extinction risk include ship strikes, entanglement in marine debris and fishing gear, anthropogenic noise, and loss of prey base due to climate and ecosystem change (NMFS 2020a). It is difficult to estimate the numbers of blue whales possibly killed and injured by fishing gear, because large whales that become entangled in fishing gear may often die later and drift far enough to not strand on land after such incidents. Vessel strikes are also a threat to all large whales, including blue whales, although reported vessel strikes are considered a minimum accounting of the total. The threat to blue whales due to underwater noise, pollutants, marine debris, and habitat degradation, are difficult to quantify. However, there is a growing concern that the increasing levels of anthropogenic noise in the ocean may be a habitat concern for whales, particularly for whales that use low frequency sound to communicate, such as baleen whales.

For the ENP stock, the observed annual incidental mortality and injury rate (0.88/year) from vessel strikes is less than the calculated potential biological removal (PBR: 4.1) for this stock. This rate does not include unidentified large whales struck by vessels, some of which may have been blue whales, nor does it include undetected and unreported vessel strikes of blue whales (Carretta et al. 2022). 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

[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).

to be present in the U.S. West Coast EEZ (Rockwood et al. 2017). Blue whales have occasionally been documented entangled in pot/trap fisheries and other unidentified fishery gear on the U.S. west coast in recent years. The annual entanglement rate of blue whales (reported and assigned) during 2015-2019 is 1.39 whales annually (Carretta et al. 2022). Observed and assigned levels of serious injury and mortality due to commercial fisheries (1.39) exceed 10% of the stock's PBR (4.1); thus, commercial fishery take level are not approaching zero mortality and serious injury rate. Considering the effects of estimated vessel strikes, plus the effects of entanglements, human impacts may be exceeding the calculated PBR of 4.1 for this stock. While it is unknown if these same threats are occurring at similar levels throughout the global population of blue whales, the current levels of human impacts in the ENP are occurring at levels that may delay recovery of this stock of blue whales. Such a circumstance, especially if happening in concert with similar or larger impacts throughout their range, means that recovery of the entire species could be delayed altogether. The blue whale recovery plan (NMFS 2020a) describes recommended actions to determine the level of threat fishery entanglements, vessel strikes, and other potential threats pose to the likelihood of survival and recovery of the species.

2.2.1.2 Fin Whale

Fin whales were listed as endangered worldwide under the precursor to the ESA, the Endangered Species Conservation Act of 1969, and remained on the list of threatened and endangered species after the passage of the ESA in 1973 (35 FR 8491). Currently there is no designated critical habitat for fin whales. Fin whales feed on planktonic crustaceans, including *Thysanoessa* sp. (euphausiids) and *Calanus* sp. (copepods), and schooling fish, including herring, capelin, and mackerel (Aguilar 2009). Association with the continental slope is common, perhaps due to abundance of prey (Schorr et al. 2010). However, fin whales aggregate to areas with large amounts of prey regardless of water depth. For example, fin whales can feed in more shallow waters during the day (less than 330 feet), and feed in deeper waters at night (can be greater than 1,320 feet) (EPA 2017a).

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. 2022). 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). Indications of recovery in CA coastal waters date back to 1979/80, but there is now strong evidence that fin whale abundance increased in the California Current between 1991 and 2018 based on analysis of line stranect surveys (Carretta et al. 2022). Mean annual abundance increased 7.5% annually during 1991 to 2014 (Carretta et al. 2022).

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. Comprehensive coast-wide data on vessel strike deaths and serious injuries assumed to result in death are compiled in annual reports on observed anthropogenic morality for the 13-year period of 2007-2019 (Carretta et al. 2022). During this 13 year period, there were 20 observations of fin whale vessel strike deaths and 1 serious injury assumed to result in death or 1.6 fin whales annually (Carretta et al. 2022). Vessel strike mortality was recently estimated to be 43 fin whales per year, although this includes only the period from July – November when whales are most likely to be present in the U.S. West Coast EEZ (Rockwood et al. 2017). The threats to fin whales due to underwater noise, pollutants, marine debris, and habitat degradation, are difficult to quantify. However, there is a growing concern that the increasing levels of anthropogenic noise in the ocean may be a habitat concern for fin whales that use low frequency sound to communicate.

For the CA/OR/WA stock of fin whales, considering the effects of estimated vessel strikes (43/yr) and serious injury (2.2/yr) due to fisheries (0.64 /yr), the total incidental mortality from human impacts are less than the calculated PBR of 80 (Carretta et al. 2022). Total fishery mortality is less than 10% of PBR and, therefore, may be approaching zero mortality and serious injury rate (Carretta et al. 2022). However, observations alone underestimate true impacts due to incomplete detection of fishery entanglements. In recent years, there have been a number of instances where fin whales were sighted at-sea with indications of injury resulting from interaction with unknown fishing gear and other debris (NMFS WCR stranding data).

2.2.1.3 Humpback Whale, Mexico DPS and Central America DPS

Humpback whales were listed as endangered under the Endangered Species Conservation Act in June 1970 (35 FR 18319), and remained on the list of threatened and endangered species after the passage of the ESA in 1973 (35 FR 8491). A recovery plan for humpbacks was issued in November 1991 (NMFS 1991). On September 8, 2016, NMFS published a final rule dividing the globally listed endangered humpback whale into 14 distinct population segments (DPSs) and categorizing four DPSs as endangered and one as threatened (81 FR 62259). NMFS identified three humpback whale DPSs that may be found off the coasts of Washington, Oregon, California, and southern British Columbia: the Hawaii DPS (not ESA-listed), the Mexico DPS (ESA-listed as threatened), and the Central America DPS (ESA-listed as endangered). DPS abundance and geographic distribution are described below. Additionally, both the Mexico and Central America DPSs are composed of demographically independent populations (DIPs) of humpback whales (Curtis et al. 2022). While information about these DIPs are generally relevant to the status of the ESA-listed DPSs, DIPs are not used as the basis for any analyses in this biological opinion.

On April 21, 2021, NMFS designated critical habitat for the Mexico and Central America DPSs in the North Pacific Ocean that include portions of the CCE, including areas off the coasts of Washington, Oregon, California, and Alaska (for the Mexico DPS). However, the proposed action does not occur within designated critical habitat for either the Mexico or Central America DPSs.

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

Much of what we know about the current status of humpback whale DPSs results from field efforts conducted on all known winter breeding regions (2004-2006) and all known summer feeding areas (2004, 2005) for humpback whales in the North Pacific (Structure of Populations, Levels of Abundance and Status of Humpbacks (SPLASH)). This study, representing one of the largest international collaborative studies of any whale population ever conducted, was designed to determine the abundance, trends, movements, and population structure of North Pacific humpback whales as well as to examine human impacts on the population (Calambokidis et al. 2008). NMFS has relied upon results from the SPLASH study for abundance estimates as well as movement proportions between wintering (breeding) and summer (foraging) grounds (Bettridge et al. 2015; Wade et al. 2016; Wade 2017), even though the field efforts took place nearly fifteen years ago. For the 2004-2006 humpback populations, the Wade (2021) revised abundance estimate for the Central America DPS is 755 (coefficient of variation (CV)=0.242) animals, and the revised abundance estimate for the Mexico DPS is 2,913 (CV=0.066) animals, using the Multistrata model (Nmulti) which uses both winter and summer data (Table 4 in Wade 2021).

Recently, Calambokidis and Barlow (2020) provided an abundance estimate for the CA/OR/WA stock of humpbacks (and the Eastern North Pacific stock of blue whales), which has been included in the draft 2021 SAR, with revised trend information and other metrics to assist managers in assessing risk of human-related activities to this stock. Capture-recapture models for humpback whales off CA/OR showed a dramatic increase in recent years, with a trend for the population starting in 1989 (~500 animals) through 2018 increasing an average 7.5% per year, with a higher rate of increase in the late 2000s. Abundance estimate for the CA/OR/WA humpback whale stock is 4,973 (CV=0.048), with an Nmin of 4,776 animals. A recent publication from Curtis et al. (2022) estimated the abundance of the humpback whale DIP that includes whales off the U.S. West Coast that originate from breeding grounds in Central America and Southern Mexico (CentAm/SMex-CA/OR/WA) to be 1,496, and the abundance of the humpback whale DIP that includes whales off the U.S. West Coast that originate from breeding grounds in northern mainland Mexico (MMex-CA/OR/WA) to be 3,477. These estimates were generated using information from Calambokidis and Barlow in concert with updated photoidentification data collected in wintering areas of Central America and Southern Mexico from 2019 to 2021. These estimates will likely be used in future marine mammal stock assessments that formally recognize the humpback whale DIPs, and NMFS will continue to evaluate the relationship between the humpback whale DPSs and recognized DIPs.

Based on the information in the draft 2022 SAR (Carretta et al. 2022), Calambokidis and Barlow (2020), and Curtis et al. (2022), it is clear that there have been changes in the abundance and/or distribution of humpback whale DPSs over the last 10-15 years since the data gathered during the SPLASH project was re-analyzed by Wade (2021). In July, 2021, NMFS WCR finalized an approach outlining the most current evaluation of the distribution and relative abundance of ESA-listed DPSs that occur in the waters off the U.S. west coast for use in ESA analyses of Federal actions (NMFS 2021c). Table 2 summarizes the estimated abundance and proportion of each DPS off the U.S. west coast.

Based on NMFS WCR's July 2021 memo (NMFS 2021c), this biological opinion evaluates effects on both the Central American and Mexico DPSs of humpback whales as both are expected to occur in the action area in the relative proportions described below in Table 2. Specifically, we assume that 42% of the humpback whales present in the California (CA) and Oregon (OR) waters would be Central America DPS, and 58% would be associated with the Mexico DPS. To the extent that effects are evaluated at an individual level, these proportions would be used as the likelihood that the affected animal is from either DPS. Of note, in the southern California where the action area is proposed, one would more likely interact with a humpback whale from the Central America DPS. It is likely that new information from Curtis et al. (2022) and other sources will be used to inform future refinement of the assumed proportions of DPS off the U.S. West Coast at a later date as NMFS continues to evaluate the relationship between the humpback whale DPSs and recognized DIPs.

Table 1. Current estimates of abundance and relative proportions of the Central America DPS (endangered), Mexico DPS (threatened), and Hawaii DPS (non-listed) found off the U.S. west coast.

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

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

2.2.1.3.1 Mexico DPS

The Mexico DPS consists of whales that breed along the Pacific coast of mainland Mexico, the Baja California Peninsula and the 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 (NMFS 2021d). No specific estimate of the current growth rate for this DPS is available, but the documented positive growth rates of humpback whales along the U.S. west coast and in the North Pacific at large likely reflect growth in this DPS, given its relative population size. As described in detail in the recent NMFS WCR July 2021 memo, we assume that the population has increased by 6% annually over the last 15 years. Using Wade's (2021) population estimate of 2,913 whales based on information from 2004-2006 and the assumed 6% annual increase, the most current estimate for the entire Mexico DPS, only some of which occur along the U.S. west coast, is 6,981 animals. While Curtis et al. (2022) estimated the MMex-CA/OR/WA humpback whale DIP to be 3,477 animals, there is no additional information available on other portions of the Mexico DPS to generate any further refined total abundance estimates for the DPS.

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, although genetic and movement data have revealed that the wintering range of the Central America DPS likely extends beyond Central America and into southern Mexico. Humpback whales off the states of Oaxaca and Guerrero likely belong to the Central America DPS instead of the Mexico DPS, and some Central America whales have been sighted as far north as Michoacán and Colima (Taylor et al. 2021). Whales from this breeding ground feed almost exclusively offshore of California and Oregon in the eastern Pacific, with only a few individuals identified at the northern Washington–southern British Columbia feeding grounds. This DPS was determined to be discrete based on re-sight data as well as findings of significant genetic differentiation between it and other populations in the North Pacific. The genetic composition of the DPS is also unique in that it shares mitochondrial DNA (mtDNA) haplotypes with some Southern Hemisphere DPSs, suggesting it may serve as a conduit for gene flow between the North Pacific and Southern Hemisphere. The breeding ground of this DPS occupies a unique ecological setting, and its primary feeding ground is in a different marine ecosystem from most other populations. Loss of this population would also result in a significant gap in the range of the species (Bettridge et al. 2015).

Population Status and Trends: The Central America DPS of humpback whales occurs along the U.S. west coast, although individuals are more likely to be found off the coast of California and Oregon. Wade (2017) estimated the abundance of the Central America DPS to be 783 whales during the SPLASH period based on a revised analysis of the SPLASH data. Wade (2021) revised the estimate of the Central America DPS from SPLASH data to 755. These estimates are not considered a reliable estimate of current abundance; however, because they are more than eight years old and humpback whales in the Pacific have recently experienced positive growth rates (NMFS 2021d). No specific estimate of the current growth rate for this DPS is available, but the documented positive growth rates of humpback whales along the U.S. west coast and in the North Pacific at large likely reflect growth in this DPS, given its relative population size. As described in detail in the recent NMFS WCR July 2021 memo, we assume that the population has increased by 6% annually over the last 15 years. Using Wade (2017)'s population estimate of 755 whales based on information from 2004-2006 and the assumed 6% annual increase, the most current abundance estimate for the Central America DPS is 1,809 animals. Curtis et al. (2022) estimated the abundance of the CentAm/SMex-CA/OR/WA DIP to be 1,496 (CV=0.17) whales as of 2021. Given that previous abundance estimates for the Central America DPS did not include whales in southern Mexico, NMFS will continue to evaluate the relationship between the humpback whale DPSs and recognized DIPs.

2.2.1.3.3 Threats to Mexico and Central America DPS

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

Along the U.S. west coast, total annual human-caused serious injury and mortality of humpback whales is the sum of commercial fishery (25.2/yr)+non-commercial sources (1.4/yr)+estimated vessel strikes (22/yr), or 48.6 humpback whales annually. Considering the effects of estimated vessel strikes plus the effects of entanglements, human impacts may be greater than the PBR allocation of 28.7 animals for U.S. waters (Carretta et al. 2022). Other than the vessel strike estimates, most data on human-caused serious injury and mortality for this population is based on opportunistic stranding and at-sea sighting data and represents a minimum count of total effects. There is currently no estimate of the fraction of anthropogenic injuries and deaths to humpback whales that are undocumented on the U.S. west coast. Based on strandings and at sea observations, annual humpback whale mortality and serious injury in commercial fisheries (25.2/yr) is less than the PBR of 28.7; however, if methods were available to correct for undetected serious injury and mortality, total fishery mortality and serious injury would likely exceed PBR. Observed and assigned levels of serious injury and mortality due to commercial fisheries (≥ 25.2) exceed 10% of the stock's PBR (28.7), thus, commercial fishery take levels are not approaching zero mortality and serious injury rate (Carretta et al. 2022).

2.2.1.4 Gray Whale, Western North Pacific DPS

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

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

Population Status and Trends: The estimated population size from photo-ID data for Sakhalin and Kamchatka in 2016 was 290 whales (90% percentile intervals = 271 – 311; Cooke et al. 2018). Systematic counts of gray whales migrating south along the central California coast have been conducted by shore-based observers at Granite Canyon most years since 1967. The current minimum population estimate for non-ESA-listed ENP gray whales is 26,960 (Carretta et al. 2022). The most recent minimum estimate of endangered WNP gray whale abundance is 271 individuals (Carretta et al. 2022). 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. 2022). 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. 2022).

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 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. (2022), 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. In September, 2021, the UME was closed by NMFS. Strandings are seasonal and generally peak in April through June of each year. Strandings in Oregon and Washington became elevated starting in 2019 and have continued to present. Strandings in these two states in 2019 were five times higher than the historical average (NMFS WCR stranding program data). Most animals were young, around one year old, post-weaning (Norris et al. 2017).

Population Status and Trends: Commercial sealing during the 19th century reduced the once-abundant Guadalupe fur seal to near extinction in 1894. Population size prior to commercial harvest is not known, but estimates range from 20,000 to 100,000 animals (Fleischer 1987). Counts of Guadalupe fur seals have been made sporadically since 1954. A few of these counts were made during the breeding season, but the majority were made at other times of the year. García-Aguilar et al. (2018) estimates the current population size to be approximately one-fifth of historical, pre-exploitation levels. The most recent population estimate of at least 31,091 is based on pup count data collected in 2013 (García-Aguilar et al. 2018; Carretta et al. 2022). 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. 2022). In the U.S., a few Guadalupe fur seals are known to inhabit California sea lion rookeries at the Channel Islands (San Nicolas Island and San Miguel Island) (Stewart et al. 1987; National Marine Mammal Lab, unpublished data).

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

Minimal conflicts with fisheries exist. Gillnet and set-net fisheries likely take some animals, particularly in areas near Guadalupe Island and San Benito Island (Aurioles-Gamboa 2015). Juvenile female Guadalupe fur seals have also stranded in central and northern California with net abrasions around the neck, fish hooks and monofilament line, and polyfilament string (Hanni et al. 1997). Guadalupe fur seals occasionally are observed hooked in the Hawaii shallow set longline fishery. Between 2013 and 2017, there were two serious and two non-serious injuries involving this species (Carretta et al. 2022).

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. 2022), in addition to the fisheries interactions mentioned above. Guadalupe fur seals are also susceptible to domoic acid toxicity, bacterial pneumonia and other associated effects from emaciation/malnourishment (Norris et al. 2017). Military activities in southern California could affect Guadalupe fur seals through behavioral and physiological effects from mid-frequency active sonar, underwater detonations, and missile launches, as well as from sonic booms felt on the Channel Islands following a rocket launch. Scientific research is conducted on Guadalupe fur seals, primarily on San Miguel Island, including capture and tagging of pups, juveniles, and adult females. There have been no documented injuries or deaths associated with such research. Lastly, with oil production occurring off southern California and within the range of Guadalupe fur seals, the potential for an oil spill exists that could threaten this species, depending on the extent of the spill.

2.2.2. Sea Turtles

2.2.2.1 East Pacific DPS of Green Sea Turtles

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

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

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

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

Two foraging populations of green turtles are found in U.S. waters adjacent to the proposed action area. South San Diego Bay serves as an important habitat for a resident population of up to about 60 juvenile and adult green turtles in this area (Eguchi et al. 2010). There is also an aggregation of green sea turtles that appears to be persistent in the San Gabriel River and surrounding coastal areas (e.g., Anaheim Bay) in the vicinity of Long Beach, California (Lawson et al. 2011; Crear et al. 2016). This group of turtles has only recently been identified, and research on their abundance, behavior patterns, or relationship with the population in San Diego Bay is still in its infancy. Over the last decade of study, we have identified well over 50 different sea turtles occurring in the San Gabriel River/Anaheim Bay area (NMFS unpublished data) through research or strandings, although the duration of residence and/or transitory patterns of individuals in this area are the subject of ongoing research. Results from genetic sampling during monitoring programs or from strandings suggest that the lesser known Revillagigedo nesting population of green sea turtles is a significant source for southern California foraging (LeRoux et al. 2020).

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

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

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

2.2.2.2 Leatherback Sea Turtles

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

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

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

In the Pacific, leatherback nesting aggregations are found in the eastern and western Pacific. In the eastern Pacific, major nesting sites are located in Mexico, Costa Rica, and to a lesser extent, Nicaragua. Nesting in the western Pacific occurs at numerous beaches in Indonesia, the Solomon

Islands, Papua New Guinea, and Vanuatu, with a few nesters reported in Malaysia and only occasional reports of nesting in Thailand and Australia (Eckert et al. 2012). Leatherbacks nesting in Central America and Mexico migrate thousands of miles into tropical and temperate waters of the South Pacific (Eckert 1997; Shillinger et al. 2008). After nesting, females from the Western Pacific nesting beaches make long-distance migrations into a variety of foraging areas including the central and ENP (off of the U.S. west coast), westward to the 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 estimate leatherback population abundance is complicated by the life history of the species. Data collected at nesting beaches are often the best available data but do not provide information on life stages away from the nesting beaches (i.e., immature and mature males and immature females). Additionally, standardized nesting surveys are difficult to maintain over many, consecutive years and at all nesting beaches. Here we provide data that have been consistently collected using a standardized monitoring approach over a recent remigration interval, providing reasonable certainty that such data are representative of recent nesting at the identified beach. Although some data may have been collected at other nesting beaches, monitoring has not been recent, consistent, or standardized, thus limiting our certainty of these data; therefore data from those nesting beaches cannot be used to calculate abundance.

The East Pacific leatherback population has undergone dramatic declines over the last three generations and to date there is no sign of recovery (Wallace et al. 2013; NMFS and USFWS 2020a). In the eastern Pacific, nesting counts indicate that the population has continued to decline since the mid 1990s, leading some researchers to conclude that Pacific leatherbacks are on the verge of extirpation (Spotila et al. 1996; Spotila et al. 2000). In Costa Rica, a negative 15.5 percent trend in nesting females has been documented at Las Baulas (NMFS and USFWS 2020a). In Mexico, a positive trend has been recorded at some nesting beaches (i.e., Barra Cruz/Grande +9.5 percent), but a negative trend has been recorded in other areas (i.e., Cahuitan -4.3 percent). Overall, the current and potential future trend for the population is uncertain and additional years of data are needed to ascertain if recovery is occurring in Mexico. Using the best data available for the East Pacific population, NMFS and USFWS (2020b) calculated the index of total nesting females to be a minimum of 755 females. We consider this the best available estimate because it is based on a complete compilation of the most recent 4-year remigration interval data for each nesting beach monitored. Model-based estimates of abundance with credible or confidence intervals are unavailable for this population. This index of nesting includes females from known nesting beaches in Costa Rica and Nicaragua, and an estimated 70 to 75 percent of total nesting in Mexico (Gaona and Barragan 2016). It does not include females nesting at inconsistently monitored beaches in Mexico, including: Agua Blanca (40 km in Baja California), Playa Ventura, Playa San Valentín, Piedra de Tlacoyunque, and La Tuza (Martínez et al. 2007). Nesting is rare in other nations (e.g., Ecuador, El Salvador, and Panama) (Sarti et al. 1999).

The Western Pacific leatherback metapopulation that nests in Indonesia, Papua New Guinea, Solomon Islands, and Vanuatu harbors the last remaining nesting aggregation of significant size in the Pacific. This metapopulation is the source for the leatherbacks that occur off of the U.S. west coast annually. This metapopulation is made up of small nesting aggregations scattered throughout the region, with a dense focal point on the northwest Coast of Papua Barat, Indonesia. The Jamursba Medi and Wermon nesting beaches in this area represent an estimated 50 to 75 percent of all nesting in the West Pacific (NMFS and USFWS 2020a)). Using the best available data for the West Pacific leatherback population (Fitry Pakiding, University of Papua, pers. comm. 2020) and a Bayesian steady-state model, Martin et al. (2020) provided a median estimate of the total number of nesting females (i.e., over one remigration interval) at Jamursba Medi and Wermon beaches of 790 females, with a 95 percent credible interval of 666 to 942 females, as a snapshot of current abundance in 2017. We consider this to be the best available estimate of total adult female abundance at these two nesting beaches in 2017 (based on data from 2014 through 2017). To estimate the total number of nesting females from all nesting beaches in the West Pacific, we need to consider nesting at unmonitored and irregularly monitored beaches. As noted above, an estimated 50 to 75 percent of West Pacific leatherback nesting occurs at Jamursba-Medi and Wermon beaches (Dutton et al. 2007; NMFS and USFWS 2020a). Applying the conservative estimate of 75 percent to the Martin et al. (2020) estimate of 790 nesting females at Jamursba Medi and Wermon beaches, the total number of nesting females in the West Pacific population would be 1,054 females with an overall 95 percent credible interval of 888 to 1,256 females. It should be noted that this estimate (i.e., 1,054) of nesting females for the West Pacific population based on more recent available information is an update of the NMFS and USFWS (2020a) estimate (i.e., 1,277), which was based on a simple calculation that did not provide confidence or credible intervals.

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

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

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

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

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

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

Threats: The primary threats identified for leatherbacks are fishery bycatch and effects at or adjacent to the nesting beaches, including degradation of nesting habitat (erosion, logging, elevated sand temperatures, human/animal encroachment), direct harvest, and predation. In the western Pacific, leatherbacks are also subjected to traditional harvest, which was well documented in the 1980s and continues today. Traditional hunters from the Kei Islands continue to kill leatherbacks for consumption and ceremony.

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's documents or anticipates in U.S. domestic Pacific ocean fisheries.

In an attempt to develop a tool for managers to use locally (e.g., in an EEZ) to reduce threats in a particular area of interest, Curtis et al. (2015) developed biological "limit reference points" for western Pacific leatherback turtles in the U.S. West Coast EEZ, similar to a PBR approach calculated for marine mammal stocks. Depending on the model used and the various objectives sought (e.g., achievement of maximum net productivity, or no more than a 10% delay in the time for the population to rebuild) and incorporation of conservative assumptions accounting for broad uncertainty in abundance and productivity estimates, the limit reference point estimate for human-caused removals in the U.S. West Coast EEZ ranged from 0.8 to 7.7 leatherbacks over 5 years. Although these results are useful for consideration, NMFS is not currently using this approach to manage threats to sea turtles foraging within the U.S. West Coast EEZ, pending further discussion of how this approach or other approaches relate to the standards of the ESA. We anticipate that the management tool presented by Curtis et al. (2015) and other approaches to managing threats to sea turtles will be subject to future discussion by scientific experts.

2.2.2.3 North Pacific DPS Loggerhead Sea Turtles

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

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

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

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

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

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

As noted above, North Pacific loggerheads have been documented in high numbers off the central Pacific coast of Baja California, Mexico. Aerial surveys conducted from 2005 through 2007 in the Gulf of Ulloa, a known "hot spot," provided an estimated foraging population of over 43,000 juvenile loggerheads (Seminoff et al. 2014). NMFS conducted aerial surveys of the SCB in 2015 (a year when sea surface temperatures were anomalously warm and an El Niño was occurring) and estimated more than 70,000 loggerheads throughout the area (Eguchi et al. 2018), likely feeding on pelagic red crabs and pyrosomes, the species' preferred prey. Recent analysis of loggerhead sea turtle presence in the SCB suggests that loggerhead presence offshore of Southern California is tied not just to warm temperatures, but to persistently warm temperatures over a period of months such as what occurred during the recent large marine heatwave in the Eastern North Pacific Ocean (Welch et al. 2019).

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

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

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

Bycatch in commercial fisheries is a major threat throughout the species range, affecting both juveniles and adults; bycatch occurs in both coastal and pelagic fisheries involving longline, drift gillnet, set-net, bottom trawl, dredge, and pound net gear (Conant et al. 2009). Specifically in the Pacific, bycatch continues to be reported in gillnet and longline fisheries operating in "hotspot" areas where loggerheads are known to congregate (Peckham et al. 2007). Interactions and mortality associated with coastal and artisanal fisheries in Mexico and the Asian region likely represent the most serious threats to North Pacific loggerheads (Peckham et al. 2007; Conant et al. 2009; Ishihara et al. 2011).

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

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

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

Estimating the total number of sea turtle interactions in other Pacific fisheries that interact with the same sea turtle populations as U.S. fisheries is difficult because of low observer coverage and inconsistent reporting from international fleets. Lewison et al. (2004) estimated 2,600 – 6,000 loggerhead mortalities from pelagic longlining in the Pacific in 2000. Beverly and Chapman (2007) more recently estimated loggerhead and leatherback longline bycatch in the Pacific to be approximately 20% of that estimated by Lewison et al. (2004), which would equate to between 520 and 1,200 loggerhead mortalities during the year assessed. Chan and Pan (2012) estimated that there were approximately 1,866 total sea turtle interactions of all species in 2009 in the central and North Pacific by comparing swordfish production and turtle bycatch rates from fleets fishing in the central and North Pacific area. In 2015, a workshop was convened to analyze the effectiveness of sea turtle mitigation measures in the tuna RFMOs and 16 countries provided data on observed sea turtle interactions and gear configurations in the Western Central Pacific Ocean. Based on the information gathered there, 549 loggerhead sea turtle interactions were reported with a total estimate of 10,980 loggerheads caught in the region from 1989-2015 by these countries. Most recently, Peatman et al. (2018) estimated that between 473 to 2,941 loggerheads were captured in longline fisheries operating in the North Pacific from 2003-2017.

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

2.2.2.4 Olive Ridley Sea Turtles

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

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

Population Status and Trends: It is estimated that there are over one million female olive ridley sea turtles nesting annually along the Pacific coast of Mexico (NMFS and USFWS 2014). Unlike other sea turtle species, most female olive ridleys nest annually. According to the Marine Turtle Specialist Group of the IUCN, there has been a 50% decline in olive ridleys worldwide since the 1960s, although there have recently been substantial increases at some nesting sites (NMFS and USFWS 2007b). A major nesting population exists in the eastern Pacific on the West Coast of Mexico and Central America. Both of these populations use the north Pacific as foraging grounds (Polovina et al. 2004). Because the proposed action occurs closer to eastern Pacific nesting and foraging sites, we assume that this population would be more likely (i.e., than the western Pacific population) to be affected by the proposed action, and that any affected turtles may have originated from the endangered Mexican breeding population. The eastern Pacific population is thought to be increasing, while there is inadequate information to suggest trends for other populations. Eastern Pacific olive ridleys nest primarily in large aggregations called arribadas on the west coasts of Mexico and Costa Rica. Since reduction or cessation of egg and turtle harvest in both countries in the early 1990s, annual nest totals have increased substantially. On the Mexican coast alone, in 2004-2006, the annual total was estimated at 1,021,500 – 1,206,000 nests annually (NMFS and USFWS 2007b). Eguchi et al. (2007) analyzed sightings of olive ridleys at sea, leading to an estimate of 1,150,000 – 1,620,000 turtles in the eastern tropical Pacific in 1998-2006.

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

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

2.2.3. White Abalone

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

Population Status and Trends: White abalone face a high risk of extinction. NMFS's listed white abalone as endangered under the ESA in 2001 (66 FR 29046; May 29, 2001), primarily due to low densities resulting from historical overfishing. White abalone were subject to serial depletion by the commercial fishery in the early 1970s and suffered the most dramatic declines of the five abalone species (Karpov et al. 2000). During the main period of commercial harvest of white abalone (1969-1981), landings peaked in 1972, but declined to nearly zero by the early 1980s and remained low until the fishery was closed in 1996 (Karpov et al. 2000). Fishery independent surveys also show severe declines in abundance and density. Abundance estimates for the 1960s to 1970s ranged from about 600,000 to 1.7 million white abalone (Tutschulte 1976; Rogers-Bennett et al. 2002), whereas estimates for the 1990s were around 2,000 white abalone, or about 0.1% of estimated pre-exploitation abundance (Hobday et al. 2001). More recent surveys indicate greater numbers than previously estimated (e.g., about 1,900 animals at San Clemente Island and 5,800 animals at Tanner Bank in 2004) (Butler et al. 2006). ROV surveys conducted at Tanner Bank show continued declines in white abalone abundance and density over the period from 2002-2010, with fewer animals in close proximity to one another (Stierhoff et al. 2012).

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

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

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

Threats: In California, the species' abundance and density have declined substantially, resulting in low reproductive and recruitment success, such that the remaining animals in the wild do not appear to be replacing themselves. The primary threat to the species is their current low densities and spatial distribution, where animals may be too far apart to reproduce successfully or at levels needed for recovery. Complete and partial closures of the abalone fishery have been proposed in Mexico, but we do not know whether they have been adopted and implemented. Illegal harvest of undersized white abalone remains a problem in Mexico, but we have limited information on the problem's extent (NMFS 2008).

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

2.3. Action Area

see Figures 1 to 3; City of San Diego 2020b, 2021a) water conveyance system will be installed, areas in Tijuana where sewer repairs will be made, and where the APTP will be constructed. Although part of the action area, we do not discuss these land-based areas further because the ESA-listed species addressed in this opinion do not occur in these areas.

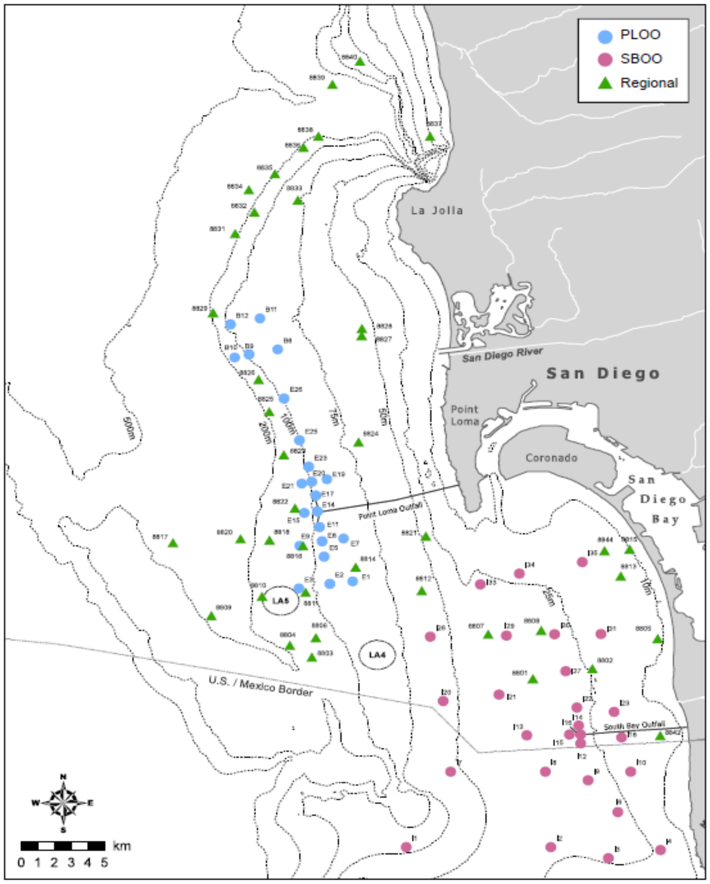

Figure 1. Action area encompassing areas where benthic monitoring stations are located in the vicinity of the coasts of La Jolla and Baja California.

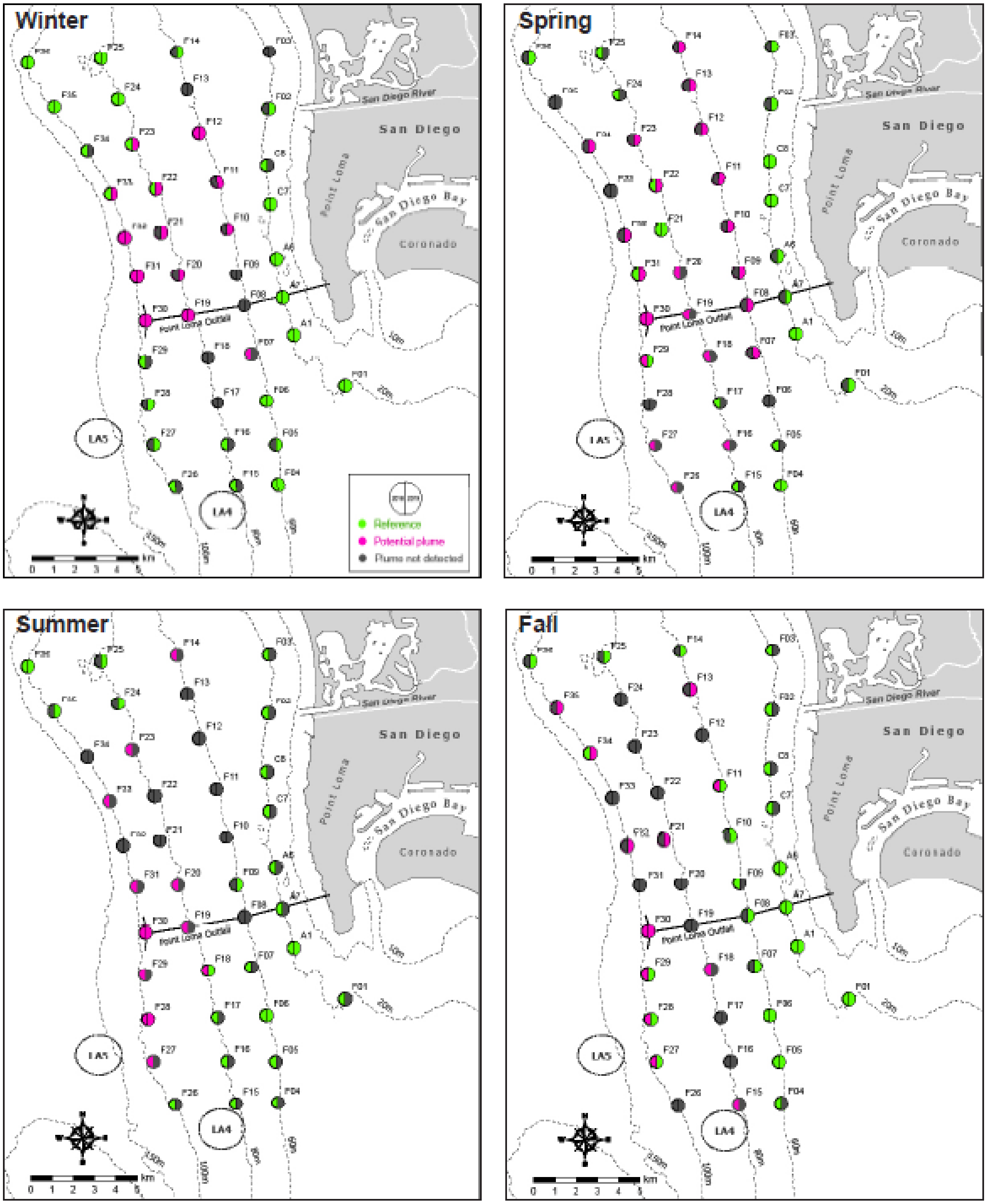

Figure 2. Distribution of stations meeting potential plume criteria (pink) and those used as reference stations (green) near the PLOO during quarterly surveys in 2018 (left half of pie) and 2019 (right half of pie).

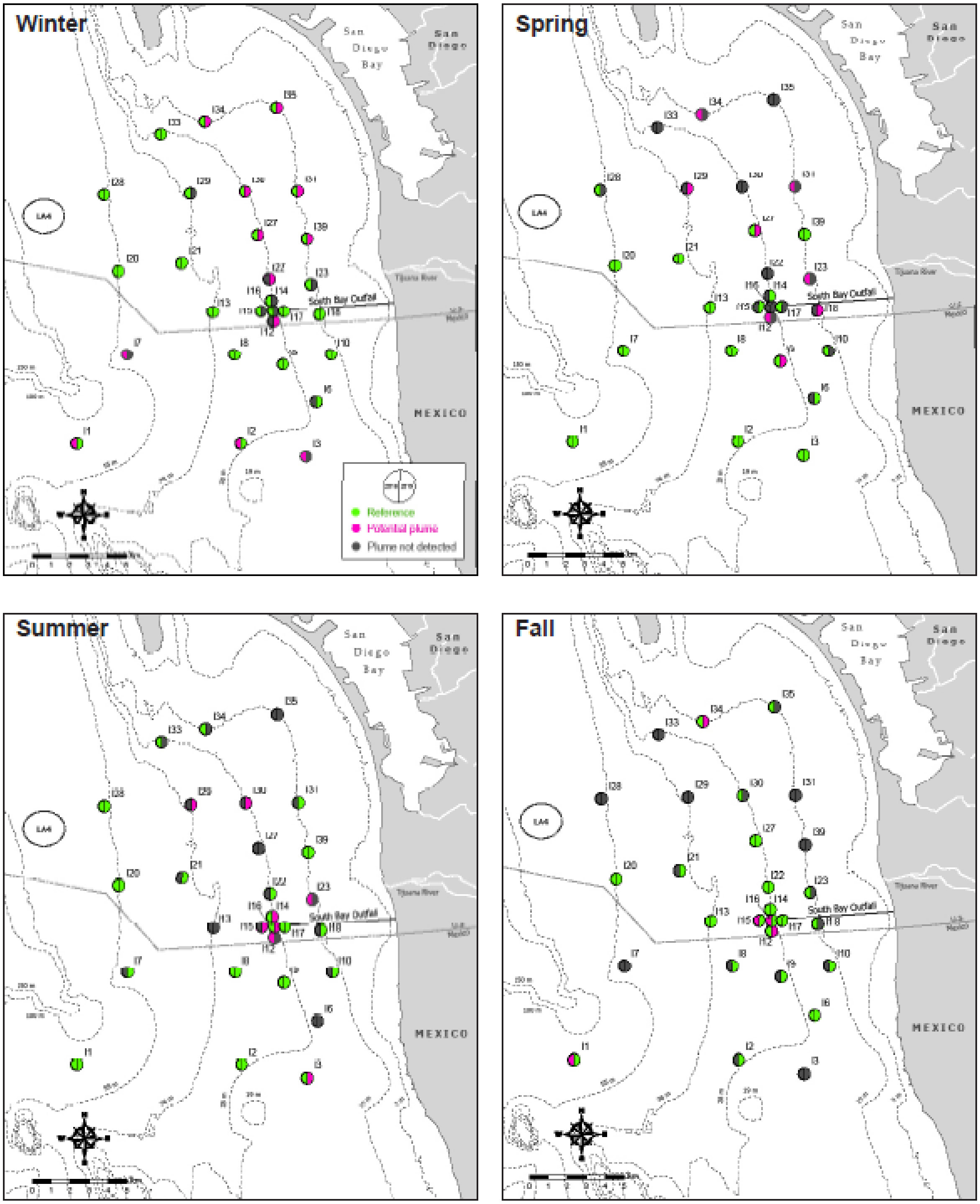

Figure 3. Distribution of stations meeting potential plume criteria (pink) and those used as reference stations (green) near the SBOO during quarterly surveys in 2018 (left half of pie) and 2019 (right half of pie).

Data from aerial imagery, models, and monitoring indicate that both effluent from the SBOO and from transboundary flows influence waters as far north as the Coronado Embayment (City of San Diego 2020b; Feddersen et al. 2021; Ocean Imaging 2021; ERG and Tenera Environmental 2022). The City of San Diego's (2020) discharge plume monitoring for the SBOO has detected the plume at stations located approximately 6.6 miles (10.6 km) upcoast and 4.9 miles (7.9 km) downcoast of the SBOO. The SBOO monitoring stations overlap with the monitoring stations for the Point Loma Ocean Outfall (PLOO; located approximately 10 miles or 16 km north; see Figure 1). Given the overlap, it is not possible to distinguish between the plume from the SBOO and the PLOO. Therefore, the action area encompasses areas where the plume has been detected from both the SBOO and the PLOO. This is consistent with our understanding that plume distribution and extent are influenced by oceanographic conditions and processes, which result in variable seawater movement within the action area including some parts of Mexico waters.

The shoreline along Imperial Beach and the surrounding area consists primarily of sandy beach (ERG and Tenera Environmental 2022). The Tijuana River, SAB Creek, and the SBOO discharge into this portion of the action area. The seabed consists predominantly of soft (sandy) substrate but also includes areas of hard (rocky) substrate, and kelp habitat. In historical surveys conducted within the action area surrounding the SBOO, soft sediment habitat made up about 80% of the surveyed area, with rocky reef habitat making up the remainder (ERG and Tenera Environmental 2022). A kelp forest lies at the mouth of the TJRE and inshore to the north of the SBOO discharge, covering an approximately 4.2 square mile area (ERG and Tenera Environmental 2022).

The SBOO structure provides an estimated 30 to 40 acres of artificial rocky reef habitat (ERG and Tenera Environmental 2022). Rock armoring (consisting of small to medium rock boulders) covers approximately one mile of the main barrel and the entire lengths of the northwest and south legs of the wye diffuser. The vertical risers and several access points along the SBOO pipeline also provide hard substrate. Footage from remotely operated vehicle (ROV) surveys indicates a healthy community of invertebrates, algae, and fishes associated with this artificial reef habitat.

At the northern extent of the action area, the shoreline along Point Loma consists primarily of rocky reef with cobble or sand pocket beaches (City of San Diego 2015a). In the nearshore waters, a large, six-mile long (10 km) kelp bed extends from the tip of Point Loma to the Mission Bay/San Diego River Jetty, from depths of 25 to 90 ft (7.6 to 27 meters) between 0.5 to 1 mile (0.8 to 1.6 km) from shore (City of San Diego 2015a). Beyond the kelp bed, the seafloor gradually slopes downward out to a shelf break at 350 ft (100 meters) depth, beyond which it continues to gradually slope downward to a depth of 1000 ft (305 meters). This shelf area consists primarily of unconsolidated bottom sediments (City of San Diego 2015a).

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., coastal marine waters off La Jolla to northern Baja California; see Section 2.3 Action Area). Given the large human population and high level of human activity in and around the coastal waters of the action area, 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 on the ESA-listed species and the quality of the habitat. We also reviewed the ESA consultation record (by conducting a search on NMFS's Environmental Consultation Organizer and the NOAA Institutional Repository) to identify other Federal actions that have occurred within the action area and have affected ESA-listed species.

2.4.1. Habitat and Environmental Health

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

Ocean conditions within the action area are affected by both regional- and local-scale currents, including the California Current System, large regional climatic processes (e.g., El Niño Southern Oscillations and Pacific Decadal Oscillations), and local gyres (City of San Diego 2021a). The meandering California Current has a large shoreward component near 32° N, splitting as it approaches the coast, and greatly affects the action area. Recent modeling by Kessouri et al (2020b) confirms previous work (Lynn and Simpson 1987; Mantyla et al. 2008; Rogowski et al. 2012a) by showing that local currents have considerable variability in position, strength, and depth, and can often be coming ashore right toward Pt. Loma. As a result, near-shore currents in the area have been observed by the City (City of San Diego 2016a, 2021a, 2021b) variously heading generally north or south directing the effluent towards Point Loma and La Jolla or the Mexican border. Plume monitoring by Terrill et al. (2009) indicates that the

SBOO plume typically remains at a depth of 8 m below the surface when the ocean is stratified, whereas the plume surfaces approximately 27 percent of the year during periods of weaker stratification. When the plume surfaces, it may reach the shoreline up to 25 percent of the time (Terrill et al. 2009).

The region encompassing the action area experiences warmer water conditions relative to the remainder of the SCB, due to entrainment of warm sub-tropical waters because of the region's oceanography (ERG and Tenera Environmental 2022). Seasonal changes in local weather patterns also affect ocean conditions, as the main driver of water column stratification (City of San Diego 2021a). Typically, the dry season from May to September is characterized by relatively warm waters and a more stratified water column, whereas the wet season from October to April is characterized by cooler waters and weaker stratification (Rogowski et al. 2012b, 2013). The transport and distribution of the discharge plume is affected by these oceanographic conditions, as well as more local influences such as outflow plumes from bays, rivers, lagoons, and estuaries; storm water discharge and surface runoff; seasonal upwelling patterns; and variable ocean currents or eddies (City of San Diego 2021a).

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

- South Bay International WWTP (the ITP, which will be modified by the proposed action) and SBWRP through the SBOO;
- Point Loma WWTP via the PLOO; and
- SABTP in Baja California

These WWTP facilities discharge either into U.S. waters or immediately to the south of U.S. waters, and are expected to continue to do so into the future. For facilities like the ITP and SBWRP that discharge into California State waters, EPA has delegated authority to the State to issue the NPDES permits through the SDRWQCB. As described above, the peak hydraulic capacity from the SBOO is 233 MGD with a total average design capacity of 174 MGD, although the dry weather discharge is typically much less (SDRWQCB 2021). The dry weather discharge primarily consists of effluent from the ITP, as the SBWRP recycles most of its tertiary treated water (EPA 2017b). The SBWRP is permitted to discharge up to 15 MGD (SDRWQCB 2021). The proposed action includes expansion of the ITP to receive flow that is currently going to the SABTP, and will result in increased discharge through the SBOO. The Point Loma WWTP discharges treated wastewater into Federal waters via the PLOO and is regulated by a joint permit issued by the SDRWQCB and the EPA. The PLOO has an overall design capacity of 240 MGD and currently discharges approximately 140 MGD of advanced primary treated effluent, with plans to reduce discharge flows by 30 MGD by 2025 and by 83 MGD by 2036 with implementation of the Pure Water San Diego potable water reuse program (City of San Diego 2021c). The SABTP treats wastewater collected from Tijuana and has an overall design capacity of 25 MGD; however, the current operations at the SABTP do not effectively treat the wastewater prior to discharge, resulting in approximately 28.2 MGD of untreated wastewater flowing out of SAB Creek into the Pacific Ocean (ERG and Tenera Environmental 2022). Importantly, as previously described, the proposed action would divert some of this flow to the expanded ITP and the APTP. Although not a WWTP, University of California, San Diego Scripps Institution of Oceanography also discharges aquaria wastewater, sea water, filter backwash water, stormwater, and urban runoff to the Pacific ocean (SDRWQCB 2019).

In addition to discharge of treated wastewater, a major source of wastes in the action area is from ongoing and seasonally variable transboundary flows of untreated and partially treated wastewater, trash, and sediment from Mexico. Deficiencies in the treatment, piping, and pump station network in Tijuana result in these transboundary flows of wastes through the Tijuana River and its tributaries into the canyons to the Tijuana River Valley and Estuary, and ultimately into the coastal waters of the Pacific Ocean in the action area (ERG and Tenera Environmental 2022). In the Tijuana River, flows mainly occur during the rainy season from October through April, characterized by intermittent by very large flows following storm events. An average wet season has about 96 days with river flows and results in approximately 9,000 million gallons of total flow over the season, though wet season flows can fluctuate from less than 1,000 to more than 25,000 million gallons (ERG and Tenera Environmental 2022). The dry season from May through September typically has less than 10 days of river flows with less than 100 million gallons of total flow over the season; however, malfunctions of the PB-CILA diversion system in Tijuana can result in extended periods of flow (between 20-30 MGD) even during the dry season (ERG and Tenera Environmental 2022). Runoff, trash, sediment, and untreated wastewater also flows into adjacent canyons and across the border into U.S. waters. The proposed action aims to reduce these transboundary flows from Mexico.

Regarding sediment quality, the 2018 Bight Regional Monitoring Program results indicate that multiple sources contribute to pollutants found in the sediment. Past discharge practices (now discontinued) have resulted in detectable levels of DDT and PCBs in almost all bottom-dwelling fish populations throughout southern California (SCCWRP 2012). Highest concentrations of DDT and PCBs are on or near the Palos Verdes shelf off Whites Point in Los Angeles, with fish tissue burdens declining to the north and south across the Southern California Bight. Concentrations of chlorinated hydrocarbons in fish from reference areas are now less than 5% of levels measured two decades ago (Allen et al. 2011). Contaminant burdens in fish tissues (e.g., PCBs, DDTs, and chlorinated hydrocarbons) at the PLOO and SBOO regions have been within ranges reported elsewhere in the SCB that do not indicate discernable patterns associated with proximity to either outfall (City of San Diego 2022a).

2.4.1.1 Water Quality in the Action Area

As described above in Section 2.4.1 (Habitat and Environmental Health), a number of factors influence water quality in the action area, including point source discharges (e.g., from WWTPs), regional non-point source discharges, local river outflows, and other local non-point sources (e.g., harbors, marinas, storm drains, and urban runoff; City of San Diego 2015a). These include wastewater discharges from the SBOO and transboundary flows from Mexico that would be affected by the proposed action.

Marine mammals, sea turtles, and abalone that are found off the coast of California can be exposed to relatively high levels of contaminants because they are generally long-lived species that are in close proximity to urban areas with high human activity. Moreover, production and/or use of some contaminants (e.g., PCBs, dioxin, furans, DDTs) have been banned (Miniero et al. 2015) but because of their persistent properties, these contaminants remain in the ecosystem and continue to impact marine species for many years including the future. Here, we describe the essential elements and metals, POPs, and other CECs that are found in the action area and that adversely affect ESA-listed species. Additional informational regarding water quality and potential impacts to habitat and marine life can be found in Section 3.2 (Adverse Effects on Essential Fish Habitat).

2.4.1.1.1 Metals and Ammonia

Metals are naturally found in the environment and some are essential to an animals' nutrition. However, human activities can increase the concentrations and metals can become toxic at certain exposure levels. Most metals settle to the ocean floor where they can accumulate in sediment. The City of San Diego (2019) provided PLOO effluent monitoring data for 2017 and ERG and Tenera Environmental (2022) provided SBOO effluent monitoring data for 2021, showing varying concentrations of metals including arsenic, chromium, copper, iron, lead, mercury, silver, and zinc. Data for additional years was not provided for PLOO or the SBOO and were not available in the latest monitoring reports; therefore, we could not assess trends in the concentrations of metals in the effluent over time. As for metals measured in fish, the City of San Diego (2022a) reported that 13 out of the 18 trace metals analyzed were detected in fish liver tissue samples and 10 out of the 18 trace metals were detected in fish muscle tissue samples from the PLOO and SBOO trawl zones in 2020-2021. The metals detected in both liver and muscle tissues included arsenic, cadmium, chromium, copper, iron, mercury, selenium, and zinc. Metal concentrations in liver and muscle tissues can be highly variable and occasionally relatively high concentrations are detected. However, metal concentrations are consistently within the ranges reported elsewhere in the SCB and have not indicated metal contaminant accumulation in the vicinity of the PLOO or SBOO that could be associated with wastewater discharge (City of San Diego 2022a).

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 2017a). In 2017, ammonia nitrogen concentrations in Point Loma WWTP's effluent averaged 40.3 mg/L, with a maximum recorded value of 44.5 mg/L (City of San Diego 2019). In 2021, ammonia nitrogen concentrations in the SBOO's effluent ranged from a monthly average of 0.87 to 55.6 mg/L, with a maximum daily value of 57.4 mg/L (ERG and Tenera Environmental 2022). The City of San Diego (2019) stated that monitoring for ammonia nitrogen in the vicinity of the PLOO indicates that after initial dilution, concentrations in the plume are at or near detection levels and virtually indistinguishable from natural background levels. Additional monitoring data for ammonia levels in the effluent, receiving waters, or fish tissue was not provided or available in the latest monitoring reports.

Discharge of untreated wastewater from SAB Creek and other transboundary flows also result in loading of metals and nutrients, including ammonia, into the action area (ERG and Tenera Environmental 2022). The metal and nutrient concentrations from these sources is not known, but is estimated to be higher than the concentrations discharged via the SBOO and PLOO, which discharge wastewater that has been treated to reduce metal and nutrient levels below a permitted concentration (ERG and Tenera Environmental 2022).

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 (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. Monitoring data for the Point Loma WWTP effluent show PCB levels below the detection limit and below applicable numeric effluent limits and the CA Ocean Plan's water quality objectives (City of San Diego 2021a). Monitoring data for 2017 also showed that DDT levels in the effluent were below the detection limit (City of San Diego 2019). Concentrations of chlorinated pesticides (e.g., DDT, PCB, and PAHs) in the sediments at Point Loma are generally low, except for DDE (a breakdown product of DDT)(City of San Diego 2015b). DDE levels at Point Loma are within the range of sediment concentrations found elsewhere in the SCB, likely due to historical discharges (City of San Diego 2015b).

Monitoring data for the ITP's effluent in 2021 show total PCB levels averaging 0.4 µg/L per month and mass emissions ranging from 0.06 to 0.08 lbs/day (ERG and Tenera Environmental

2022). The average monthly DDT level in 2021 was 0.01 µg/L (ERG and Tenera Environmental 2022)). Several PAHs were also detected in the ITP's effluent in 2021 at varying concentrations (ERG and Tenera Environmental 2022).

As for PCB and DDT levels in fish tissue, the City of San Diego (2022a)reported that PCBs were detected in 58% of muscle tissue samples from the PLOO and 25% of samples from the SBOO rig fishing zones in 2020-2021, at concentrations ≤ 8.1 ppb. Several samples had PCB levels in exceedance of the Office of Environmental Health Hazard Assessment (OEHHA) threshold of 3.6 ppb. PCBs were also detected in all fish liver tissue samples from the PLOO trawl zones and in 71% of the samples from the SBOO trawl zones in 2020-2021, at concentrations ≤ 868.3 ppb. Historically, PCBs have been detected in 89–100% of the liver tissue samples from sanddabs, scorpionfish, and hornyhead turbot analyzed since 1995, with total PCB concentrations generally within ranges reported elsewhere in the SCB (e.g., Mearns 1988; LACSD 2020; City of San Diego 2021b). The City of San Diego (2022) also reported that DDTs were detected in all muscle tissue and liver samples collected in the vicinity of the PLOO and in 75% of muscle tissue and 96% of liver samples collected in the SBOO region in 2020-2021, at concentrations ≤ 15.4 ppb in muscle tissues and ≤751.3 ppb in liver tissues. In 2020-2021, PAHs were detected in 8% of the muscle tissue samples from PLOO and SBOO rig fishing zones at concentrations ≤ 299 ppb, but were not detected in any liver tissue samples from the PLOO or SBOO trawl zones (City of San Diego 2022a). The City of San Diego (2022) reports that concentrations of PCBs, DDTs, and PAHs have been widely variable over time with most being detected at levels within ranges reported elsewhere in the SCB. Overall, levels of PCBs, DDTs, and PAHs near the PLOO and SBOO are not higher than at reference sites, and there are no discernable patterns associated with proximity to the PLOO or SBOO (City of San Diego 2022a).

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. Preliminary Bight 2018 data indicate non-detectable to low levels of PBDEs in coastal sediments (EPA 2021). However, due to restrictions on the use of brominated forms of flame retardant such as PBDEs, increased use of alternative flame retardants, such as chlorinated organophosphate esters, has most likely occurred to meet flammability standards for many consumer products, such as mattress pads, furniture, or automobile seating (EPA 2015).

Organophosphate esters is one class of the flame retardants increasingly used as manufacturing additives not only as flame retardants, but also as plasticizers, antifoaming agents, in 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 are not available for the ITP's discharge effluent. Limited data on TCEP and TCPP are available for the Point Loma WWTP. Data from a single study show that TCPP was not detected but TCEP was detected at a concentration of 0.16 micrograms per liter in Point Loma WWTP's effluent (Vidal-Dorsch et al. 2014). Additional information is available for other areas, including north along the coast in Santa Monica Bay. In 2013, PBDEs, TCEP, TCPP, and TDCPP were consistently detected in the effluent for the City of Los Angeles' Hyperion Treatment Plant (Hyperion; EPA 2017a). In a special study conducted in 2019, PBDEs were detected at low levels in Hyperion's effluent, whereas TCEP, TCPP, and TDCPP were detected in all effluent samples for both Hyperion and the Terminal Island Water Reclamation Plant (TIWRP; LASAN 2020). Average mass loadings were the highest for TCPP in Hyperion's effluent (approximately five pounds per day; LASAN 2020). The results were consistent with those of Vidal-Dorsch et al. (2012), who also detected TCPP in all effluent samples from four publicly owned southern California treatment works. Further up the coast in San Francisco Bay, a 2018 monitoring study also detected organophosphate flame retardants (including TCPP) at concentrations comparable to or greater than PBDEs (Lin and Sutton 2018).

Tributyltin (TBT) is a persistent pollutant that has been used as an antifoulant on ships, buoys, nets and piers to restrict or retard growth of fouling organisms. Although it may pose a toxic threat to species, bioaccumulation appears to be less than for other persistent pollutants (e.g., PCBs, DDTs, and PBDEs). Butyltins have been measured in nearshore and in the deeper basins in southern California (Venkatesan et al. 1998). TBT was detected in the ITP's discharge effluent in 2021, with average monthly levels of 1.4 µg/L (ERG and Tenera Environmental 2022). TBT and other butyltins were not detected in 2017 effluent monitoring for Point Loma WWTP (City of San Diego 2019). The NPDES permits authorizing discharge of effluent from the Point Loma WWTP and ITP include performance goals and mass emission benchmarks, as well as monthly influent and effluent monitoring for TBT.

Discharge of untreated wastewater from SAB Creek and other transboundary flows may also result in loading of POPs into the action area. The levels of POPs from these sources is not known, but is likely to be higher than the concentrations discharged via the SBOO and PLOO, which discharge treated wastewater that has undergone processes to remove POPs. Monitoring is needed to assess the levels of POPs from transboundary flows.

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 as well as discharge of untreated wastewater from transboundary flows can be major sources 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.
- Industrial/commercial compounds like benzophenone, bisphenol A, and nonylphenol; and
- Microplastics such as microbeads.

Only limited ITP effluent monitoring data are available for these CECs. In 2021, average monthly concentrations for POPs and endocrine-disrupting chemicals were non-detectable or well below the 30-day average limits. Monitoring was not conducted for PPCPs, veterinary medicines, industrial EDCs, nanomaterials, industrial/commercial compounds, or microplastics.

The Point Loma WWTP's effluent discharged through the PLOO has not been routinely monitored for CECs, except for those that are included in the facility's broad category of priority pollutants. Effluent monitoring shows that CECs have been detected in the Point Loma discharge, including several PPCPs (e.g., ibuprofen, naproxen), pesticides (e.g., lindane, DEET), hormones (e.g., estradiol, estrone, progesterone, testosterone), and industrial/commercial compounds (e.g., TCEP, benzophenone, bisphenol A, nonylphenol; City of San Diego 2021d). Mean concentrations varied widely among individual CECs, ranging from not detected to 99 micrograms per liter for PPCPs, from not detected to 0.51 micrograms per liter for pesticides, from not detected to 0.18 micrograms for hormones, and from not detected to 0.9 micrograms per liter for industrial/commercial compounds (City of San Diego 2021d).

The City of San Diego has conducted special studies to evaluate the effects of several of these CECs in the effluent on fish species (City of San Diego 2021d). Levels of PBDE flame retardants in the ITP and PLOO effluent were not evaluated; however, the City of San Diego (2021d) notes that PBDEs appear to be declining in the environment since being banned from use and WWTPs are no longer considered to be major sources of PBDEs (Dodder et al. 2012). This is consistent with the 2018 Bight Regional Monitoring results that show non detect or low levels of PBDEs in coastal sediment samples (EPA 2021).

Discharge of untreated wastewater from SAB Creek and other transboundary flows may also result in loading of CECs into the action area; however, levels of CECs from these sources is not known. In addition, advanced primary and secondary treatment may not be effective at removing some CECs from the wastewater. Monitoring is needed to assess the levels of CECs from transboundary flows.

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

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

2.4.1.1.4 Harmful Algal Blooms (HABs)

In the SCB, eastern boundary currents drive seasonal upwelling that brings significant amounts of nutrients into shallower coastal waters. In the SCB, there is also a continuous source of nutrients discharged from several WWTPs through the PLOO and the SBOO. Some of the WWTPs in this area include the ITP, the SBWRP, the Point Loma WWTP, and the SABTP (Howard et al. 2014). Currently, the ITP and the SBWRP are permitted to discharge 40 MGD of treated effluent via the SBOO, but actual discharges are less (~30 MGD) as most of the SBWRP flows are reclaimed (ERG and Tenera Environmental 2022). The volume from the SABTP facility is estimated at 35 MGD with nearly 30 MGD of that being untreated sewage and Tijuana River water. The Point Loma WWTP is permitted to discharge up to 240 MGD of advanced primary treated effluent. In recent years, the average daily discharge has been approximately 140 MGD; this is expected to decrease due to additional water reclamation and recycling over the next few years. In addition to effluent discharged from the WWTPs, transboundary flows of untreated wastewater from Mexico also contribute an unknown but potentially substantial level of nutrients to the action area.

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

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

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

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

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

The Howard et al. (2014) study does not match up perfectly with the action area for this consultation, however, and has important uncertainty in estimating the contribution of nitrogen from upwelling. Howard et al. (2014) analyzed four WWTPs that discharge into the SCB, including Point Loma and three smaller WWTPs (Hale Avenue Resource Recovery Facility, SBWRP, and the ITP). The Hale Avenue Resource Recovery facility (~18 MGD/day discharge) is in Escondido, California and discharges out of the action area through a shared outfall in Cardiff, California to the north of the action area. Howard et al. (2014) did not include information on discharges from the SABTP (~35.5 MGD/day discharge) in Tijuana, Mexico, which discharges largely untreated wastewater into the action area resulting in high nutrient loading to the nearshore area.

Howard et al. (2014) also noted that the San Diego region was at the edge of the Regional Oceanic Modeling Sysems (ROMS) model boundary, which presents a large amount of uncertainty. In particular, the total contribution of nitrogen from upwelling was likely underestimated in the San Diego region. The wastewater contribution in the study is estimated to be ~3 times higher than the estimated upwelling contribution (7,400 Kg N/km^2/year v. 2,400 Kg N/km^2/year) which the authors recognized as unlikely. While the information in Howard et al. (2014) is insufficient to precisely assign the nutrient contributions of the ITP as a percent of the total in the action area, now or in the future, it is evident that the facility's constant contribution of nitrogen to the receiving water is significant. Diverting the flow from the SABTP will increase the impact of the ITP's discharge offshore while simultaneously reducing the inshore impacts.

Nitrogen from upwelling is largely in the form of nitrate (98.7%) while nitrogen in effluent is largely ammonium (92%), a reduced form (Howard et al. 2012b, 2014). Concentrations of nitrogen in the effluent plumes are up to three orders of magnitude greater than maximal ambient concentrations and they also entrain deeper, nutrient laden water as they rise into or through the euphotic zone when stratification is weak (Reifel et al. 2013; Seegers et al. 2015). This surfacing most often occurs during the winter months in the SCB as a whole (Kessouri et al. 2021), but the discharge from the SBOO is within the euphotic zone all year. Stratification typically maintains the SBOO plume at a depth of only 26 ft during the warmer months of the year, but the plume surfaces during other periods of the year when thermal stratification ceases at this location (Largier et al. 2004; Terrill et al. 2009).

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 had "hot spots" of high offshore chlorophyll-*a* (CHL-*a*), which is indicative of phytoplankton production. Nezlin et al. (2012) found that these "hot spots" occurred throughout most of the year along the south San Diego Region and at the WWTP outfalls. The San Diego area has been identified as a hot spot with longer residence times and higher CHL-*a* levels (Trainer et al. 2010; Nezlin et al. 2012; Hess 2021; Smith et al. 2021; Svejkovsky and Hess 2022). Additional nutrients may enter the San

Diego region from the south due to the Southern CA Counter Current (Howard et al. 2012b; Feddersen et al. 2021; Kessouri et al. 2021) and from winter runoff.

HAB species are known to persist in the subsurface zone and then be advected into the upper water column during the upwelling season, where the combination of nutrient availability and increasing sunlight may result in a bloom (Trainer et al. 2010; Seegers et al. 2015). WWTP discharges, including discharges from ITP, 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 a year round source of nitrogenin the shallow water column in the action area. These subsurface populations of HAB species can then be uplifted into the surface waters and are a possible explanation for the occurrence of "instant" domoic acid events immediately following upwelling rather than a typical delayed bloom development (Seegers et al. 2015; Smith et al. 2018).

Monitoring of the SBOO and PLOO discharge plume shows the distribution to be primarily alongshore to both the north and south of the outfalls (Rogowski et al. 2012a; Hess 2021; Svejkovsky and Hess 2022; City of San Diego 2022a) with a periodic inshore component. Feddersen et al. (2021) shows that the shoreline discharge of the SABTP frequently moves northward into the action area in the shallow near-shore waters before mixing into the deeper waters south of the Point Loma headland where the SBOO and PLOO discharges have also been monitored. In 2021, monthly average ammonia levels in the SBOO effluent varied from 870 to 55,600 µg/L (close to the daily maximum limit of 57,400 µg/L) (ERG and Tenera Environmental 2022). Previous monitoring (for the PLOO) shows that ammonia concentrations in the receiving waters in the vicinity of the discharge after initial dilution were at or near detection levels and virtually indistinguishable from natural background levels (City of San Diego 2019). This is not surprising given that ammonia levels become more difficult to detect as it is converted to nitrate and as both forms are taken up by phytoplankton.

Both forms of nitrogen discharged by the facility, ammonium and nitrate, can support the growth of HAB species. Kudela et al. (2008) showed that ammonium uptake by *A. sanguinea*, a red tide forming dinoflagellate, was approximately threefold higher than uptake of nitrate. Kudela et al. (2010) later showed that *P. spp.* grew equally well or better on reduced nitrogen sources and Howard et al. (2007) showed that *P. australis* could use either nitrate or ammonium simultaneously, a trait that may give it a competitive advantage in areas subjected to wastewater effluent discharges and upwelling.

There are several sources which summarize numerous studies and conclude that reduced forms of nitrogen (ammonium, urea) significantly tilt the phytoplankton community toward the development of HABs (Howard et al. 2012b; Reifel et al. 2013; Booth 2015; Seegers et al. 2015). Schnetzer et al. (2007) cites several studies that examined *P-spp.* and noted that their effective toxicity can be highly variable. These diatom species seem to produce higher levels of domoic acid when under silica or phosphate stress (i.e., the N:P and/or N:Si ratios are higher than or altered from natural conditions) (Anderson et al. 2006; Schnetzer et al. 2013). The presence of large amounts of nitrogen in the discharge has the effect of unbalancing these ratios at the local level and may be partially responsible for the very potent HABs that have been occurring in the spring for ~10 years in the SCB.

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

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

As discussed in details in Section 2.5.3, we anticipate that the effluent discharged via the SBOO and PLOO and untreated wastewater discharged from the SABTP and other transboundary flows would help encourage HAB formation within the action area. Any ESA-listed marine mammals or sea turtles foraging nearby would experience an increased risk of exposure to biotoxins that may impact their health.

2.4.2. Marine Mammals and Sea Turtles

2.4.2.1 Health and Contamination

POPs can be highly lipophilic (i.e., fat soluble) and are primarily stored in the fatty tissues in marine mammals and sea turtles (O’Shea 1999; Aguilar et al. 2002). Therefore, when marine mammals consume contaminated prey, they store the contaminants primarily in their blubber; POPs are stored in the fatty tissues and plasma in sea turtles. POPs can resist metabolic degradation and can remain stored in the blubber or fatty tissues of an individual for extended periods of time. When prey is scarce and when other stressors reduce foraging efficiency, or during times of fasting, a marine mammal metabolizes their blubber lipid stores, causing the pollutants to either become mobilized to other organs or remain in the blubber and become more concentrated (Krahn et al. 2002). Adult females can also transmit large quantities of POPs to their offspring, particularly during lactation in marine mammals. Mature female sea turtles offload their burdens to their eggs and hatchlings (Van de Merwe et al. 2010; Stewart et al. 2011). The mobilized pollutants can then become bioavailable and may cause adverse health effects. As described above, metals and CECs have widespread occurrence, and some have the potential for biomagnification, and biological activity. However, we have little data on concentration levels in ESA-listed species or their prey of these contaminants. Below, we provide a summary of what information is available on POP levels in ESA-listed marine mammals and sea turtles.

2.4.2.1.1 Marine Mammals

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

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

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

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

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

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

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

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

More recently, POP levels were reported in baleen whales from their earplugs (Winfield et al. 2020). Earplugs represent a unique opportunity to examine the spatiotemporal trends of POPs in the marine ecosystem, so analytical techniques capable of reconstructing lifetime (i.e., birth to death) chemical exposure profiles in baleen whale earplugs are being developed. Earplugs from six baleen whales (one blue whale and five fin whales) were collected, and DDT and PCBs were the most dominant POPs (spanning the past 80 years). Lifetime bioaccumulation rates were 56 times higher in the North Pacific as compared to the North Atlantic suggesting that baleen whales fom the North Pacific may be exposed to increased levels of POPs. However, caution is warranted in interpreting these results because only a small sample size (n=6) was collected and analyzed.

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 the coasts of Japan and USA are similar. Transplacental transfer of TBT from mother to fetus is relatively low compared to other persistent pollutants. For example, TBT concentrations in the liver of a pregnant female killer whale (150 nanogram per gram wet) was much higher compared to concentrations in the liver of the fetus (26 nanogram per gram wet) (Tanabe et al. 1998). TBTs do not appear to differ between males and females, however, increasing levels have been observed in immature stages of Risso's dolphins (Tanabe 1999).

2.4.2.1.2 Sea Turtles

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

These relatively higher levels of POPs off California are likely the result of higher concentrations in sediment and biota in the region. Different chemical signatures between populations or groups within a population highlight the influence of foraging locations on exposure. In loggerhead sea turtles off Florida, POP profiles in the blood plasma revealed some loggerheads migrate up and down along the coast, whereas others remain resident (Ragland et al. 2011). Komoroske et al. (2011) and Barraza et al. (2020) also suggests the higher concentrations in Southern California green sea turtles may also be attributed to potential increased foraging rates as a result from elevated temperatures from power plant discharge, ultimately increasing consumption rates and thus elevated exposure risk. Furthermore, the high DDE levels in San Diego green sea turtles may suggest immunological effects because the levels exceeded lymphocyte proliferation no-effect levels established for loggerheads (Keller et al. 2006). Little is known about contaminant levels in olive ridley and leatherback sea turtles.

2.4.2.2 Strandings

Strandings of ESA-listed marine mammals and turtles have been documented within and vicinity of the action area (NMFS WCR unpublished stranding data). From 2016-2021, the following total number of strandings were documented within the action area (and immediate surrounding area) for each ESA-listed marine mammal species: six humpback whale (it is unknown if these humpback whales belongs to the Mexico or the Central America DPS), three fin whales, and eight Guadalupe fur seals. The cause of many of these strandings is unknown. Three humpback whales were entangled, and one humpback whale appeared to strand due to a vessel strike. Two of the three fin whales also appeared to strand due to a vessel strike, and five of the eight Guadalupe fur seals appeared to be suffering from illness and/or malnutrition.

From 2016-2021, the following total number of strandings were documented within the action area (and immediate surrounding area) for each ESA-listed sea turtle species: 60 green turtles, three loggerheads, two olive ridley, and three unidentified sea turtles.[2] Of the 60 green turtles, 45 and nine turtles were documented in San Diego Bay and Mission Bay, respectively, both of which are located within the action area. Twenty-one of the green turtles were struck by vessels, and six interacted with recreational and/or commercial fishing gear, including one entangled in anchor line and another one that had a monofilament line protruding from the turtle's mouth that continued into and out of their gastrointestinal (GI) system. One loggerhead appeared to have been struck by a vessel. The cause of strandings of the remaining sea turtles is unknown.

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

2.4.3. Abalone

2.4.3.1 Monitoring of Wild Populations and Outplanting Efforts

Comprehensive surveys for white abalone have not yet been conducted throughout the coast off San Diego, though most survey efforts have concentrated on the area off La Jolla and Point Loma in recent years, with several white abalone in these areas since 2010 (Neuman et al. 2015). In 2017, researchers collected four of these white abalone individuals to serve as broodstock for the white abalone captive breeding program, under the Scientific Research and Enhancement Permit 14344-2R issued by NMFS to the Bodega Marine Laboratory (BML) under ESA Section 10(a)(1)(A). These animals were deemed as "singletons" with a low likelihood of reproducing in the wild given their distance (>10 m) from other white abalone (NMFS 2016). Researchers collected these individuals to serve as broodstock in the captive breeding program, to enhance the genetic diversity and productivity of the captive breeding program and to advance species recovery efforts.

Since 2019, NMFS has worked with partners to outplanted more than 4,200 captive-bred white abalone at two sites, including a site off San Diego, as part of experimental outplanting studies (NMFS 2021a). Post-outplant monitoring via dive surveys as well as time-lapse cameras are being used to assess the survival and behavior of the outplanted abalone and the presence of wild adult white abalone.

2.4.3.2 Impacts of Harvest and Other Factors

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

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

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

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

2.4.4. Other Federal Actions

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

- Reissuance of an NPDES permit for the Point Loma Wastewater Treatment Plant and Ocean Outfall for continued discharge of advanced primary treated effluent into the Pacific Ocean (NMFS 2022). Discharges conducted under this NPDES permit are expected to affect water quality and exposure of ESA-listed marine mammals, sea turtles, fish, and abalone species to contaminants, which could result in non-lethal harm or injury. We discuss some of these effects in the sections above.
- Monitoring and outplanting of white abalone at subtidal reefs within the action area, to enhance wild populations to self-sustaining levels (NMFS 2019c).
- Fisheries research surveys conducted by the SWFSC throughout the North American west coast, typically from ship-based platforms, with potential lethal and sublethal effects on several ESA-listed sea turtle and fish species from capture or entanglement (NMFS 2020b).
- The Navy's military readiness training and testing (including research, development, testing, and evaluation) activities in the Hawaii-Southern California Training and Testing Study Area, which includes the action area (NMFS 2018a). Training and testing activities include the use of active sonar and explosives that can adversely affect several ESA-listed marine mammals and sea turtles. Effects include harassment and harm from impulsive and non-impulsive acoustic stressors, as well as mortality and non-lethal injuries resulting from vessel strikes.
- Uniform National Discharge Standards for vessels of the Armed Forces operating in U.S. waters (NMFS 2019d). Discharges conducted under these Standards are expected to affect water quality and exposure of ESA-listed marine mammals, sea turtles, and fish species to contaminants, which could result in non-lethal harm or injury.

2.5. Effects of the Action

Under the ESA, "effects of the action" are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action (see 50 CFR 402.02). A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur.

Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action (see 50 CFR 402.17). In our analysis, which describes the effects of the proposed action, we considered the factors set forth in 50 CFR 402.17(a) and (b).

For the Effects Analysis, we have identified the following potential effects associated with the future discharge of wastewater by ITP and APTP through the SBOO:

- 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.

EPA and USIBWC anticipate that the proposed action (consisting of four Core Projects) will reduce the transboundary flow of untreated wastewater from Mexico into U.S. waters off San Diego County, and the overall discharge flow amount and rate of treated wastewater will increase following completion of ITP expansion and building of APTP. Thus, this consultation focuses on the impacts to ESA-listed species that will result from the anticipated overall discharge of effluent by the ITP and APTP through the SBOO in the future as a consequence of the proposed action. Specifically, we will evaluate the anticipated effects associated with the capacities that are being created to accommodate wastewater treatment and discharge needs out to 2050. We also identified that activities related to expanding ITP and construction/operation of APTP may have effects to ESA-listed species.

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 the proposed discharge of effluent by the ITP and APTP through the SBOO include: (1) uptake of pollutants from the water; (2) ingestion of prey that have accumulated pollutants; and (3) exposure to harmful algal blooms resulting from the discharge effects. Then, we evaluate how ESA-listed species may respond to this exposure and how their responses may reduce the fitness of individuals of the affected populations. If a potential reduction in individual fitness is expected, then we consider how the effects on individual reproductive development, growth, and survival may affect the population's growth, reproductive potential, and survival. We also evaluate how these effects may affect the population's recovery potential considering the importance of the population to the species' survival and recovery, as appropriate.

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

2.5.1. Effects from Recommissioning SBOO to Accommodate Expanded ITP and Future APTP Operation

As described in Section 1.3 Proposed Action, up to 55 diffuser risers on the south leg may be recommissioned (opened) in the near future (by no later than 2027) to accommodate the increase in effluent discharge with the proposed expansion of the ITP, and construction and operation of the APTP. As part of construction activities to recommission or open the diffuser risers, divers will remove flanged covers on risers and replace these with port units. Vessels will be used to transport the divers and equipment to the site.

If anchoring is required (anticipated needing one or two), the vessel will deploy anchors on sandy habitat to avoid damaging the wye diffuser and associated structures, as well as any white abalone habitat. It is likely that the anchor lines will remain under tension, but details ultimately will depend on configuration and operation choices of the specialized recommissioning contractor. Alternatively, a permanent mooring may be used. Although having lines in water can pose an entanglement risk to marine species that have the possibility of encountering those lines, high tensioned lines and anchors would likely help animals bounce off the lines and prevent them from getting entangled and/or decrease the chance of them becoming seriously injured from an encounter. Thus, we conclude that the entanglement risk to ESA-listed species is very low.

To minimize potential risk for ship strikes, at least one crew, most likely the vessel operator, maintains a constant watch of the ocean surface in front and adjacent to the vessel at all times. If marine mammals and sea turtles are observed distant to the vessel, vessel operators will adjust their course as necessary to ensure they do not disturb the natural behavior of these animals. If marine mammals are in close proximity, they will:

- Slow down and operate at a no-wake speed.
- Stay out of the path of the animal's direction of travel.
- Not put their vessel between whales, especially mothers and calves
- Not chase or harass animals, and will not approach the animals head-on, from directly behind them, or from the side. If animals are following a trajectory closely parallel to the direction of vessel travel, they will gradually steer the vessel to be parallel to the animals from the side and stay at least 100 yards away (i.e., the length of a football field).

Due to the above mitigations measures, the likelihood of an ESA-listed species to be struck by vessels is extremely low.

During removal of flanged covers on risers by divers, some noise will be produced. However, only relatively low-noise producing methods (e.g., hand tools) will be used avoiding the use of noisy activities like cutting and hammering. EPA and USIBWC determined that mobile animals (e.g., marine mammals, turtles, and fish) may occur in the action area during these activities, and these animals may be disturbed by the noise causing them to leave the immediate area and/or change their natural behavior like foraging. However, these animals are unlikely to move far relative to their typical home range or short-term foraging range as the diver activities and associated disturbance will remain restricted to a small area around the riser. Additionally, the activity will occur over a relatively short period of time (a few hours each day for a few weeks) within a small area, and it is likely to occur in phases over the course of several years. Although there may be minor effects to these animals from disturbance as a result of noise associated with the proposed action, the anticipated effect is temporary.

When removing flanged covers on risers and replacing them with port units, divers will likely remove a small area (up to a 6-ft by 6-ft area) of habitat and species on and around the diffuser heads that require modification. This includes the removal of rock armoring (small to medium boulders placed to protect the SBOO structure) and algae and would result in a reduction in artificial reef habitat for any white abalone that may already be on the SBOO wye structure. We expect this loss in habitat to be small compared to the overall artificial reef habitat provided by the SBOO structure. We also expect the loss of habitat to be temporary as natural ecological succession processes gradually replace the lost habitat and species over time.

There is the potential for white abalone individuals to occur within the habitat to be removed. We estimate that divers could encounter and remove up to two white abalone as part of the proposed action based on what is known about the habitat and the presence of white abalone within the action area. The SBOO structure is surrounded by soft (sandy) substrates that are not suitable habitat for white abalone. The nearest kelp bed is about 1.5 mi (2.5 km) away off Imperial Beach. The closest known population of white abalone is off Point Loma; however, this reef is at least 6.2 mi (10 km) north along the coast from the SBOO structure (see Section 2.5.2.1.4 White Abalone), limiting the potential for dispersal of white abalone and their larvae from this population to the SBOO structure. Isolated white abalone have been found on man-made structures such as underwater power cables (e.g., a single white abalone found on a cable off Santa Barbara; NMFS 2005) and in areas surrounded by unsuitable habitats, such as on the inside of breakwaters (NMFS 2019e). Given the above information, we estimate up to two white abalone to be present on the SBOO structure within the area that is anticipated to be affected by the proposed action. We expect any white abalone removed as part of these activities to be injured and killed, unless they are removed by experienced personnel and relocated to suitable habitat nearby (and monitored for a period of time to ensure survival) or transported to an approved captive holding facility (as permitted under Scientific Research and Enhancement Permit 14344-2R issued to the UC Davis-Bodega Marine Laboratory under Section 10(a)(1)(A) of the ESA).

Vessel activities bring a small risk of grounding or oil spill. Vessels are likely to carry hydraulic fluids and fuel that would be toxic to marine life if spilled. Most marine vessel groundings or spills are the result of mechanical failures or pilot negligence. However, it is assumed that construction vessel operators will follow best practice by maintaining their vessels to a high standard. Furthermore, vessel operators will maintain industry standard health, safety, and environmental standards that apply specifically to the intended construction operations. This is likely to include the storage and maintenance of spill kits appropriate for dealing with small vessel-based spills such as sand buckets, absorbent pads and cloths, and other emergency containment devices to stop small spills of hydraulic fluids and other polluting fluids from entering the water if they are accidentally spilled on deck. Vessels must be maintained to a standard that eliminates the likelihood of diesel or hydraulic oil spills during normal operation. In the case of a catastrophic loss of engine power that may result in a grounding, vessel captains must have procedures in place to raise Coast Guard support rapidly. Thus, having these measures in place, the risk of vessel activities impacting ESA-listed species is low.

In summary, we expect the recommissioning activities associated with expanding the ITP and the construction and operation of the APTP pose a low risk to ESA-listed marine mammals and sea turtles. We expect construction activities to result in a temporary loss of artificial reef habitat for white abalone on the SBOO wye diffuser. If white abalone are present within the habitat to be removed, we expect removal activities to injure and kill the white abalone unless conducted by experienced personnel and the abalone are relocated to nearby suitable habitat or transported to an approved captive holding facility. We estimate that up to two white abalone could be adversely affected by the proposed project.

2.5.2. Exposure and Response to the Toxicity of the Proposed Discharge of Effluent by the ITP and APTP through the SBOO

2.5.2.1 Species Occurrence and Exposure

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

2.5.2.1.1 Marine Mammals

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

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

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

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

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

2.5.2.1.2 Sea Turtles

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

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

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

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

As described above, the discharge plume extends throughout the action area, with greater plume probabilities in the vicinity of the outfall (see Figure 2; Figure 3). Based on seasonal plume detections (see Figure 2; Figure 3) and modeled plume probabilities (ERG and Tenera

Environmental 2022), we expect that all ESA-listed marine mammal and sea turtle species considered in this biological opinion have the potential to occur and overlap with the effluent discharge essentially anywhere throughout the action area, as all these species are highly mobile and could occur anywhere in the area.

2.5.2.1.4 White Abalone

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

Where monitoring data is limited, we use habitat as a proxy for the likely presence of white abalone. White abalone adults occupy open, low relief rocky reefs or boulder habitat surrounded by sand, within depths of 5 to 60 meters (Hobday and Tegner 2000). Most of the subtidal rocky reef habitat exists in nearshore areas off La Jolla, Point Loma, and Imperial Beach (Figure 4). Off Imperial Beach, white abalone presence has not been confirmed, though an area of rocky reef appears to occur off Imperial Beach (indicated by kelp persistence; Figure 7). EPA and USIBWC characterize this area as a seabed with predominantly cobble habitat and that may be intermittently covered by sand (ERG and Tenera Environmental 2022). Based on this, white abalone have a lower likelihood of occurring within the rocky reef off Imperial Beach. However, the ballast rock structure protecting the emergent portions of the SBOO may provide suitable habitat for white abalone.

Overall, white abalone are confirmed to occur on rocky reefs off La Jolla and Point Loma. and may be present in rocky reefs off Imperial Beach as well as on the SBOO structure itself.

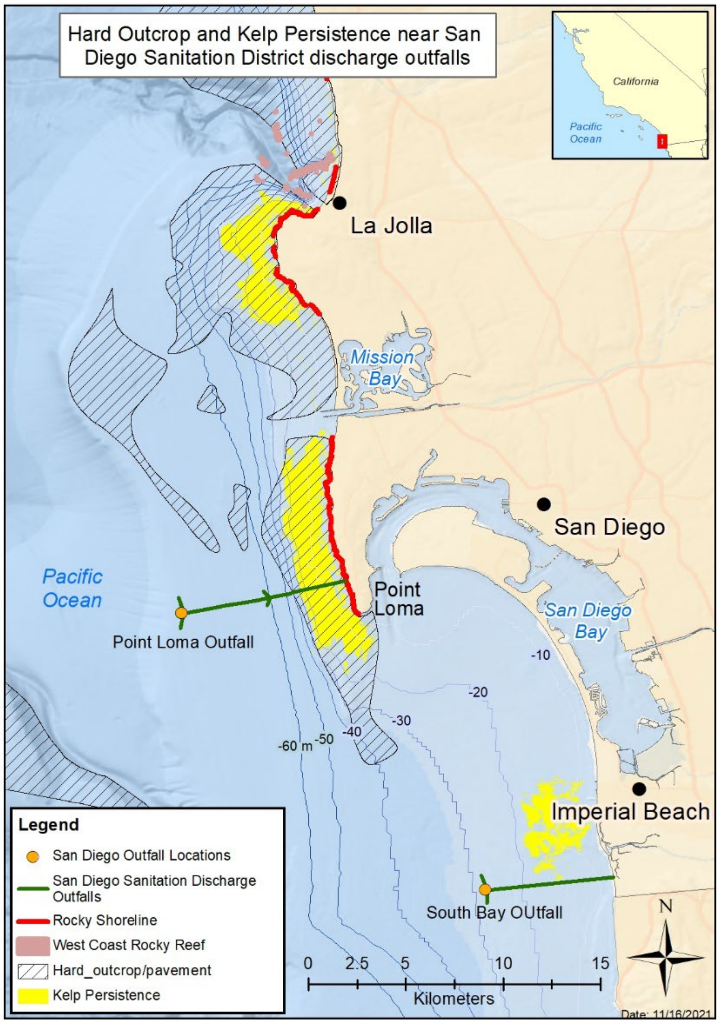

Figure 4. Colored map showing potential white abalone habitat within the action area, primarily consisting of subtidal rocky reefs in nearshore waters (within 20-40 m depth) off La Jolla, the Point Loma peninsula, and Imperial Beach.

2.5.2.1.5 Overlap with effects of the discharge – white abalone

As described above, the discharge plume extends throughout the action area, with greater plume probabilities in the vicinity of the outfall and both upcoast/downcoast and lower plume probabilities inshore of the 20 meter contour (see Figures 2 and 3). Based on seasonal plume detections (see Figures 2 and 3) and modeled plume probabilities (ERG and Tenera Environmental 2022), we expect white abalone on rocky reefs off Imperial Beach, Point Loma, and La Jolla to be exposed to low plume concentrations based on the location of suitable rocky reef habitat.

Any white abalone occurring on the SBOO structure would be exposed to greater effluent concentrations, with the greatest exposure within the ZID. Surveys conducted to date have not observed any white abalone on the outfall structure (ERG and Tenera Environmental 2022). The likelihood that white abalone occur on the SBOO structure is low given that the emergent portions of the SBOO are surrounded by soft bottom habitat and are at least 2.5 km from potential white abalone habitat. The likelihood of white abalone occurring within the ZID is even lower, because the high effluent concentrations within the ZID potentially preclude abalone survival and presence. We note that in 2016, NMFS issued Scientific Research and Enhancement Permit No. 14344-2R to BML at UC Davis (under Section 10(a)(1)(A) of the ESA), allowing collection of white abalone from the wild to serve as broodstock for the captive breeding program. White abalone found on the outfall structure may be collected by researchers under this Permit, if they meet the Permit's collection criteria.

In summary, based on the best available information regarding white abalone presence and the extent of the discharge plume and ZID, we conclude that: (a) white abalone may occur on the SBOO structure (though the linkelihood is low) and may be exposed to the plume and its effects, with greater levels of exposure in areas within and adjacent to the ZID; (b) white abalone habitat (and any individuals occurring in that habitat) may also be exposed to the discharge plume and its effects; and (c) because of the sedentary life history of abalone, the risks of exposure are persistent over time.

2.5.2.2 Constituents of the Proposed Discharge of Effluent by ITP and APTP through the SBOO

Typical effluent limits in the NPDES permits for WWTPs include TSS, BOD, chlorine, oil and grease, setteable solids, chronic toxicity, pHs, turbidity, and aldrin. Additional elements and compounds such as metals are also discharged. Other constituents of the discharge that are typically monitored include metals (arsenic, mercury, and cadmium), PAHs, PCBs, vinyl chloride, endosulfan, and nitrogen (ammonia), and hexachlorocyclohexane. Other constituents of the discharge that are suspected or known to be present in wastewater discharge include: PBDEs, organophosphate esters, TBT; PPCPs including prescribed and over-the-counter medications and numerous other products; and EDCs, including estrogen, androgens, and pesticides. In this Effects Analysis, we consider the potential effect of these pollutants on ESA-listed species.

2.5.2.3 Response of Marine Mammals and Sea Turtles to Exposure to the Plume from the Proposed Discharge of Effluent by ITP and APTP through the SBOO

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 this proposed action does not appear to pose a threat to larger vertebrates that breathe air and have integumentary systems that limit direct uptake from the environment. Within the ZID, marine mammals and sea turtles could be exposed to relatively higher concentrations of various effluent constituents that are potentially toxic. We cannot precisely estimate exposure times within the ZID for ESA-listed marine mammals and sea turtles given their dynamic movements and occasional occurrence in the action area, but we anticipate exposure to concentrated effluent in the ZID will be relatively minimal.

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

The available data indicate that ESA-listed marine mammals and sea turtles are generally not at risk of health effects from most of the compounds or elements (typically metals) measured in the proposed discharge of effluent by ITP and APTP through the SBOO. 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 the proposed discharge of effluent by ITP and APTP through the SBOO 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 this proposed action'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.

2.5.2.4 Response of White Abalone to Exposure to the Plume from the Proposed Discharge of Effluent by ITP and APTP through the SBOO

We evaluated how different life stages of white abalone may be affected by exposure to the discharge effluent. Because species-specific information is not available, we used the best available information from studies involving other abalone species. We note that Section 2.5.3 (Accumulation of Potentially Harmful Contaminants) addresses the potential effects associated with other more persistent and/or harmful constituents that may accumulate.

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

For juvenile and adult white abalone, several studies have evaluated the effects of water borne and/or dietary exposure of juveniles and adults to copper, zinc, silver, and cadmium. They found that abalone can accumulate these metals in their foot muscle, mantle tissues, and viscera, and experience adverse effects on growth, behavior, and survival (Martin et al. 1977; Liao et al. 2002; Gorski 2006; Huang et al. 2008, 2010; Chen et al. 2011). Martin et al. (1977) exposed adult abalone to concentrations of copper ranging from 10 to 640 ug/L and found 96-hour LD50 values of 65 µg/L for *H. rufescens* and 50 µg/L for *H. cracherodii*, compared to 87 µg/L for *H. rubra* (Gorski 2006). Viant et al. (2002) suggested asphyxial hypoxia and reduced muscle function as possible mechanisms for mortality due to copper exposure.

Acute (7-day) and chronic (28-day) toxicity studies exposing *H. diversicolor supertexta* to zinc found reduced growth rates of individuals at 120-125 ug/L and increased mortality at 500-1000 ug/L (Liao et al. 2002; Chen and Liao 2004; Tsai et al. 2004). Chen et al. (2011) modeled the effects of exposure to cadmium and silver on growth and predicted that growth of larvae, juveniles, and adults was inhibited by exposure to levels as low as 10 ug/L Cd and 5 ug/L Ag. In studies exposing *H. diversicolor supertexta* to water borne and dietary silver (5 or 50 ug/L) or cadmium (50 or 500 ug/L), Huang et al. (2010) observed reduced growth and feeding rates in the first few weeks, but similar rates to controls by the end of the 7-week exposure period.

To apply these study results to the the discharge of proposed discharge of effluent by ITP and APTP through the SBOO, we considered the concentrations of these pollutants in the effluent and the concentrations to which white abalone may be exposed. In 2021, varying concentrations of cadmium, copper, lead, mercury, silver, and zinc were detected in the ITP's effluent (ERG and Tenera Environmental 2022). Listed below are the range of monthly average concentrations and monthly maximum concentrations in the ITP's effluent for 2021. These represent the concentrations that abalone may be exposed to within the ZID. In parentheses, we include EPA and USIBWC's estimated changes to the concentrations discharged by ITP and APTP through the SBOO as a result of full implementation of the proposed action (i.e., following ITP expansion and construction of APTP).

- Cadmium: average from 1.9 to 2.28 µg/L: maximum of 2.6 µg/L (estimated decrease by 5-17 percent);
- Copper: average from 11 to 18.8 µg/L; maximum of 50 µg/L (estimated increase by 29-45 percent);
- Lead: average from 2 to 4 µg/L; maximum of 12 µg/L (estimated increase by 49-62 percent);
- Mercury: average from 0.05 to 0.12 µg/L; maximum of 0.12 µg/L (estimated increase by 16-83 percent);
- Silver: average from 3 to 44 µg/L; maximum of 44 µg/L (no estimates provided);
- Zinc: average from 5 to 49.25 µg/L; maximum of 110 µg/L (estimated to be between a 3 percent decrease and 2 percent increase).

For larval abalone, the recorded concentrations of copper and zinc in the effluent exceeded those found to cause increased developmental and morphological abnormalities (Conroy et al. 1996; Gorski and Nugegoda 2006). However, the studies by Conroy et al. (1996) and Gorski and Nugegoda (2006) used an exposure time of 48 hours. Given the limited extent of the ZID and the planktonic nature of abalone larvae, we expect abalone larvae to spend much less than 48 hours within the ZID. We cannot estimate the exposure time, but it would likely be short, given the small size of the ZID and the movement of water currents. There is the potential for shorter exposure times to cause adverse effects; however, we do not have information to evaluate the effects of exposure times shorter than 48 hours.

For juvenile and adult abalone, the recorded concentrations of copper and silver in the effluent exceeded those found to cause mortality and reduced growth and feeding rates (50 µg/L for copper and 5 µg/L for silver (Martin et al. 1977; Huang et al. 2010; Chen et al. 2011). Based on this, individual white abalone occurring in the ZID (i.e., on the SBOO structure) may be exposed to levels of these metals that inhibit growth, feeding, and survival.

Outside of the ZID, we expect dilution to reduce the concentrations of these metals in the plume to well below those documented to kill or cause sublethal effects on abalone at all life stages. Based on this, we do not expect exposure to these metals in the plume to negatively affect the development, growth, behavior, or survival of white abalone in the action area outside of the ZID. Results of chronic toxicity tests using red abalone larvae confirm this. In 2021, the ITP's effluent did not exceed the chronic effluent limits, meaning that there was no observable effect of the discharge on red abalone larvae and kelp germination (compared to the control) at the concentrations that larvae and kelp would be exposed to outside of the ZID (ERG and Tenera Environmental 2022). These results indicate that exposure to effluent concentrations found in the plume (outside of the ZID) would not be expected to reduce the survival and development of white abalone larvae.

In summary, the types of pollutants found within the discharge effluent can adversely affect white abalone at all life stages. White abalone occurring within the ZID may be exposed to high concentrations of pollutants that can cause developmental abnormalities, reduced growth and feeding, and mortality. For example, the reported concentrations of copper, silver, and zinc in the effluent in 2021 exceeded the levels found to affect development, behavior, and survival of larval, juvenile, and adult white abalone. For larvae passing through the ZID, we expect effects on development to be low given the likely short duration of exposure. However, larvae that settle on the outfall structure within the ZID would likely not survive or would experience developmental abnormalities. We also expect any juvenile and adult white abalone occurring on the outfall structure within the ZID to experience reduced growth, feeding, and survival due to exposure to pollutants in the effluent. We expect the presence of juveniles and adults on the SBOO structure to be very low, given the low likelihood that larvae can settle and survive and the distance of the structure from suitable rocky reef habitat. Outside of the ZID, we expect pollutant concentrations in the effluent plume to be lower than the values found to cause adverse effects on white abalone.

2.5.3. Accumulation of Potentially Harmful Contaminants

2.5.3.1 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; Mori et al. 2006); endocrine disruption (Darnerud 2003, 2008; Legler and Brouwer 2003; Legler 2008); neurotoxicity (Darnerud 2003, 2008; Viberg et al. 2003, 2006); and cancer in humans and wildlife (Ylitalo et al. 2005; 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 California waters and marine waters around the world (Lin and Sutton 2018).

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

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

Vidal-Dorsch et al (2014) measured CECs from Point Loma's discharge via the PLOO including flame retardants. PBDEs are being phased out and replaced by the next generation of flame retardants (i.e., chlorinated organophosphates; Lin and Sutton 2018), which may have similar toxic effects on ESA-listed species. Moreover, Bekele et al (2019) demonstrated both bioaccumulation and biomagnification of organophosphate flame retardants in the marine food webs. Of particular concern are the following three chlorinated phosphate ester compounds being assessed by EPA (2015) for risks to aquatic organisms and human health: TCEP, TCPP, and TDCPP. ESA-listed species may receive the majority of these pollutants from their diet. To estimate the potential for the proposed action to expose ESA-listed species to these pollutants, it is important to monitor the concentrations of these flame retardants in the effluent and understand how they move through the food web.

To understand and isolate the effects of organophosphate flame retardant loading from the proposed discharge of effluent by ITP and APTP through the SBOO, we estimated the amount of organophosphate flame retardant that is expected to be discharged annually. To estimate the mass loading of the overall organophosphate flame retardants (i.e., combination of numerous organophosphate compounds), we used data from a recent special study at Hyperion and the TIWRP that monitored for these organophosphate flame retardants in the effluent. One of the highest detections of organophophate flame retardants from Hyperion effluent was from a wet event sample. A total concentration of 3.22 mg/L of organophosphate (0.1 mg/L of TCPP and 3.12 of TCEP) was detected during a period where the average discharge of Hyperion was approximately 230 MGD (LASAN 2020).

The average flow of ITP following ITP expansion (projected to be completed by 2027) is 60 MGD. The projected average flow rate following completion of ITP expansion and construction of APTP is 76.4 MGD. Because Hyperion and this proposed action discharging effluent by ITP and APTP through the SBOO represent relatively large facilities handling domestic, commercial, and industrial wastewater from adjacent highly urbanized areas, we assume that organophosphate flame retardant concentrations and loadings from the proposed effluent by ITP and APTP through the SBOO are likely to be similar to what has been measured at Hyperion. Therefore, we use the highest detection of organophophate flame retardants from Hyperion effluent (3.22 mg/L for 230 MGD) as a surrogate for estimating the loading of the organophosphate flame retardants in this prosposed action using two flow rates: 1) ITP's average flow rate of 60 MGD following ITP expansion; and 2) projected flow rate of 76.4 MGD upon completion of both ITP expansion and APTP construction. As a result, we estimate approximately 0.7 metric tons per day of organophosphate flame retardants (TCPP and TCEP combined) may be loaded into the action area as a results of ITP expansion. This equates to 267 metric tons of organophosphate flame retardant discharged per year. Following both ITP expansion and APTP construction, we estimate approximately 0.9 metric tons per day of organophosphate flame retardants (TCPP and TCEP combined) may be loaded into the action area. This equates to an estimated 340 metric tons of organophosphate flame retardants discharged per year.

This estimate of organophosphate loading from the proposed action will add to the long-term accumulation of POPs in the action area that has already occurred from historical discharges under past WWTP operations and other sources. This accumulation through other sources will also continue to occur simultaneously with the proposed action moving into the future.

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

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

Recently, researchers from Washington Department of Fish and Wildlife have been tracking the movement of PCBs and other toxic chemicals in Puget Sound, WA, and found comparatively lower levels of these POPs in the sediment, but higher levels in the resident pelagic species. For example, POPs in resident Pacific herring (*Clupea pallasii*) in Puget Sound cannot be predicted by POP levels or trends in the sediment. The three known herring populations in the Puget Sound region reflected different POP patterns, suggesting differential exposure to contaminants and that this difference was related to where these species feed (West et al. 2008). Pacific herring heavily rely on krill, 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 that are then consumed by pelagic organisms, exposing the pelagic food web. This, in addition to deposition in sediments and the benthic food web, is likely a route by which ESA-listed marine mammals and sea turtles would be exposed to POPs from the proposed discharge of effluent by ITP and APTP through the SBOO.

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

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

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

The timing of exposure to POPs can affect the degree of toxicity. The most critical or sensitive period for developmental neurotoxicity appears to occur during the height of the brain growth spurt. For example, neonatal mice exposed to BDE-99 during a critical period of brain development experienced impaired spontaneous behavior (i.e., behavior important for survival such as foraging and predator avoidance). However, mice exposed after the growth spurt did not experience the neurotoxic effects (Eriksson et al. 2002). This study indicates that adverse health effects are not only dose-dependent and species-specific, but also dependent on the timing of exposure. Other studies where animals are exposed to POPs during the defined critical period have shown to cause reductions in sperm and spermatid counts in adult rats, increase hyperactivity in their offspring, cause morphological effects in the thyroid, liver, and kidneys, increase circulating thyroid hormones, and alter spontaneous behavior (Viberg et al. 2003, 2007; Kuriyama et al. 2005). Additionally, neonatal exposure may produce long-term modifications in the cholinergic or neurotransmitter system (Talsness 2008). Therefore, marine mammal calves and pups are likely more susceptible to adverse health effects than are adult whales and pinnipeds because the young are exposed to contaminants during the critical period of development. The influx of toxicants in calves and pups is a cause for concern because the growth and development of an individual is highly dependent on normal levels of thyroid hormones (Boas et al. 2006).

While POPs can present direct health threats to hormonal regulation, neural development and function, and reproduction as discussed above, they can also alter susceptibilities to infectious diseases. One mechanism of action of inducing contaminant effects is through interactions with the aryl hydrocarbon receptor, generally described as “dioxin-like” effects. “Dioxin-like” contaminants are particularly effective at engendering immunotoxicity across a range of species.

PCBs, PBDEs, and DDTs have well documented effects on the immune system in a wide range of experimental animals (Thomas and Hinsdill 1978, 1980; Safe et al. 1989; Dahlman et al. 1994). In the absence of a robust immune system, the individual animal's health, or its ability to endure and thrive, can become compromised. The immune system is important in patrolling and eliminating cells that undergo malignant transformation. If this immune surveillance is compromised, the potential exists for tumors to develop. For example, St. Lawrence beluga whales have a high occurrence of tumors and lesions, and some evidence of immunosuppression, along with high PCB concentrations (Béland et al. 1993; Martineau et al. 1994). California sea lions that died of carcinoma had higher PCB concentrations compared to California sea lions that died without carcinoma (Ylitalo et al. 2005). Contaminants may play a role in the development of disease by suppressing the immune system or through genotoxic mutation and tumor promotion (Ylitalo et al. 2005).

Data from toxicity testing, epidemiological studies, and risk assessments all suggest that there are health concerns at current exposure levels for organophosphate flame retardants (Blum et al. 2019). There are known animal carcinogens classified as organophosphate esters such as TCEP and TDCPP (EPA 2015) in addition to a potential carcinogenic compound, TCPP (Van der Veen and de Boer 2012). The National Toxicology Program is in the process of evaluating TCPP in a 90-day toxicity study and a 2-year cancer bioassay and developmental toxicity study (EPA 2015). In addition to the carcinogenic effects, TDCPP induces acute-, nerve-, developmental-, reproductive-, hepatic-, nephron-, and endocrine-disrupting toxicity in animals, which has caused increasing concern worldwide (Wang et al. 2020). As for TCPP, the structure is similar to a an established class/family of neurotoxic chemicals of organophosphate pesticides, (Xia et al. 2021), and it has been reported to show developmental and reproductive toxicity in pregnant rats (Ji et al. 2020).

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

Less is known about early POP exposure among sea turtles. Recent studies have identified POPs transferred from nesting females to eggs and hatchlings likely have consequences on development. For example, POP concentrations in green sea turtles were significantly negatively correlated with body condition of hatchlings, an indication of effects on development (Van de

Merwe et al. 2010). POPs may disrupt normal hormone function by altering the concentrations of circulating thyroid hormone (e.g., Hall et al. 2003) as well as interfere with developmental processes (Eriksson et al. 2002, 2006). Recently, Finlayson et al. (2016) reviewed the available sea turtle toxicological research and identified only 49 papers on sea turtle toxicology, highlighting the need for more toxicological endpoints and mixture effects studies.

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

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

2.5.3.3 Mixture Effects and Non-Linear Dose-Response Curves

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

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

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

2.5.3.4 Summary – marine mammals and sea turtles

Effluent discharged from this proposed action contains potentially harmful contaminants that have been well established to adversely affect laboratory and wildlife species. PCBs and DDTs continue to be measured in the action area through sediment, mussel and fish liver/muscle sampling. The threat of TBT may be uncertain, but it will be subject to monitoring in the effluent. The potential presence of organophosphate flame retardants (e.g., TPPO, TCEP, TCPP) is of concern as well. Recent information indicates that increasing use of organophosphate flame retardants is leading to an increased presence and therefore threat from potential exposure to these constituents. Based on surrogate data from a similar WWTP, we estimate approximately

267 metric tons per year of organophosphate flame retardants may be loaded into the action area following ITP expansion and approximately 340 metric tons per year of organophosphate flame retardants may be loaded into the action area following APTP construction. Once in the aquatic system, these constituents become bioavailable to food webs for uptake by ESA-listed species.

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

As described above, we expect that individuals from all of the ESA-listed marine mammal and sea turtle species that may occur in the action area may make numerous or possibly frequent and extended visits to the area and be exposed to additive accumulation of POPs, increasing the risks of adverse effects that these contaminants are known to present. As described in Section 2.4 (Environmental Baseline), there are numerous other potentially harmful contaminants for ESA-listed marine mammals and sea turtles, and many of those are likely to also be present in the proposed discharge of effluent by ITP and APTP through the SBOO. However, the information describing the levels of most of these contaminants in wastewater discharge (and the extent of their potential harmful effects) is limited.

2.5.3.5 Adverse Abalone Health Effects from Exposure to Potentially Harmful Contaminants

Studies evaluating the effects of more persistent and potentially harmful contaminants such as POPs on white abalone and other California abalone species are lacking. Thus, we use the best available information from studies involving other abalone species worldwide to infer potential effects of exposure on white abalone.

Field studies in Japan found that exposure of *H. madaka* and *H. gigantea* to the organotin compounds TBT and triphenyltin (TPhT; found in anti-fouling paint) caused ovarian spermatogenesis (masculinization) and altered the timing of reproductive maturity in males and females (Horiguchi et al. 2001, 2005). Because abalone rely on synchronous spawning, this altered timing can result in reduced reproductive potential. Lab studies exposing *H. gigantea* to 100ng TBT/L or 100ng TPhT/L for two months found similar effects: ovarian spermatogenesis, contracted primary oocytes, and high concentrations of TBT and TPhT in head and muscle tissue (Horiguchi et al. 2002). Horiguchi et al. (1998) also observed adverse effects on larval survival, behavior, and development from exposure to TBT and TPhT. For *H. discus discus*, 48 hour LC_{50} values were 5.4 µg/L for TBT and 1.4 µg/L for TPT; for *H. madaka,* 24 hour LC_{50} were 3.9 µg/L for TBT and 2.4 µg/L for TPT and 48 hour LC_{50} were 1.2 µg/L for TBT and 1.5 µg/L for TPT

(Horiguchi et al. 1998). Horiguchi et al. (1998) observed irregular swimming and cilia movement in larvae exposed to concentrations of TBT and TPhT below these LC_{50} values.

Effects on abalone can also occur at the cellular and molecular level. Gaume et al. (2012) found toxic effects on immune and respiratory cells of *H. tuberculata* when exposed to concentrations of triclosan (an antibacterial agent) ranging from 2 to 10µM for 24 to 48 hours. Zhou et al. (2010) exposed *H. diversicolor supertexta* to diallyl phthalate (50 ug/L) and bisphenol A (100 ug/L) for three months and found altered protein expression that could affect physiological functions such as detoxification, immunity, metabolism, and hormonal modulation.

Tributyltin has been detected in the ITP's effluent at levels at or exceeding those described above. In 2021, the monthly average concentration of TBT in the effluent was 1.40 µg/L (ERG and Tenera Environmental 2022). TPhT, triclosan, diallyl phthalate, and bisphenol A were not included in the list of CECs monitored in the effluent. Several other persistent and potentially harmful contaminants have been detected in the SBOO effluent, but we do not know how these may affect abalone (City of San Diego 2022a). In general, what these studies and their results show is that exposure to endocrine disruptors and other chemicals, including those found in the SBOO's effluent, can have harmful effects on abalone growth and reproductive development. We expect any white abalone occurring on the SBOO structure (within the ZID) to be exposed to the highest concentrations. White abalone occurring within the action area but outside the ZID would be exposed to low concentrations due to dilution of the plume; however, we do not know how these contaminants may accumulate in abalone over time nor what effects this accumulation may have on their health, though reduced growth and reproductive capacity are possible.

Overall, we expect white abalone within the action area to be exposed to potentially harmful contaminants in the proposed discharge of effluent by ITP and APTP through the SBOO and to accumulate these contaminants over time, which may result in adverse effects on individual health. Further studies are needed to evaluate the concentrations of potentially harmful contaminants in the effluent, the concentrations to which white abalone may be exposed, and the effects of these potentially harmful contaminants on abalone at those concentrations over time.

2.5.4. Harmful Algal Blooms

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

2.5.4.1 Effect of the Proposed Discharge of Effluent from ITP and APTP through the SBOO on HAB Occurrence

This proposed action's discharge may have the effect of fertilizing or kick-starting the springtime HABs by sustaining or even increasing populations of HAB species in the subsurface water (Trainer et al. 2007; Cochlan et al. 2008; Seeyave et al. 2009; Kudela et al. 2010; Nezlin et al. 2012; Seegers et al. 2015) providing nitrogen to the euphotic zone of the action area. Concentrations of nitrogen (and phosphorus) in the effluent plumes are up to three orders of magnitude greater than maximal ambient concentrations, and they rise to the thermocline or to the surface (Reifel et al. 2013; Seegers et al. 2015). 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.

Mantyla et al.(2008) noted chlorophyll layers at 85 meters deep in offshore waters of the SCB, while a recent paper by Kessouri et al. (2021) defined the euphotic zone in the SCB as 0-40 meters deep. Stratification typically maintains the SBOO plume at a depth of only 26' during the warmer months of the year, but the plume surfaces during other periods of the year when thermal stratification ceases at this location (Largier et al. 2004; Terrill et al. 2009). Therefore the nutrients are clearly present in the euphotic zone for use by phytoplankton, including HAB species.

The physical oceanography in the vicinity of the proposed discharge of effluent by ITP and APTP through the SBOO influences the fate and transport of the nutrients and any subsequent phytoplankton or zooplankton that utilize the nutrients to grow. As discussed in the environmental baseline section, ocean conditions within the action area are affected by both regional- and local-scale currents, and local gyres, particularly in the area south of the Point Loma headland (City of San Diego 2021a). The meandering California Current has a large shoreward component near 32º N; this component splits as it approaches the coastand greatly affects the action area. Recent modeling (Kessouri et al. 2020b) confirms previous work (Lynn and Simpson 1987; Mantyla et al. 2008; Rogowski et al. 2012b) by showing that local currents have considerable variability in position, strength, and depth, and can often be coming ashore right toward the action area. As a result, near-shore currents in the area have been observed by the City of San Diego (City of San Diego 2016, 2021a, 2021b) variously heading generally north or south directing the effluent towards La Jolla or the Mexican border. Nezlin et al (2012) identified the south San Diego area as a hot spot area with longer residence time of its water and higher CHL-α levels. Additional nutrients may enter the immediate vicinity of the SBOO from the SABTP, Tijuana River flows, and other urban stormwater runoff (Feddersen et al. 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. 2012b, 2014). There are several sources that summarize numerous studies and conclude that reduced forms of nitrogen (ammonium, urea) significantly tilt the phytoplankton community toward the development of HABs (Howard et al. 2012b; Reifel et al. 2013; Booth 2015; Seegers et al. 2015). Kudela et al. (2008) showed that ammonium uptake by *A. sanguinea*, a red tide forming dinoflagellate that does not produce a toxin, was approximately threefold higher than uptake of nitrate. Kudela et al. (2010) later showed that *P. spp*. grew equally well or better on reduced nitrogen sources and Howard et al. (2007) showed that *P. australis* could use either nitrate or ammonium simultaneously. Schnetzer et al. (2007) cites several studies that examined *P-spp*. and noted that their effective toxicity can be highly variable. 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 can have the effect of unbalancing these ratios at the local level and may be partially responsible for the very potent HABs that have been occurring in the spring for ~10 years in SCB (Nezlin et al. 2012).

Howard et al. (2014) estimated that nutrient loading of the San Diego subregion through wastewater effluent increased total nitrogen in the San Diego subregion by about 16,310 pounds (7400 kg) of nitrogen per km^2 per year. Given the area of the San Diego subregion (1020 km^2), this amounts to about 16.6 million pounds (7.5 million kg) of nitrogen over the course of a year. As described in Section 2.4.1.1.4 (Harmful Algal Blooms), this amount of nitrogen was estimated by Howard et al. (2014) to be roughly equal to three times the contribution of upwelling in the San Diego area (as defined therein) although the upwelling estimate is likely an underestimate.

Recent estimates of average effluent discharge volume at the PLOO have been approximately 140 MGD (City of San Diego 2021b), which is less than the dischages examined by Howard et al. (2014) but more than twice as large as recent effluent discharges from the SBOO (~40 MGD maximum) and San Antonio de los Buenos WWTP (~25 MGD) combined. An additional facility to the north (University of California, San Diego – Scripps Institution of Oceanography) discharges ~1.25 MGD of mostly aquarium related and filter backwash flows (no "sewage" processed here) that we consider inconsequential in this analysis. Unfortunately, the San Antonio de los Buenos WWTP is in disrepair and we cannot accurately estimate the nutrient loading from it. Relative comparison of the nutrient loading from the existing ITP and the SBOO discharge in total with certainty in the action area is not possible due to these uncertainties. Although the available information is insufficient to precisely assign the nutrient contributions of historical discharge from ITP through the SBOO as a percent of the total in the action area, it is evident that upon ITP expansion and completion of APTP construction, the discharge of effluent by ITP and APTP through the SBOO will significantly contribute to the overall discharge amount of the receiving water within the action area.

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

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

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

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

In mammals, saxitoxins appear to affect the peripheral nervous system, and the primary cause of death in response to exposure is respiratory paralysis. This biotoxin is considered to be responsible for PSP, and can act quickly in species following exposure through accumulation in prey (O'Hara and O'Shea 2001). Some of the symptoms of exposure to this biotoxin can include lethargy, lack of motor control, paralysis, and death (Van Dolah et al. 2003). Domoic acid causes the syndrome known as Amnesic Shellfish Poisoning. In humans, permanent loss of short term memory was experienced following domoic acid exposure (Van Dolah et al. 2003). Other symptoms in humans from domoic acid have included nausea, vomiting, diarrhea, dizziness, disorientation, lethargy, and seizures (Van Dolah et al. 2003). Signs of domoic acid toxicity in wildlife can include seizures, head weaving, decreased responsiveness to stimuli and scratching behavior (Work et al. 1993; Van Dolah et al. 2003).

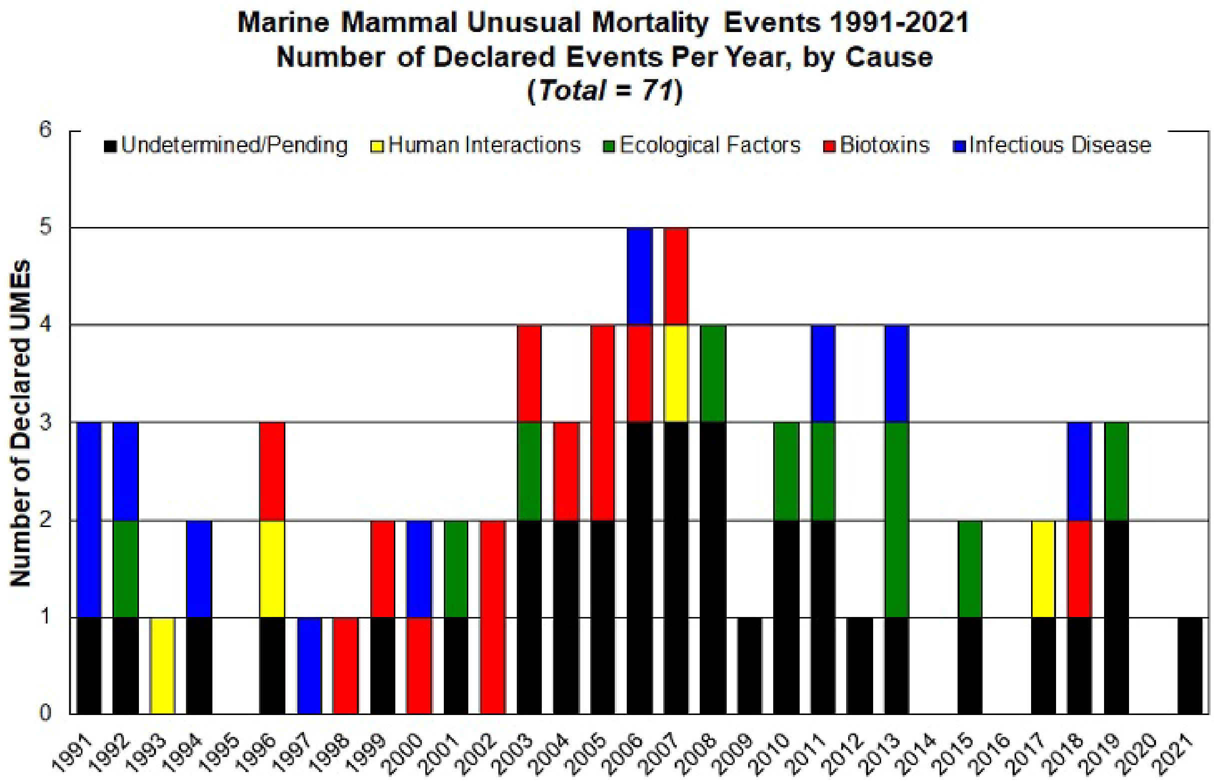

Figure 5. Number of unusual mortality events between 1991 and 2021 per year and by cause (figure reprinted from http://www.nmfs.noaa.gov/pr/health/mmume/events.html).

These biotoxins clear rapidly from the blood. Consequently, diagnosis in marine mammals is difficult without a thorough examination. For example, the highly endangered Mediterranean monk seals (*Monachus monachus*) found in coastal waters off West Africa experienced a mortality event in 1997 that reduced the population abundance to almost half its size (Forcada et al. 1999). Osterhaus et al. (1997) had identified morbillivirus in several of the monk seal carcasses, and that was considered a likely cause of the mortality event. However, a subsequent competing theory suggested the mortality event may have been caused by biotoxin exposure (Hernández et al. 1998). Terminally ill individuals exhibited the known clinical symptoms of exposure to saxitoxin (e.g., lethargy, motor incoordination, and paralysis). Furthermore, they found that the time between onset of these clinical signs and death through drowning by paralysis was short. Hernandez et al. (1998) also detected high concentrations of the dinoflagellate *Alexandrium minutum* in the coastal waters as well as in the dead seals. However, because there are no data on the background levels of these toxins in the seals or their prey, and there are no baseline data on the prevalence of virus antibodies, a conclusive diagnosis could not be made (Harwood 1998).

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

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

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

Following a *P. australis* bloom in Monterey Bay in 2000, 25 gray whales stranded in the San Francisco Bay area (Van Dolah et al. 2003). Approximately half of the whales were sampled for domoic acid, one of which had levels at concentrations that would implicate domoic acid toxicity. Because clearance of this biotoxin is fast, it is not clear if the other whales had been exposed as well. It was previously believed that gray whales typically do not forage during their northern migrations from the nursery grounds to their feeding grounds. However, gray whales have been observed feeding off California and Washington (Van Dolah et al. 2003). Krill were also collected offshore of Monterey Bay following the bloom and identified as a potential vector for domoic acid to higher trophic level species (Bargu et al. 2002). Based on the maximum domoic acid concentrations measured in krill, Bargu et al. (2002) estimated krill could transfer domoic acid levels up to 62 grams to a blue whale per day, or 0.62 mg per kg.

In 2002, a UME was declared as over 2,000 animals stranded in southern California from April to June with neurological symptoms. The affected species included mostly California sea lions and long-beaked common dolphins (*Delphinus 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 affect sea turtles. Brevetoxins are produced by the dinoflagellate *Karenia brevis* and best known to be responsible for Florida red tides. Between 2005 and 2006, 318 sea turtle strandings were documented off Florida's Gulf of Mexico (a four-fold increase). The cause of death in approximately 90% of the individuals was from red tide intoxication (Fire and Van Dolah 2012) (http://www.whoi.edu/redtide/page.do?pid=153356). The data revealed the turtles were consuming contaminated prey and inhaling the toxin.

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

More recent evidence was acquired from a stranded leatherback in Santa Cruz, CA, that was in good body condition and had no obvious cause of death (NMFS WCR strandings data). High domoic acid was measured, and the pattern was consistent with it being metabolized (i.e., increasing in concentration down the gastrointestinal tract and was observed in the intestines, stomach, feces, and bladder). The domoic acid ranges detected in the sea turtle were within the range detected in acutely intoxicated seizuring California sea lions. These data reveal sea turtles can be exposed to and affectred by biotoxins, but it is currently unclear whether the effects are similar to that found in birds and marine mammals.

2.5.4.3 Potential Adverse Abalone Health Effects from Exposure to HABs

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

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

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

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

2.5.4.4 Summary

Similar to the analysis of potential effects of adding potentially harmful contaminants like POPs to the environment and increasing the accumulation of these contaminants by ESA-listed species, we conclude that the discharge of effluent by the proposed action can potentially increase HAB frequency, extent and intensity. At this time, we cannot predict the precise extent that the proposed action will contribute to increased probabilities of HABs, or distinguish which HABs may be more or less associated with or influenced by the additional nutrient input created by the proposed discharge of effluent by ITP and APTP through the SBOO What is clear is that (1) HABs pose a significant health risk for ESA-listed marine mammals, sea turtles, and abalone; (2) increasing the probability of HAB occurrence further increases the likelihood of adverse effects from HABs, including impaired health (injury) and mortality; and (3) the proposed effluent by ITP and APTP through the SBOO increases that probability. As described above, we expect that all of the ESA-listed marine mammal and sea turtle species that may occur in the action area have individuals that may: (a) make numerous or possibly frequent and extended visits to the area; and (b) be exposed to increased frequency or extent of HABs during those visits, increasing the HAB-based risks of adverse effects. White abalone in the area would be exposed to the greater risks associated with increased HAB frequency, extent and intensity.

2.5.5. Risks to Populations

2.5.5.1 Marine Mammals and Sea Turtles

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

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

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

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

However, given the available information, it is difficult to assess how these effects are expected to affect ESA-listed marine mammals and sea turtles at the population and species level. Long-term effects on individuals, including diminished reproductive capacity and lower survival rates, could result from the continued accumulation of potentially harmful contaminants that would likely be intensified by the proposed action. Increased and augmented HABs resulting from the proposed action are also likely to produce more acute effects, such as physical impairments, reduced foraging, disorientation, and even death among listed animals. These 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 the proposed discharge of effluent by ITP and APTP through the SBOO, based on evidence that individuals of these species visit the action area. Given the transitory nature of most of these species and their broad distribution in the Pacific, exposure to the proposed action is likely somewhat limited at the population scale to relatively small segments of populations that may occasionally visit or favor the area as opposed to large proportions or entire populations. There may be some exception to this premise for smaller populations like Central American DPS humpback whales where many, if not all, members of the population may visit the SCB on an annual basis and potentially could occur near or within the action area occasionally and be exposed. However, although we can generally describe the extent of the exposure of individuals to the discharge and its effects, we cannot further describe exactly how many individuals or what percentage of these populations will be exposed or potentially affected. In addition, as explained above, the extent of effects (reduced health, changes in reproductive success or capacity, or even death) that can be expected at the individual level are also highly uncertain, making it difficult to anticipate what the population level effects may be. As described in Section 2.5 (Effects of the Action), we generally expect that exposure will be limited to relatively few individuals (adults or juveniles) or small portions of these populations over time. Exposure is more likely for individuals that may have some preference for or site fidelity to the action area. Although there is uncertainty in the specific extent of population level exposure, at this time we generally do not anticipate widespread effects across populations that could potentially produce reduced productivity or fitness at a population level for any of these species.

2.5.5.2 Abalone

In summary, the discharge effluent could affect white abalone in the following ways:

- Reducing the amount of artificial reef habitat available to white abalone on the SBOO structure;
- Injuring and killing individuals that are removed as part of the diffuser riser recommissioning activities;
- Exposing individuals to potentially toxic pollutants in the effluent;
- Exposing individuals to more persistent pollutants in the effluent that can accumulate; and
- Exposing individuals to increased HAB frequency, intensity, and extent.

There is the potential, though low, for white abalone to be found on the SBOO structure itself, including on the wye diffuser. We expect reductions in artificial reef habitat to result in limited disturbance to any white abalone on the SBOO structure, because the loss of habitat would be small compared to the total amount of habitat available on the SBOO structure and the loss would be temporary as organisms recolonize the cleared areas. We estimate up to two white abalone to be present within the areas to be cleared, and that removal would injure and kill those white abalone, unless conducted by experienced personnel and followed by relocating the white abalone to suitable habitat nearby (and monitoring to assess survival over time) or by transporting the white abalone to an approved captive holding facility (permitted under ESA Scientific Research and Enhancement Permit 14344-2R issued to the UCD-BML).

The remaining effects would result from the continued and increased discharge of treated wastewater via the proposed effluent by ITP and APTP through the SBOO. We expect white abalone within the action area to be exposed to varying concentrations of the ITP and APTP's discharge effluent. White abalone on rocky reefs off La Jolla, Point Loma, and Imperial Beach would be exposed to diluted concentrations of the effluent plume, whereas any white abalone on the SBOO structure would be exposed to higher concentrations, particularly within the ZID.

Within the ZID, the concentrations of heavy metals in the effluent discharge exceed the levels found to inhibit growth, feeding, and survival in juvenile and adult abalone. However, we expect the presence of juveniles and adults on the SBOO structure to be low, given the low likelihood that larvae can settle and survive and the distance of the structure from suitable rocky reef habitat. Outside of the ZID, we expect the concentrations of heavy metals in the plume to be well below the levels found to cause harmful effects on other abalone species, and therefore likely to have minimal effects on white abalone. However, we also expect white abalone to be exposed to low concentrations of more persistent contaminants in the plume, which can accumulate over time and result in adverse effects on the health of individual white abalone. The potential increase in HAB frequency, intensity, and extent due to the proposed action's discharge also poses an increased risk of killing abalone.

Based on our analysis, we conclude that effects of the proposed action, including SBOO recommissioning activities and increased discharge of contaminants and nutrients over time have the potential to reduce the fitness and survival of individual juvenile and adult white abalone in the action area. We cannot discount these effects as extremely unlikely to occur or dismiss them as insignificant.

The extent of these effects at the individual level are highly uncertain, making it difficult to anticipate what the population level effects may be. However, we expect effects on individual fitness to be minimal and restricted to a few animals only, thereby also limiting effects at the population level. We expect any white abalone occurring on the SBOO structure to experience the most severe effects; however, we also expect few (if any) white abalone on the SBOO structure given the distance to the nearest rocky reef habitat. White abalone occur on rocky reefs off La Jolla and Point Loma, but would be exposed to highly diluted, low concentrations of the plume, given the distance from the SBOO discharge point. We do not expect that all abalone in the action area will be exposed to contaminants in the plume at the same concentrations, resulting in varying levels of exposure, uptake, and accumulation across individuals. We also do not expect that all abalone in the action area will be exposed to all HABs that occur within the area. Given the distribution of abalone in the action area and the best available information on past effects, we expect HAB-related impacts, including mortality, to generally be limited to a few abalone in a confined area at any given time, limiting the effects on the population and species as a whole.

2.6. Cumulative Effects

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

Some continuing non-Federal activities are reasonably certain to contribute to climate effects within the action area. However, it is difficult if not impossible to distinguish between the action area's future environmental conditions caused by global climate change that are properly part of the environmental baseline *vs.* cumulative effects. Therefore, all relevant future climate-related environmental conditions in the action area are described earlier in the discussion of environmental baseline (Section 2.4).

As described in the environmental baseline, the SABTP facility discharging near the U.S.-Mexico border is expected continue to discharge in the vicinity of the action area. Dischages from this facility may cease in the future, but when that may occur is uncertain.

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

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

2.7. Integration and Synthesis

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

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

The proposed action is the issuance of U.S. appropriations under the U.S.-Mexico-Canada Agreement (USMCA) implementation Act to the EPA and USIBWC to design and construct water infractrure projects to address impacts from transboundary flows in the Tijuana River watershed and adjacent coastal areas. EPA and USIBWC expect the overall proposed action will reduce the transboundary flow of untreated wastewater from Mexcio into the U.S. waters off San Diego County. EPA and USIBWC also expect that the proposed action will increased the discharge of treated wastewater through the SBOO off San Diego County through the expansion of the capacity of the ITP and the additional capacity that will be created by the APTP once constructed. The sections below consider the proposed action and anticipated effects associated with the capacities that are being created to accommodate wastewater treatment and discharge needs out to 2050.

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 the proposed discharge of effluent by ITP and APTP through the SBOO 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 the proposed discharge of effluent by ITP and APTP through the SBOO may contain numerous other contaminants that could potentially harm ESA-listed species, but the lack of information on these contaminants, their effects, and their concentrations limits our ability to analyze those effects further.

As described in Section 2.4 (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 the proposed discharge of effluent by ITP and APTP through the SBOO 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 the proposed discharge of effluent by ITP and APTP through the SBOO.

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. The effects from these factors pose potential continuing threats to the health of ESA-listed marine mammals and sea turtles that may visit the action area and to the action area as a while. Climate change could influence the migration and distribution of prey species, the relative exposure of various individuals and ESA-listed populations within the action area, and increase the probability and/or magnitude of HAB occurrence in the action area over time.

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

2.7.1.1 Blue Whale

Over the course of the proposed action, we anticipate that some individual blue whales may occasionally enter the action area and be harmed by the proposed action, especially during the summer months. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the proposed discharge of effluent by ITP and APTP through the SBOO is a gradient and overlap of listed species with the action area is a transitory in nature. Although the ENP stock of blue whales is relatively small (1,898 individuals), exposure to the proposed action will likely be limited to a small number of individuals and the population that may be affected constitutes only a small portion of the globally-listed blue whale species. At this time, additional information is needed to more fully evaluate the exposure of blue whales to the proposed discharge of effluent by ITP and APTP through the SBOO and the anticipated effects at an individual and population level.

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

2.7.1.2 Fin Whale

Over the course of the proposed action, we anticipate that some individual fin whales may occasionally enter the action area and be harmed by the proposed action at any time during the year. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the proposed discharge of effluent by ITP and APTP through the SBOO is a gradient and overlap of listed species with the action area is a transitory in nature. The CA/OR/WA stock of fin whales is estimated to consist of 11,065 individuals, although exposure to the proposed action will likely be relatively limited to a small number of individuals, and the population that may be affected constitutes only a portion of the globally-listed fin whale species. At this time, additional information is needed to more fully evaluate the exposure of fin whales to the proposed discharge of effluent by ITP and APTP through the SBOO and the anticipated effects at an individual and population level.

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

2.7.1.3 Humpback Whale; Mexico DPS

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

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

2.7.1.4 Humpback Whale; Central America DPS

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

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

2.7.1.5 Gray Whales; WNP DPS

Over the course of the proposed action, we anticipate that some individual WNP gray whales may occasionally enter the action area and be harmed by the proposed action during the winter and spring migrations each year. As described before, there is a small likelihood (less than 1%

chance) that any individual gray whale that may enter the action area could belong to the WNP population of gray whales. It is likely that at least one or more WNP gray whales would enter the action area during the proposed action and be at risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the proposed discharge of effluent by ITP and APTP through the SBOO is a gradient and overlap of listed species with the action area is a transitory in nature. The WNP population of gray whales is very small (271 individuals), but exposure to the proposed action will likely be extremely limited given their migratory behavior through such a small action area, the limited number of WNP gray whales that may occur in the action area, and the limited potential for foraging to occur.

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

2.7.1.6 Guadalupe Fur Seals

Over the course of the proposed action, we anticipate that some individual Guadalupe fur seals may occasionally enter the action area and be harmed by the proposed action, especially during the summer months. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the proposed discharge of effluent by ITP and APTP through the SBOO is a gradient and overlap of listed species with the action area is a transitory in nature. The Guadalupe fur seal population is estimated to be at least 31,091 individuals (Carretta et al. 2022), although exposure to the proposed action will likely be limited to a small number of individuals and a small portion of the population.

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

2.7.1.7 Green Sea Turtle; East Pacific DPS

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

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

2.7.1.8 Leatherback Sea Turtle

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

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

2.7.1.9 Loggerhead Sea Turtle; North Pacific Ocean DPS

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

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

2.7.1.10 Olive Ridley Sea Turtle

Over the course of the proposed action, we anticipate that some individual olive ridley sea turtles, most likely from Mexican nesting beach origins, may occasionally visit the action area and be harmed by the proposed action. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only, and death is an unlikely outcome in all cases. Moreover, the concentration of contaminants from the proposed discharge of effluent by ITP and APTP through the SBOO is a gradient and overlap of listed species with the action area is a transitory in nature. While there is no specific estimate of abundance for the Mexican nesting beach population, the total abundance of olive ridleys in the eastern tropical Pacific exceeds one million individuals, which includes hundreds of thousands of individuals from the Mexican nesting beach population. We expect that exposure to the proposed action will be limited to a small number of individuals and a small portion of the population.

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

2.7.2. Abalone

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

The proposed action is expected to reduce the overall discharge of pollutants and nutrients into the action area from SAB Creek and TJRE, but is also expected to increase the volume of treated effluent discharged into the action area via the SBOO. As described in Section 2.5 (Effects of the Action), the proposed action would result in continued exposure of white abalone to the effluent plume, potential accumulation of harmful contaminants, and to HABs that may occur more frequently and to a larger extent due to the discharge. These individuals will be at increased risk of diminished health and fitness, and even death. However, these fitness effects are expected to be minimal and restricted to a few animals only.

In general, the levels of heavy metals and persistent contaminants that have been reported in the SBOO effluent are lower than the levels found to significantly reduce survival, growth, and reproductive development in abalone, except for levels of copper, silver, zinc, and TBT. White abalone present within the ZID would be exposed to higher concentrations of these pollutants; however, the likelihood that white abalone occur within the ZID is very low. Most, if not all, white abalone within the action area are expected to occur outside of the ZID, where they will be exposed to highly diluted, low concentrations of these contaminants in the plume. However, for the more persistent contaminants, it is unknown how exposure may increase accumulation in and affect individual abalone. We note that the effects of many of the CECs and other potentially harmful contaminants in the effluent have yet to be evaluated, singly or in combination with one another. We expect that the continued and increased discharge of effluent under the proposed action is likely to increase the uptake of potentially harmful contaminants by white abalone within the action area, but the degree of exposure and the effects on the health of individuals and the population as a whole are highly uncertain.

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

In addition, the recommissioning of SBOO diffuser risers may result in the loss of artificial reef habitat and the removal, injury, and killing of white abalone from the SBOO wye diffuser. However, we expect the loss of habitat to be limited and temporary, and the likelihood of white abalone on the wye diffuser to be low, resulting in few (if any) white abalone being removed during recommissioning activities. If white abalone are found and need to be removed, their likelihood of survival will increase if removal is conducted by experienced personnel and the abalone are relocated to suitable habitat nearby or transported to an approved captive holding facility permitted under ESA Permit 14344-2R issued to the UCD-BML.

In summary, the proposed action may adversely affect survival, growth, and reproductive development of white abalone within the action area, further exacerbating the risks of low density and reduced reproductive capacity for this population. These effects would be in addition to the ongoing effects of other discharges into the action area, warming water temperatures, ocean acidification, and other threats. However, we expect these effects to be limited to a few individuals in a confined area. We expect the greatest effects on any white abalone occurring on the SBOO structure, which is likely to be few (if any) individuals. White abalone are most likely to occur on rocky reefs off La Jolla and Point Loma, where the proposed action’s discharge plume is expected to be diluted and at low concentrations.

We acknowledge there is uncertainty regarding the specific occurrence and magnitude of expected effects based on the available information. The following information would inform our evaluation of the proposed action’s effects on white abalone: the abundance and distribution of white abalone in the action area, levels of more persistent and potentially harmful contaminants in the effluent and their effects on white abalone, and the effects of the effluent on the frequency and extent of HABs within the action area that may harm white abalone.

Overall, we do not expect the proposed action to reduce the likelihood of survival and recovery of white abalone, based on our current understanding of the proposed action’s potential effects and the limited number of white abalone likely to be affected, despite the acknowledged uncertainties regarding the magnitude and intensity of those effects on the species’ status.

2.8. Conclusion

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

2.9. Incidental Take Statement

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

2.9.1. Amount or Extent of Take

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

NMFS anticipates that take of up to two white abalone could occur during SBOO construction activities to recommission the diffuser risers, when divers remove habitat and species on the SBOO structure around the risers to be modified. We expect the removal of white abalone from the SBOO structure to kill the abalone, unless removal is conducted by experienced personnel and the abalone are relocated to suitable habitat nearby (and monitored to assess survival) or transported to a captive holding facility (as permitted under ESA Permit 14344-2R issued to the UCD-BML). Additional incidental take of white abalone is expected under the proposed action as described below.

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

We cannot further enumerate the anticipated take of ESA-listed species from the proposed action as a result of the uptake and/or accumulate potentially harmful contaminants including POPs such as organophosphate flame retardants, due to uncertainty in the number of individuals that may be subject to exposure and uncertainty in the response and level of harm that would occur for individuals exposed from each ESA-listed species. Instead, we can describe the extent of take associated with the potential accumulation of potentially harmful contaminants by relating the extent of take to the amount of these potentially harmful contaminants being discharged into the action area via the proposed effluent by ITP and APTP through the SBOO as a consequence of the proposed action. 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 future organophosphate flameretardant 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 ITP and APTP through the SBOO as a consequence of this proposed action. As we described in Section 2.5 (Effects of the Action), the levels of organophosphate flame retardants discharged by ITP have not been documented historically. Using available information from a similar WWTP, we estimated that this proposed action by ITP and APTP through the SBOO could discharge up to approximately 267 metric tons and 340 metric tons of organophosphate flame retardants per year, following ITP expansion, and construction of APTP, respectively. These organophosphate flame retardants will be released into the ecosystem and will be potentially bioavailable for uptake into the food web and ESA-listed species.

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

At this time, we cannot predict the precise extent that the proposed action's effluent discharge will contribute to increased probabilities of HABs, or distinguish which HABs may be more or less associated or influenced by the additional nutrient input from the proposed action of discharge by ITP and APTP through the SBOO. Consequently, we cannot further enumerate the anticipated take of ESA-listed species from the proposed action. Instead, we can describe the extent of take associated with increased probabilities of harmful effects from exposure to HABs by relating the extent of the increased probability of HABs to the amount of nitrogen, specifically in the form of ammonia, that will be released via the ITP and APTP through the SBOO as a consequence of the proposed action into the action area. We elect to use the extent of ammonia discharged as a surrogate to describe the extent of take associated with risks of increased probability of HAB exposure for ESA-listed species as a result of the proposed action because nitrogen is the primary nutrient limiting phytoplankton production in coastal waters (Booth 2015). We also use ammonia because it has been routinely monitored in the effluent (on a weekly basis) in contrast to other forms of nitrogen. Ammonia is estimated by Howard et al. (2014) as making up 92% of the nitrogen in effluents discharged by WWTPs in the SCB, and therefore likely accounts for the vast majority of nitrogen discharged by this WWTP.

We have therefore quantified the potential incidental take of the proposed action in terms of ammonia that we expect to be discharged by ITP and APTP through the SBOO as a consequence of this proposed action. In 2021, the maximum recorded value of ammonia nitrogen concentrations in ITP WWTP's effluent was 57.4 mg/L (ERG and Tenera Environmental 2022). This value of ammonia nitrogen concentration is useful to calculate an expectation for ammonia discharge based on current flows that we can use as a take surrogate upon ITP expansion. Assuming that the wastewater treatment processes at ITP (following ITP expansion) remain approximately the same in their effectiveness at removing ammonia from the wastestream, a maximum annual emission rate was estimated following completion of ITP expansion. Using an average discharge rate of 60 MGD by ITP via the SBOO following ITP expansion, the expected maximum annual mass emission discharge rate of ammonia will be up to 4,760 metric tons per year[3] as a result of the proposed action. ERG and Tenera Environmental (2022) estimated that total nutrients (consisting of both nitrogen and phosphorus) of 40.9 mg/L will be discharged from the SBOO following APTP contruction, although they did not separate nitrogen and phosphorus in this estimate. Therefore, we used both the provided total nutrient load estimate of 40.9 mg/L and the maximum recorded ammonia concentration from ITP effluent from 2021 (57.4 mg/L) for estimating the mass emission range of ammonia following ITP expansion and APTP construction. Using an average discharge rate of 76.4 MGD by ITP and APTP (following APTP construction) through the SBOO, the annual mass emission discharge rate of ammonia is expected to be between 4,318 and 6,061 metric tons per year.

2.9.2. Effect of the Take

In the biological opinion, NMFS determined that the amount or extent of anticipated take, coupled with other effects of the proposed action, is not likely to result in jeopardy to the species. 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 measures that are necessary or appropriate to minimize the impact of the amount or extent of incidental take (50 CFR 402.02).

1. EPA and USIBWC shall monitor, document, and report the extent of incidental take of ESA-listed species resulting from several components of the proposed action (e.g., expansion of ITP and construct/operate APTP) using the surrogates described in Section 2.9.1 of this biological opinion.

2. EPA and USIBWC shall notify NMFS regarding the observation of white abalone on the SBOO structure and coordinate with NMFS and UCD-BML regarding the removal and transfer of the white abalone from the SBOO structure to an approved captive holding facility under ESA Permit 14344-2R, issued to the UCD-BML.

[3] The following was used to convert 57,400 ug/L to pounds per day using the assumed range of discharge flow rates:[ug/l]* [mg/L/1000]*[MGD]*[8.34 pounds/gallon]. For converting pounds to metric tons, we used a 0.000454 pounds to metric tons conversion factor.

2.9.4. Terms and Conditions

In order to be exempt from the prohibitions of section 9 of the ESA, the Federal action agencies must comply with the following terms and conditions. The EPA and USIBWC have 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 agencies do 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 and USIBWC shall ensure that the necessary data are collected to determine mass emission levels of organophosphate and PBDE flame retardants in the ITP's and the proposed APTP's effluent using sampling and analysis protocols that are consistent with the E.W. Blom Point Loma Wastewater Treatment Plant as identified in Addendum No 1 to Order No. R9-2017-007 NPDES No. CA0107409, Table E-4a. Flame Retardant Monitoring. EPA and USIBWC shall estimate the loading of organophosphate and PBDE flame retardants (pounds or kg) from the ITP and proposed APTP into the action area per year, using the monitored concentrations and effluent discharge (flow) rates. Data collection shall occur in multiple phases: before ITP expansion and at least once every five years after ITP expansion. Monitoring of APTP effluent will begin following completion of APTP construction. The frequency of sampling will be based on a monitoring plan to be developed by EPA and USIBWC, subject to agreement by NMFS. These analyses will continue for the defined period of the proposed action (i.e., until 2050) unless sufficient information is developed that NMFS agrees to alter this monitoring schedule at some time in the future.
>
> 1b. EPA and USIBWC shall monitor the concentration of the various forms of nitrogen (i.e., ammonia, total nitrogen, total organic nitrogen, nitrate (as N), nitrite (as N)) in the effluent from the ITP before the proposed ITP expansion is operational and after the expanded ITP is operational, and after the proposed construction of APTP. While the extent of nitrogen loading can be approximated by monitoring ammonia as described in section 2.9.1 (Amount or Extent of Take), monitoring all nitrogen forms is needed to better understand the nitrogen characteristics of the discharge and the nitrogen loading that results from the ITP's and proposed APTP's discharge into the action area. These monitoring results would:
>
> - Improve understanding of the proposed action's contribution to nutrient loading and HABs in the action area.
> - Assist efforts by the EPA, San Diego Regional Water Quality Control Board and the dischargers to investigate if there is a need for future studies or other feasible efforts to minimize the discharge of nutrients that may increase the probability of HAB occurrence in the action area during future NPDES permit actions.
> - Validate the analysis made by EPA and USIBWC in their Biological Assessment.

Therefore, the EPA and USIBWC shall monitor the concentration of the various forms of nitrogen (ammonia, total nitrogen, total organic nitrogen, nitrate (as N), nitrite (as N)) in the effluent from the ITP and proposed APTP at appropriate frequencies to characterize the discharge:

- Following ITP expansion (when it becomes operational) and before APTP construction, monitoring shall be conducted on a similar schedule as the E.W. Blom Point Loma Wastewater Treatment Plant as identified in Addendum No 1 to Order No. R9-2017-007 NPDES No. CA0107409 for the Phytoplankton Stimulation Study described therein (i.e., weekly basis initially for a period of 6 months, reducing to once per month if the percent variance from the median concentration is 30% or less). This pattern will be repeated following construction of the APTP when it becomes operational in order to determine nitrogen loading levels and patterns.
- Nitrogen monitoring for ITP effluent prior to ITP expansion and APTP construction (aka "baseline monitoring") will be conducted for all of the nitrogen forms noted above. EPA and USIBWC will prepare a baseline monitoring proposal, for NMFS review and approval that will generate sufficient data to establish a reliable baseline estimate of nitrogen concentration in the effluent.
- These analyses will continue for the defined period of the proposed action (i.e., until 2050) unless sufficient information is developed that NMFS agrees to alter this monitoring schedule at some time in the future.

The EPA and USIBWC will use the data collected to 1) calculate a mass emission rate that accounts for and characterizes all forms of nitrogen noted above in the effluent, and 2) calculate the loading of the various forms of nitrogen (pounds/tons or kg/metric tons) discharged by the ITP and proposed APTP into the action area per year.

Also, the EPA and USIBWC shall ensure that nitrogen forms in receiving waters are measured to the extent feasible (e.g., ammonia and total nitrogen measured once/quarter in offshore and kelp/nearshore stations) and, at minimum consistent with the ITP's existing NPDES permit. At USIBWC's discretion, these efforts may be coupled with existing monitoring of the action area and the SBOO discharge. These results will produce a more consistent and robust dataset that can be used in regional efforts (e.g., the Southern California Bight Regional Monitoring Program and the BEC-ROMS model).

1c**.** EPA and USIBWC shall provide NMFS WCR a report of the estimated annual discharge of organophosphate and PBDE flame retardants and estimated annual discharge of nitrogen in the effluent from ITP and the proposed APTP. If any of these estimated annual discharge amounts exceed the total amounts/levels that have been anticipated described above, EPA and USIBWC will provide an explanation of why this has occurred and any steps that may be taken by USIBWC to reduce or mitigate for this condition. NMFS, EPA and USIBWC will schedule a meeting to discuss the report and potential steps to address the exceedance(s).

The report shall be submitted to the NMFS WCR Protected Resources Division's Long Beach Office Branch Chief (Dan Lawson) at the following addresses:

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

1d. EPA and USIBWC shall ensure that the monitoring studies described above are submitted to NMFS WCR and the San Diego Regional Water Quality Control Board within 90 days of completion and/or within 180 days of the next relevant NPDES permit reissuance associated with each stage of the proposed action.

The following terms and conditions implement RPM 2:

2a. EPA and USIBWC shall ensure completion of a survey of the SBOO wye diffuser for the presence of white abalone, by a method presented to and agreed upon by NMFS, prior to recommissioning of the diffuser risers during two stages of the proposed action: 1) before the expanded ITP enters operations, and 2) before the APTP enters operations. If any white abalone are observed on the SBOO wye diffuser, EPA and USIBWC shall leave the abalone in place and notify NMFS, providing sufficient information about the abalone's location to relocate it.

2b. EPA and USIBWC shall ensure coordination with NMFS regarding the removal of the white abalone from the SBOO structure. EPA and USIBWC shall ensure that NMFS, UCD-BML, and other personnel authorized under ESA Permit 14344-2R are able to collect the white abalone. Only experienced personnel authorized under ESA Permit 14344-2R may remove and handle the white abalone

2.10. Conservation Recommendations

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

Contaminants of Emerging Concern (CECs)

Effluent discharged from WWTPs can be a major source of CECs to the receiving waters, and it is unknown what constituents and levels of CECs may be associated with the previously untreated transboundary wastewater flows. The following conservation recommendation related to CECs would provide information for future consultations and address questions related to the effects of the proposed action's discharge on the frequency and extent of CECs in the action area and SCB.

1. EPA and USIBWC will develop a workplan to initiate special studies, or explore opportunities to contribute to existing studies, measuring the discharges of CECs resulting from the proposed action in and around the action area or of discharges with similar treatment types as that of the proposed action. Such a workplan or existing studies may address the same set of CEC compounds as measured by other WWTPs in the Southern California Bight (e.g., Orange County, Hyperion). CECs include:

- 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; and
- Nanomaterials such as carbon nanotubes or nano-scale particulate titanium dioxide, of which little is known about either their environmental fate or effects.

Monitoring methods can be implemented similarly to other WWTP permits (e.g., Orange County). Research objectives targeted through this conservation recommendation should include constituents and levels of CECs that may be associated with wastewater flows from facilities with treatment types similar to those of the ITP and proposed APTP (i.e., secondary treatment and advanced primary treatment). This effort can be coupled with other "before and after "monitoring that may be required by EPA or the State of California for other CECs, as EPA and USIBWC determine appropriate

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 the proposed action's discharge on the frequency and extent of HABs in the action area and SCB.

1. EPA and USIBWC should support additional data collection within the action area and the SCB to help understand the potential influence on harmful algal bloom dynamics from the proposed action's discharge. This could include:

    a. Studying the generation of nitrogen form, timing, and mass balance data from upwelling and stormwater runoff events in the coastal areas between La Jolla and Baja California to couple with the required generation of nitrogen data from this proposed action's discharge and feed into regional modeling efforts (e.g., Southern California Bight Regional Monitoring Program).

b. Assessing what HAB species are in the coastal areas between La Jolla and Baja California. This may include if the HAB species are being maintained within a subsurface zone; and if 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 USIBWC or the City of San Diego by SBOO related monitoring efforts or through the SCCWRP led Regional Bight Monitoring Program that examines multiple WWTPs within the Southern California Bight.

c. Synthesizing the results from additional data collection, monitoring and/or evaluation can be provided to NMFS in a report or reports, submitted on a schedule to be determined.

d. Having the EPA and USIBWC explore other means of reducing effluent associated nitrogen (as well as biological or biochemical oxygen demand, flame retardant and other CEC inputs) to the action area from the proposed ITP expansion and potential APTP and other future projects mentioned in EPA's Biological Assessment document. This could include projects to further reduce nitrogen loading to the receiving waters and potential impacts related to HABs.

Reduction of Contaminant Loading

The following conservation recommendations related to contaminant loading and associated environmental degradation in the action area would provide information showing the benefits of the proposed action and potentially make it easier for EPA and USIBWC to acquire future funding for related actions necessary to alleviate effects due to growth in the region as well as provide information for future evaluations.

1. EPA and USIBWC should assess and report on the beneficial impacts of the proposed action associated with the reduction in untreated wastewater discharge in the action area. This can include estimates of the untreated transboundary flows that have been reduced, and the concurrent reductions in biological or chemical oxygen demand, suspended sediments, nutrients, heavy metals, pathogens, POPs, CECs, and other contaminants or detrimental environmental conditions (e.g. depressed dissolved oxygen levels) that have been achieved.

2.11. Reinitiation of Consultation

This concludes formal consultation for issuance of U.S. appropriations under the U.S.–Mexico–Canada Agreement (USMCA) Implementation Act.

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

In this biological opinion, we describe the extent of take of the proposed action in terms of the amount of potentially harmful contaminants discharged in the effluent by ITP and APTP through the SBOO as a consequence of the proposed action; specifically the total loading of organophosphate flame retardants. We estimated that the proposed action of ITP expansion will discharge approximately 267 metric tons of organophosphate flame retardants (TCEP, TCPP, and TDCPP combined) into the action area each year. Following contruction of APTP, we estimate approximately 340 metric tons of organophophates of discharge into the action area each year. If the discharge of these organophosphate flame retardants per year by ITP and APTP through the SBOO is determined to be greater than these estimates (through monitoring by the EPA and USIBWC 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 (via ammonia and/or total nutrients) discharged in the effluent by ITP and APTP through the SBOO. We estimate that the proposed discharge by ITP through the SBOO (following ITP expansion) will include up to 4,760 metric tons per year of ammonia into the action area per year. Following construction of APTP, we estimate between 4,318 and 6,061 metric tons of ammonia per year discharged through the SBOO. If the proposed discharge of ammonia by ITP and APTP through the SBOO per year is determined to be greater than these estimates measured at the appropriate time periods (through monitoring by the EPA and USIBWC 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 the proposed action'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, gulf grouper, oceanic whitetip sharks, East Pacific DPS scalloped hammerhead sharks, Southern California DPS steelhead, or black abalone. This opinion does not consider effects on critical habitat because none is designated in the action area.

In our effects analysis, we identified three potential stressors to result from the proposed discharge of effluent by ITP and APTP through the SBOO: (1) uptake of pollutants from the water; (2) ingestion of prey that have accumulated pollutants; and (3) exposure to harmful algal blooms resulting from the discharge. In this section, we analyze each species, or species group, as applicable relative to all of these potential stressors.

2.12.1. Other ESA listed marine mammals

2.12.1.1 North Pacific Right Whales

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

2.12.1.2 Sei Whales

Similarly, no sei whales have been sighted during dedicated marine mammal surveys by NMFS in Southern California since 1991 (Carretta et al. 2022). Therefore, due to their rarity, we do not expect these whales to be exposed to any stressors deriving from the proposed action.

2.12.1.3 Sperm Whales

Sperm whales reach peak abundance in waters offshore California from April through mid-June, and from the end of August through mid-November (Carretta et al. 2022). Sperm whales are typically found foraging offshore in deep waters and/or canyons and are more commonly sighted off central California. Sperm whales primarily prey on medium and large-sized squid (e.g., the giant squid) and fishes (e.g., sharks) that occupy deep ocean waters. Sperm whale occurrence in nearshore waters of Southern California is rare; no sperm whales have been sighted during dedicated marine mammal surveys by NMFS in Southern California since 1991 (Carretta et al. 2022). Although the action area does include some areas of deep water canyons, sperm whales are very unlikely to forage in the action area and none of their probable prey would be directly connected to the food web in nearshore waters that are most likely to be impacted by wastewater discharge. As a result, we do not expect these whales to be exposed to any stressors deriving from the proposed action.

2.12.1.4 Conclusion for other ESA-listed marine mammals

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

2.12.2. Giant manta ray

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

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

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

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

2.12.3. Green sturgeon, Southern DPS

Green sturgeon are anadromous, long-lived, bottom-oriented (demersal) fish that range from the Bering Sea, Alaska (Colway and Stevenson 2007) to El Socorro, Baja California, Mexico (Rosales-Casián and Almeda-Jáuregui 2009). They are one of the most marine-oriented and widely distributed of the sturgeon, spending much of their lives migrating between estuaries along the coast. Relatively little is known about how green sturgeon use coastal marine habitats. While in the ocean, they generally occur between 0 and 110 meters, spending most of their time in areas between 20 to 80 meters in depth (Erickson and Hightower 2007; Huff et al. 2011). While in marine waters, they may be feeding or simply migrating between estuaries.

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

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

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

2.12.4. Gulf grouper

NMFS listed the gulf grouper as endangered under the ESA in 2016 (81 FR 72545, October 20, 2016). Gulf groupers occur in the subtropical eastern Pacific Ocean and Gulf of California from La Jolla, California, to Mazatlán, Sinaloa in Mexico (Heemstra and Randall 1993). Gulf groupers typically inhabit reefs and seamounts in waters up to 30 to 45 m deep (Heemstra and Randall 1993; Sala et al. 2003; Moreno-Báez 2010). The primary factor in the decline of gulf groupers has been direct harvest. Gulf groupers are particularly susceptible to fishing pressure because they are long-lived, late-maturing, and protogynous hermaphrodites (maturing as females and later transitioning into males; Dennis 2015). They also aggregate to spawn at known times and locations. Other threats to gulf grouper include indirect harvest (as bycatch), habitat loss and degradation, and climate change effects on coral ecosystems (Dennis 2015).

Recent observations of gulf groupers outside of the Gulf of California are rare and limited to a few observations in Bahía Magdalena (pers. comm. with Octavio Aburto-Oropeza, Scripps Institution of Oceanography, on August 29, 2014, cited in Dennis 2015). Gulf groupers were observed off the San Diego coast in the 1930s (Hubbs 1948), with no records since then (Dennis 2015).

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

2.12.5. Oceanic whitetip shark

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

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

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

2.12.6. Scalloped hammerhead shark, Eastern Pacific DPS

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

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

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

2.12.7. Steelhead, Southern California DPS

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

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

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

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

Given their location in the very upper part of the water column steelhead would have limited exposure to the discharge plume. Because the SBOO 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 associated with the proposed action expected within the ZID. Any exposure of steelhead to the wastewater effluent would be limited to low concentrations and short duration, and would not be expected to result in lethal or sub-lethal effects. Also, the potential exposure duration would most likely be on the order of minutes rather than days based on steelhead swimming speed and distribution of the effluent plume, further reducing the likelihood of lethal or sub-lethal effects on steelhead.

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

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

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

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

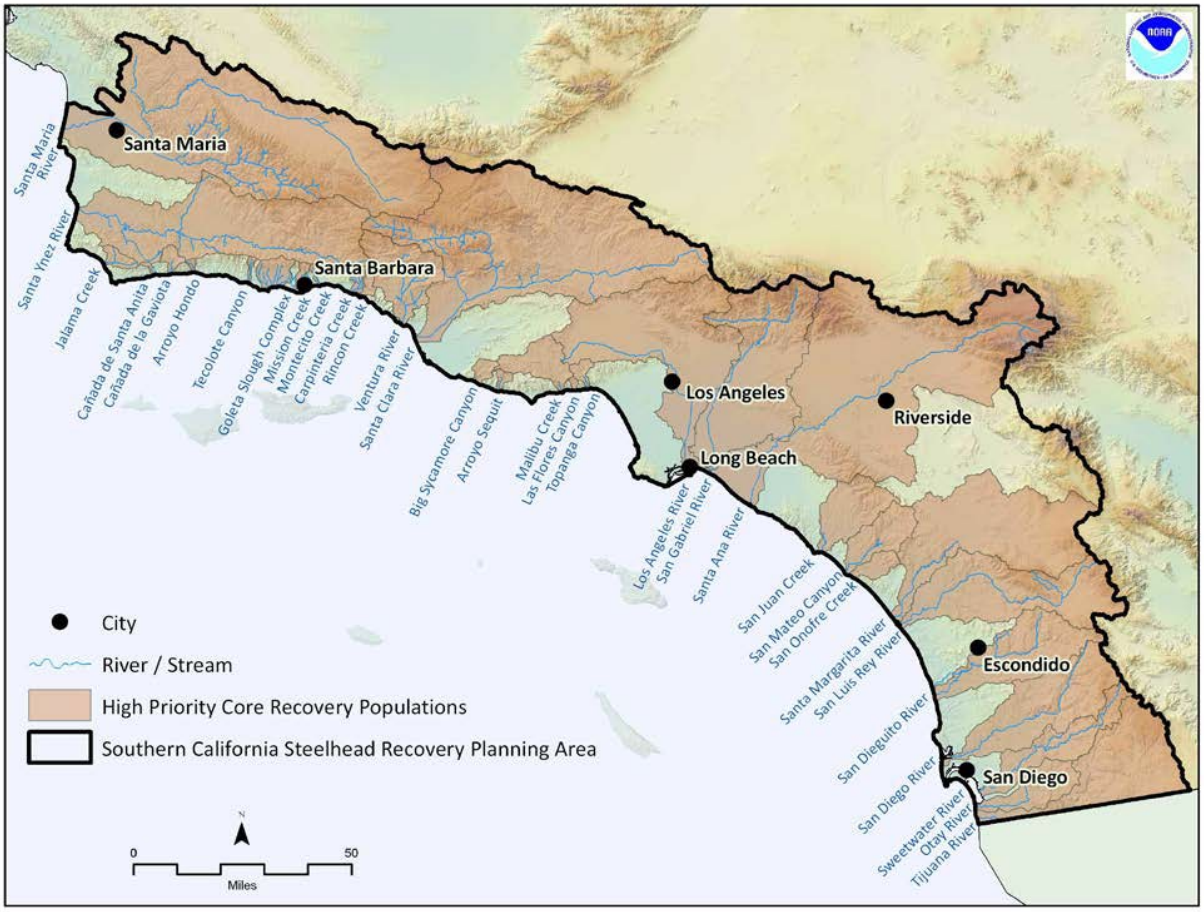

Figure 6. Southern California Steelhead Recovery Planning Area; maps shows the rivers and streams within the range of Southern California Steelhead (Figure 1-1 from Southern California Steelhead Recovery Plan (NMFS 2012a)).

2.12.8. Black abalone

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

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

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

Even if black abalone were present in the action area, we expect any exposure of black abalone and their habitat to be limited to very diluted concentrations of the discharge plume associated with the proposed action, given the distance of suitable habitat from the SBOO discharge point. Based on this, if black abalone were to be exposed to the plume, we expect the exposure to be minimal and unlikely to result in adverse effects.

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

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

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

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

3.1. Essential Fish Habitat Affected by the Project

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

3.2. Adverse Effects on Essential Fish Habitat

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

In the following sections, we evaluate the adverse effects on EFH from pollutants discharged from the proposed effluent by ITP and APTP through the SBOO and discuss ongoing studies to better understand those effects and efforts to address them, such as compliance with existing water quality standards. Much of the information used in this effects analysis was taken from EFH Assessments provided by the EPA and USIBWC for this and other similar wastewater treatment projects.

3.2.1. Metals

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

3.2.2. Toxicity

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

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. Under the Ocean Plan, chronic toxicity testing is used to measure Toxic Units Chronic (TUc) and relies on the No Observed Effect Limit. Under the NPDES permit for ITP (Order No. R9-2014-0009[4]) weekly monitoring of an effluent limit of 95.6 Toxic Units Chronic (TUc) was required (California Regional Water Quality Control Board 2021). Since it was reported in March 2017 that the maximum effluent chronic toxicity value was 200 TUc, it was determined that the ITP effluent has reasonable potential to cause an exceedance of the narrative water quality objective for chronic toxicity (i.e., Endpoint 1: effluent limitation and monitoring are required as refered to in the Ocean Plan). Therefore, this Order retains an effluent limit for chronic toxicity and weekly monitoring for ITP.

[4] Acute testing was not conducted nor was it required under this NPDES permit since chronic testing has a more stringent requirement.

3.2.3. Nutrients and HABs

As described above in Section 2.4.1.1.4 (Harmful Algal Blooms, in the Environmental Baseline) and Section 2.5.4 (Harmful Algal Blooms, in the Effects Analysis), nutrient loading can cause increased plant and algal growth leading to eutrophication and increased instances of HABs.

As noted previously, *P. spp*. are domoic acid-producing diatoms that are the most frequently noted HAB species in the action area. Domoic acid is a water soluble neurotoxin that accumulates in shellfish and planktivorous fish such as anchovy and sardine (Lefebvre et al. 2007, 2012; Smith et al. 2018). Although the effects on piscivorous birds and marine mammals are well documented and wide spread (Schnetzer et al. 2013; Smith et al. 2018), impacts on other species are less certain. Behavioral or schooling impacts on fish species are not believed to be occurring and laboratory work has shown that fish species ingesting domoic-acid-producing phytoplankton seem to be able to isolate and eventually depurate the compound (Lefebvre et al. 2007, 2012). It is unknown if there is a metabolic cost to this process for the fish. When domoic acid was microinjected into zebrafish eggs, the study did not find any effects on egg hatching and development (Vasconcelos et al. 2010). However, we could not readily find any studies reporting effects on fish egg or larval development under realistic exposure scenarios in our literature search. Vasconcelos et al. (2010) also reported that most studies showed little or no effects to survival or reproduction to several species of mollusks and crustaceans. Yet Liu et al. (2007) found significantly compromised growth and survival of king scallop larvae at environmentally realistic exposures to domoic acid. Further research on potential effects to wildlife under realistic environmental concentrations and conditions seems warranted.

The subsurface, HAB-prone dinoflagellate *A. 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 among brine shrimp to sublethal effects on crab larvae. In addition to numerous study references indicating toxicity to winter flounder, red sea bream and Japanese anchovy, Gosselin et al (1989) found heavy mortality among capelin and Atlantic herring larvae and juveniles exposed to environmentally realistic concentrations of *A. tamarense* complex species through both vectorial poisoning and direct intoxication. Lefebvre et al. (2004) conducted a dietary uptake experiment with zebrafish larvae. They found that 24 hour exposures to high levels of saxitoxin induced paralysis had morphological impacts. Reduced growth rates were also prevalent resulting in depressed cumulative survival compared to control fish (40% vs 80%). A follow-up study with dissolved saxitoxin by Lefebvre et al. (2005) was conducted on larval Pacific herring. At levels greater than 47 µg/L, the saxitoxin caused significant reductions in sensorimotor function within one hour. Interestingly, the effect was transient, lasting only a few hours indicating there may be significant variability in species effects. However, impacts on the early life stages of Pacific herring are still likely due to an inability to avoid predators or escape the hypoxic conditions caused by significant HABs. There is a paucity of data on extracellular saxitoxin concentrations (Lefebvre et al. 2008), but the patchiness of HABs and the highly variable release of toxins when the cells lyse could produce localized high levels in the field.

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

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

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

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

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

The proposed action will reduce overall nutrient loadings to the action area, but will increase the nutrient loadings discharged by ITP and APTP through the SBOO by approximately 8 to 12 percent. Coastally trapped raw effluent presently discharged from SAB Creek and the TJRE contribute to an increased likelihood of HAB events. The proposed action seeks to significantly reduce this pathway by treating the SABTP effluent at the ITP and reducing the overall nitrogen loading of coastal waters in the action area. While contributions of excessive nutrients, such as nitrogen, increase HAB occurrence in the action area, the proposed action is expected to result in a net reduction in nutrient loadings. Therefore, this project is expected to result in an overall reduction in HAB events and will improve water quality that benefits EFH in the project area.

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

Aquatic organisms require sufficient levels of dissolved oxygen to breathe and grow. As a result, dissolved oxygen is an important measure of water quality and an indicator of a water body's ability to support aquatic life (see previous Section 3.2.3 for information on potential effects from HABs and nutrients on dissolved oxygen). Biological oxygen demand and total suspended solids both affect the level of oxygen in a receiving water, either directly or indirectly. Oxygen is depleted more rapidly with higher biological oxygen demand. Although the impact mechanisms are more indirect with respect to total suspended solids (e.g., decreased photosynthesis resulting from reduced light; increased water temperature, which holds less oxygen; etc.), elevated total suspended solids can also reduce dissolved oxygen levels. Based on EPA and USIBWC's estimated annual average pollutant concentrations within the SBOO mixing zone, biological oxygen demand is expected to increase (between 15 to 57% depending on distance from the SBOO) by 2050 compared to the current condition as a result of the proposed action. However, biological oxygen demand will simultaneously be reduced in the nearshore area. Both total dissolved and suspended solids are expected to decrease between 5 to 33% by 2050 in the action area compared to its current condition.

3.2.5. Contaminants of Emerging Concern and Persistent Organic Pollutants

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

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

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

CECs currently have no Clean Water Act regulatory standard (e.g., no established water quality standards and/or notification levels). However, the California SWRCB and Regional Water Quality Control Boards have identified monitoring strategies and sampling plans for CECs. The City (2021d) discusses POPs and other CECs and notes that effluent results for many POPs, including PCBs, chlorinated pesticides, dioxins, and furans are at levels below detection limits and below the applicable numeric effluent limits as well as California Ocean Plan water quality objectives.

There are no routine monitoring data for other POPs and CECs to evaluate, though CECs have been detected previously in the SBOO discharge. However, special studies evaluating possible effects of CECs on fish exposed to the SBOO effluent have included one-time surveys of the discharge for representative compounds in the CEC category, including pharmaceuticals, personal care products, hormones, current use pesticides, and industrial/commercial products. Additionally, it is uncertain that discharge would result in any other effects beyond the ZID. An extensive environmental monitoring program conducted by the City of San Diego on the ongoing SBOO discharge has not identified any effects to EFH so far, including the Imperial Beach Kelp Forest (City of San Diego 2013, 2016b, 2018, 2020a, 2021c, 2022b). However, the current monitoring program may be unable to detect the effects of CECs.

The use of PBDEs is declining, and the 2018 Bight Regional Monitoring results showed non-detect or low levels of PBDEs in coastal sediment samples (EPA 2021). However, because of persistent properties of these PBDEs, these contaminants remain in the ecosystem and continue to impact marine species. Moreover, as mentioned previously, PBDEs are being replaced by organophosphate flame retardants that can have similar adverse effects on fish and have the potential to biomagnify in a marine food web (Siddiqi et al. 2003; Bekele et al. 2019). Organophosphate flame retardants were not included in previous ITP or SBOO monitoring efforts, but have been detected in the effluent for two other facilities in the SCB (Hyperion and TIWRP; LASAN 2020). Given the potential adverse effects to EFH, NMFS is requesting that measurements of organophosphate flame retardants and PBDEs in the effluent and the loading of receiving waters occur as described under the terms and conditions of this biological opinion.

3.2.6. Sediment Contamination

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

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

Despite these legacy contaminant issues, benthic habitat condition in the action area has improved substantially. For instance, the 2018 Bight Study concluded that sediment toxicity across the Bight was low, with a continued trend of decreasing toxicity and continued improvement in sediment quality (with the exception of marinas; SCCWRP 2020) including in the vicinity of the SBOO. The Benthic Response Index results reveal little evidence of disturbance, and the EFH Assessment notes a balanced indigenous population of shellfish, fish, and wildlife exists immediately beyond the ZID. Other major community metrics (species richness, macrofaunal abundance, diversity, evenness, and dominance) show no evidence of impact or habitat degradation. Moreover, the results of all sediment toxicity testing conducted for the SBOO region during a 3-year pilot study (2016-2018), and during follow-up monitoring in the summer of 2019, did not indicate any evidence of toxicity at any of the monitoring sites. The proposed action is expected to treat effluent from its growing service region, as well as address failing infrastructure that is discharging untreated effluent just south of the international border in the near shore area. This should allow for the continuation of unimpaired sediment conditions in the action area

3.2.7. Effects of In-water Construction Activities for Recommisioning

Most recommissioning construction activities under the proposed action will occur inland at least 2 miles from the coastline and will have no direct or indirect effect on the marine environment. However, it is anticipated that proposed construction work on diffuser ports will result in minor disturbance to marine habitat. Diving activity will occur nearby ballast rock reef, and other SBOO structures not constituted as EFH. Potential disturbance of surrounding groundfish species caused by anchor deployment is expected to be minimal and have negligible impact. Boat operators will minimize adverse effects to groundfish EFH by minimizing anchor size and decreasing the number of anchor deployments. There are potential risks of grounding or oil spills from vessel activities; the PEIS recommends safety procedures to be conducted by vessel captains in the instance of vessel grounding or an oil spill.

3.2.8. Impacts on Sensitive Habitats

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

Kelp beds are present in the action area located approximately 15 km to the north of the SBOO, and maintain kelp canopy at some point in the year every year. However, many kelp beds are intermittent year-to-year, therefore multiple years of observations are necessary to determine areas where kelp canopy is likely to form over the life cycle of a project. Research and ongoing monitoring, including quarterly mapping of kelp beds in the area in accordance with regulations promulgated by the San Diego Regional Water Quality Control Board, conducted at the Point Loma kelp bed, has not found any impacts associated with the SBOO discharge. It is uncertain whether the effluent discharge will result in any other effects beyond the ZID. An extensive environmental monitoring program conducted by the City of San Diego on the ongoing SBOO discharge has, to date, not identified any effects to EFH, including the Imperial Beach Kelp Forest (City of San Diego 2013, 2016b, 2018, 2020a, 2021c, 2022b). However, the proposed action intends on reducting pollution events in the action area and would therefore result in a net beneficial effect to kelp canopy habitat such that any expected adverse effects would be minimal.

Eelgrass habitat does exist within shallower regions near Point Loma, such as within the entrance channels to Mission and San Diego Bays, and east of Zuniga Jetty just offshore of Breakers Beach (Figure 7). Both the Zuniga Jetty seagrass bed and the smaller beds within the entrance channel to San Diego Bay are on the perimeter of the action area beyond the farthest stations at which SBOO plume detections have occurred. Monitoring data rarely detect a potential plume at the most nearshore stations in the action area, which are located farther offshore and in deeper water than the eelgrass beds. Therefore, we expect any adverse effects on eelgrass from the proposed project would be minimal.

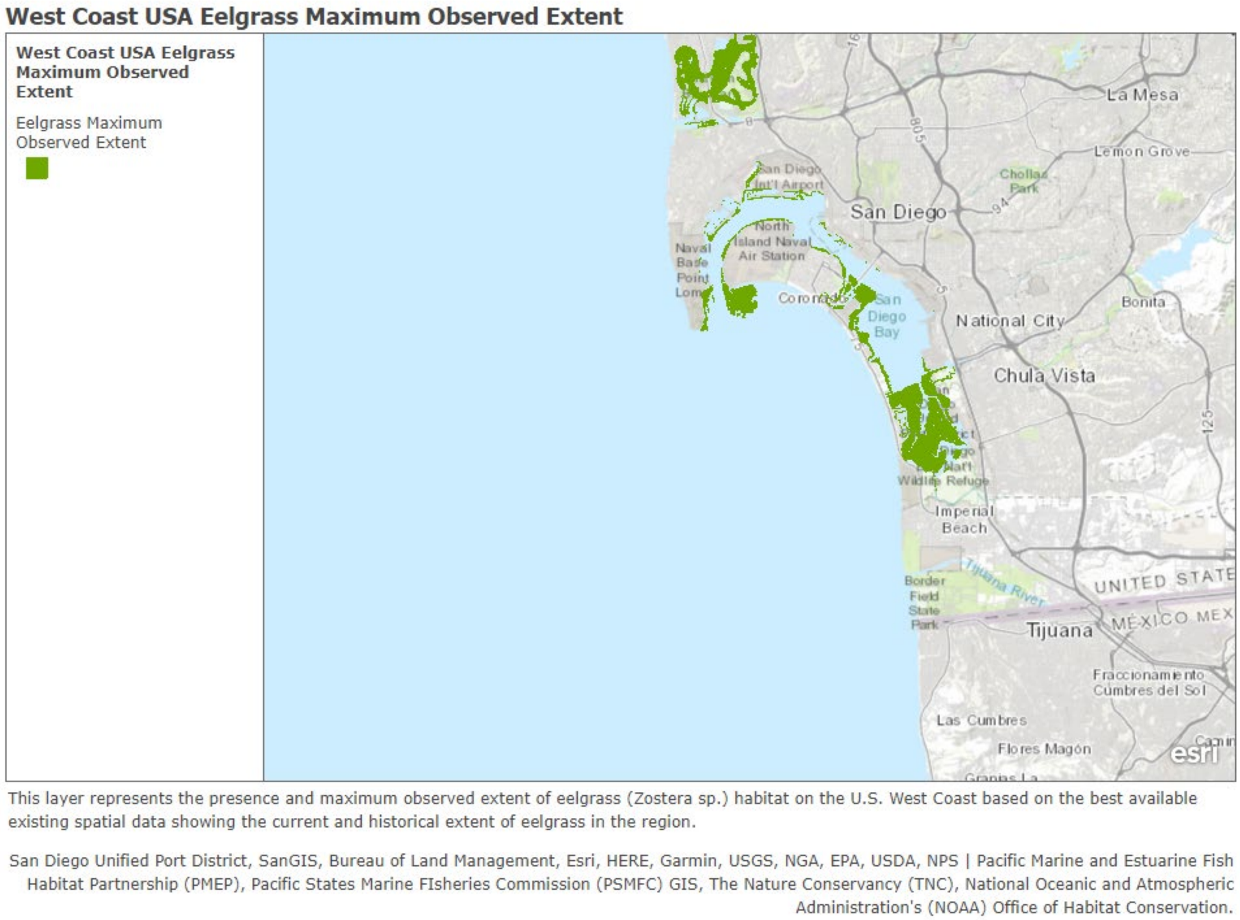

Figure 7 Map of maximum observed eelgrass extent within the action area

The Tijuana River Estuary system is highly variable, depending on seasonal flows, and provides biological ecosystem servies related to EFH HAPC. Based on analysis of a long-term monitoring dataset from the estuary, Desmond et al. (2002) found that seasonal changes in water temperature most strongly correlated with changes in fish assemblage observed in the estuary, but discharge was also important. Peak abundance was in summer/fall when discharge was low. Interannual trends showed that periods of increased sewage input affected fish assemblage with more rapidly maturing fish (e.g., arrow goby) being more dominant under increased sewage inputs (Desmond et al., 2002). A sharp reduction in nearshore pollution between SAB Creek and the TJRE, and the elimination of the accumulation of plume in the retention zone expected to result from the proposed action is clear in the models ran by Feddersen et al. (2021). The relative magnitude of reduction in nearshore pollution (between SAB Creek and the TJRE) due to the implementation of the proposed action is greater than the increase in the discharge of treated effluent from the SBOO. Therefore, the net effect of the proposed action is a positive impact on the marine environment and the species and habitats that reside.

3.2.9. Cumulative Impacts

Wastewater discharges have historically been one of the largest contributors to pollutants in coastal waters, and contaminants released into the action area from the SBOO and other permitted discharges (e.g., PLOO) result in cumulative impacts on EFH. However, the EFH Assessment notes that mass emissions of most constituents in wastewater discharges have been substantially reduced coincident with increasingly stringent requirements over the last 40 years.

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

Additionally, EFH throughout the action area will gain a net benefit from reductions in nearshore pollution because of the proposed action to fund infrastructure to address transboundary flows. However, the proposed action will result in an increase in effluent water from the SBOO, resulting in an increase in the ZID and an extension of the detectable extent of the plume. When considered in isolation from the net benefits of the proposed action described above, this increase in the discharge of effluent at the SBOO would result in adverse effects to EFH in the ZID. EFH for Pacific Coast Groudnfish, CPS, and HMS overlap the ZID. These EFH would therefore be affected by the increased discharge volume of toxic pollutants and subsequent expansion of the ZID due to the proposed action. There remains uncertainty as to whether, and to what degree, EFH located outside of the ZID would be affected by the discharge.

3.3. EFH Adverse Effects Determination

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

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

The proposed action includes measures to minimize adverse effects to EFH and HAPC caused by the in-water construction activities. Due to the volume of raw wasterwater treatment and increased capacity of the ITP and contruction of APTP under the proposed action, our analysis concludes an overall beneficial impact to EFH. We also conclude that any adverse effects would be minimal and temporary.

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

3.4. Supplemental Consultation

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

4. Data Quality Act Documentation and Pre-Dissemination Review

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

4.1. Utility

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

4.2. Integrity

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

4.3. Objectivity

Information Product Category: Natural Resource Plan

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

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

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

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

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