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

Re-Issuance of a permit to the City of Los Angeles for wastewater discharge by the Hyperion Treatment Plant under the National Pollutant Discharge Elimination System (NPDES)

NMFS Consultation Number: WCR-2017-6428

Action Agencies: U.S. Environmental Protection Agency

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?
Southern California steelhead (*Oncorhynchus mykiss*)	Endangered	No		No	
green sturgeon; Southern distinct population segment (DPS) (*Acipenser medirostris*)	Threatened	No		No	
Scalloped hammerhead shark; Eastern Pacific DPS (*Sphyrna lewini*)	Endangered	No		No	
Blue whale (*Balaenoptera musculus*)	Endangered	Yes	No	No	
Fin whale (*Balaenoptera physalus*)	Endangered	Yes	No	No	
Humpback whale; Mexico DPS (*Megaptera novaeangliae*)	Threatened	Yes	No	No	
Humpback whale; Central America DPS	Endangered	Yes	No	No	
Gray whale; Western North Pacific population(*Eschrichtius robustus*)	Endangered	Yes	No	No	

Guadalupe fur seal (*Arctocephalus townsendi*)	Threatened	Yes	No	No	
Green sea turtle; East Pacific DPS (*Chelonia mydas*)	Threatened	Yes	No	No	
Leatherback sea turtle (*Dermochelys coriacea*)	Endangered	Yes	No	No	
Loggerhead sea turtle; North Pacific Ocean DPS (*Caretta caretta*)	Endangered	Yes	No	No	
Olive ridley sea turtle (*Lepidochelys olivacea*)	Endangered	Yes	No	No	
White abalone (*Haliotis sorenseni*)	Endangered	Yes	No	No	
Black abalone (*Haliotis cracherodii*)	Endangered	Yes	No	No	

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

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

Issued By:

FOR Barry A. Thom
Regional Administrator

Date: April 10, 2018

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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 U.S. Environmental Protection Agency (EPA) regulates the discharge of wastewater into all surface waters under the federal Clean Water Act (33 U.S.C. §1251 et seq.). Permits for wastewater discharge are issued either by the EPA or the states under the National Pollutant Discharge Elimination System (NPDES). When the discharge of wastewater occurs in federal waters, EPA retains the authority to issue NPDES permits to those dischargers. The City of Los Angeles has been discharging wastewater into Santa Monica Bay (referred to as “the Bay”) since the late 1800’s at the Hyperion Treatment Plant (Hyperion), and currently Hyperion’s primary discharge occurs approximately 5 miles offshore into federal waters. As a result, EPA must evaluate and permit the discharge of wastewater by the City of Los Angeles at Hyperion.

The National Marine Fisheries Service (NMFS) prepared the biological opinion (opinion) and incidental take statement (ITS) portions of this document in accordance with section 7(b) of the Endangered Species Act (ESA) of 1973 (16 USC 1531 et seq.), and implementing regulations at 50 CFR 402. We also completed an essential fish habitat (EFH) consultation on the proposed action, in accordance with section 305(b)(2) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA) (16 U.S.C. 1801 et seq.) and implementing regulations at 50 CFR 600.

We completed pre-dissemination review of this document using standards for utility, integrity, and objectivity in compliance with applicable guidelines issued under the Data Quality Act (DQA) (section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001, Public Law 106-554). The document will be available through NMFS’ Public Consultation Tracking System [https://pcts.nmfs.noaa.gov/pcts-web/homepage.pcts]. A complete record of this consultation is on file at NMFS West Coast Region Long Beach office.

1.2 Consultation History

On September 2, 2016, NMFS West Coast Regional Office (WCR) received a letter from the EPA requesting informal Section 7 consultation under the Endangered Species Act (ESA) and concurrence regarding its conclusion that the reissuance of a NPDES permit (CA 00109991) to Hyperion is not likely to adversely affect a host of species listed as threatened or endangered under the ESA, along with a Biological Evaluation (BE) of the impacts of wastewater discharge from Hyperion into the Bay. An Essential Fish Habitat (EFH) Assessment was also included with the BE that determined the project may have minimal adverse effect or less than substantial adverse effect on EFH for federally managed species within the Coastal Pelagic Species, Pacific Coast Groundfish, and Highly Migratory Species Fishery Management Plans. Subsequently, information was exchanged between NMFS and EPA staff, including conference calls held on November 22, 2016, and December 1, 2016, to discuss outstanding questions related to the consultation requests.

On December 9, 2016, we responded with a letter informing EPA that we are not able to concur with the effect determinations made by EPA regarding the proposed issuance of this NPDES permit to Hyperion at that time, and that there were a number of outstanding questions and concerns that needed additional explanation or needed to be addressed before we could conclude consultation. Included in the letter were specific questions and highlighted issues that needed further attention. In particular, we reiterated the need for a more complete understanding of the monitoring program and data analysis, the need for additional analysis of the potential impacts associated with exposure and uptake of persistent organic pollutants (POPs) by marine species and habitats (referred to herein as "bioaccumulation") resulting from wastewater discharge and for consideration of additional monitoring requirements for POPs and other contaminants of emerging concern (CECs), and a more extensive analysis of cumulative impacts, in order to support their effects determinations. In addition, we invited the EPA to consider engagement in a programmatic consultation that would address all EPA-permitted wastewater discharges that occur throughout the range of migratory ESA-listed species and designated EFH that occur on the U.S. west coast. EPA committed to helping facilitate a briefing from the City of Los Angeles on the monitoring program at Hyperion, which was provided to NMFS staff on January 10, 2017.

On January 26, 2017, we received a letter and revised BE from EPA in response to our requests for additional information and changes in the proposed permit conditions for Hyperion. In particular, EPA added substantial analysis and supporting information to the BE in response to the list of questions we provided on December 9, 2016. In addition, EPA proposed to require special studies that (among other things): (1) evaluate the projected effects of water conservation and planned recycling on effluent acute toxicity and ammonia, including a mass balance of nitrogen species through the treatment plants and an assessment of operation alternatives to address projected compliance with acute toxicity and ammonia water quality objectives; and (2) evaluate flame retardants and hormone concentrations in the effluent and loadings to the receiving water. Finally, additional monitoring requirements have been included in the proposed NPDES permit, which concern conditions in the zone of initial dilution (ZID), acute toxicity, organic pesticides, and chlorinated hydrocarbons.

In a February 21, 2017 letter, we informed EPA that we were not able to concur with all of the effects determinations made by EPA regarding the proposed issuance of this NPDES permit to Hyperion. In brief, we identified a number of outstanding questions and concerns about the potential exposure and/or response of ESA-listed species and the local environment to the wastewater discharge of Hyperion such that our interpretation of the available information precluded us from reaching all of the same conclusions that EPA has reached. In particular, we noted that the absence of current data regarding the discharge of polybrominated diphenyl ethers (PBDEs) makes it very difficult to draw conclusions regarding the magnitude of exposure of ESA-listed species or potential responses to this constituent in the action area (Santa Monica Bay). While we acknowledged that the proposed addition of a special study to collect data on the discharge of flame retardants, including PBDEs, represents an important foundation for developing the information necessary to more accurately assess the potential impacts of discharging flame retardants consistent with the comments and suggestions we have provided, we also recognized that we could not assume the risks of exposure to PBDEs are insignificant or discountable (the applicable standards in a finding of "not likely to adversely affect") in lieu of having more data available at this time. We stated that we anticipate being able to better address this question in future ESA and EFH consultations on future Hyperion permits after data has been collected from the Hyperion discharge in conjunction with a well-designed and executed study.

Given that there were at least some effect determinations made by EPA that we could not concur with at the time, we indicated to EPA in the February 21, 2017 letter that we would begin preparing our biological opinion on the proposed issuance of the NPDES permit by EPA to Hyperion, in accordance with the standards and procedures for formal consultation under section 7 of the ESA as described in 50 CFR §402 *et seq*. For the purposes of initiating and completing formal consultation on the proposed action, we have evaluated the information provided by EPA through informal consultation, including the revised BE submitted on January 26, 2017. After reviewing all the information provided, we believe that EPA has satisfied the requirements for initiating formal consultation under 50 CFR §402.14(c), and consider that formal consultation to have been initiated on January 26, 2017.

On November 16, 2017, we transmitted a draft biological opinion to EPA describing our analysis and conclusions regarding effects to ESA-listed species and designated critical habitats that can be expected as a result of the proposed action, per a request from EPA received by NMFS on March 14, 2017, in response to our determination that formal consultation was necessary. Specifically, EPA requested receipt of a draft biological opinion to review and discuss any Reasonable and Prudent Measures (RPMs), as provided by 50 CFR § 402.14(g). On January 12, 2018, staff from EPA and NMFS convened a call to discuss outstanding questions and concerns from EPA regarding the opinion, including language relevant to implementing the specified RPMs along with associated Terms and Conditions. On January 18, 2018, EPA submitted some suggested revisions to language in the RPMs and Terms and Conditions to NMFS via email. Consideration of suggested revisions has been taken into account in preparation of this final biological opinion on the proposed action.

1.3 Proposed Federal Action

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by Federal agencies (50 CFR 402.02). For the purposes of EFH consultation, a Federal action means any action authorized, funded, or undertaken, or proposed to be authorized, funded, or undertaken by a Federal Agency (50 CFR 600.910).

1.3.1 Hyperion Operation

The EPA has proposed to reissue the NPDES permit for Hyperion that authorizes the discharge of treated wastewater through two outfalls – discharge point 001 (the "one-mile outfall") and discharge point 002 (the "five-mile outfall") into the ocean for a period of time lasting up to five years.[1] The one-mile outfall is a 12-ft diameter outfall terminating approximately 1 mile west southwest of the treatment plant at a depth of 50 feet. The five-mile outfall is a 12-ft diameter outfall terminating at approximately 5 miles (i.e. 26,525 feet) west-southwest of the treatment plant at a depth of approximately 187 feet below the ocean surface, and therefore is in federal waters. Today, the 5-mile outfall is continuously active, while the 1-mile outfall is used only for emergencies and for preventative maintenance.[2] In 1987, Hyperion decommissioned a previously used outfall located approximately 7 miles west southwest of the treatment plant that used to discharge sludge. The plant has a capacity of 450 million gallons per day ("MGD") during dry weather and 850 MGD during wet weather. In 2014, the Hyperion Plant treated an average effluent flow of approximately 230 MGD. The service area for Hyperion covers about 90% of Los Angeles, collecting wastewater from around 4 million people and covering over 600 square miles. Domestic wastewater comprises approximately 79% of the wastewater flow with the remaining 21% from industrial and commercial sources. Wastewater processing consists of preliminary treatment, advanced primary treatment, secondary treatment, and if applicable, disinfection. Effluent is only chlorinated when discharged from the 1-mile outfall and for in-plant recycled uses.

The 5-mile outfall design maximizes dilution to lessen the potential impacts of the discharge on the marine environment. The 5-mile outfall ends in a "Y" shaped diffuser consisting of two 3,840-foot (0.7 miles) legs. On each diffuser leg, a series of 83 ports are alternately placed every six feet (Appendix 9.2 in the BE contains a diagram of the 5-mile and 1-mile outfalls). The 5-mile outfall is in an intermediate to low energy zone, which generally disperses and dilutes the effluent discharged at any given moment to very low concentrations after a week, although effluent is constantly being discharged (Uchiyama et. al 2014). The 5-mile outfall has a dry weather capacity of 450 MGD; with a peak hydraulic capacity of 720 MGD.[3] The 1-mile outfall is an emergency outfall operational during intense wet weather events and increases the hydraulic capacity of the facility to 850 MGD. However, actual wastewater flows into Hyperion have decreased over time due to a variety of conservation measures and drought conditions. During the period from January 2013 to December, 2014, Hyperion treated an average of 277 MGD and discharged an average of 242 MGD through the 5-mile outfall.[4] During 2015, the 5-mile average monthly flow ranged from 214 MGD (August) to 256 MGD (September). During intense wet weather, stormwater may overwhelm the storage capacity of the facility's stormwater wet wells and discharge from the 1-mile outfall. Preventative maintenance activities are performed up to four times a year to test the emergency valve for the 1-mile outfall and usually

[1] Because the facility discharges to waters of the United States both within and beyond state territorial waters, U.S., EPA and the California Regional Water Board jointly issue the NPDES permit.

[2] During 2013 to 2014, Hyperion discharged from the 1-mile outfall 6 times, and 2 times in 2015, excluding an approved bypass 5-week timeframe (September 21, 2015 to October 28, 2015) where the 1-mile outfall was used continuously as part of a rehabilitation project addressing repairs to the facility's effluent pumping plant.

[3] The 2017 proposed permit limits are based on the design flow rate of the treatment plant under the 1994 permit of 420 MGD, although the capacity has been increased since to 450 MGD (EPA 2017).

[4] The remaining flow, approximately 35 MGD, was recycled by the West Basin Municipal Water District.

results in a discharge of less than 5 MGD from the 1-mile outfall.

Hyperion recycles a portion of the effluent either in-house or at the West Basin Edward C. Little Recycling facility. In-house approximately 11 MGD are processed at the facility's Service Water Facility for internal plant use (i.e. line flushing, equipment seal water, cooling water, etc.). Approximately 35 MGD of effluent are sent to the West Basin facility for advanced treatment to produce recycled water (i.e. tertiary treatment, microfiltration, and/or reverse osmosis).[5] The Edward C. Little Recycling facility is permitted to discharge 5.2 MGD via the 5-mile outfall. Effluent from in-house use is also eventually discharged via the 5-mile outfall. These flows reflect only a small portion of the discharge, less than 3% (i.e. 2% brine and 1% recycled in-house flow), from the 5-mile outfall.

1.3.2 Permitted Effluent Limits

The proposed wastewater discharge requirements and permitted effluent limits for the 5-mile outfall and 1-mile outfall are described in detail in the Revised Tentative Permit dated January 20, 2017, submitted to us by the EPA along with the revised BE on January 26, 2017 (Table 5 and 6 respectively). These limitations, as well as other performance goals[6] that are monitored under the proposed NPDES permit, apply to an extensive list of constituents or parameters that represent markers of potential harm to marine life, human health, as well as overall impact on the local marine environment. Permitted effluent limits and performance goals may be measured over varying time scales, such as average monthly, weekly, or annual values, as well daily or instantaneous maximum values. Largely, proposed permitted effluent limits and performance goals are derived from or reflect the objectives that are laid out it in the California Ocean Plan (SWRCB 2015), along with site-specific considerations and the performance of Hyperion's discharge under previous NPDES permits.

In the BE, EPA evaluated effluent quality based on the 19 pollutants identified by the Santa Monica Bay Restoration Commission (SMBRC) as pollutants of concern in the Bay as well as those that are among the top, in terms of mass, discharged from Hyperion. Permitted effluent limits and performance goals for these constituents are established through the proposed permit. These 19 pollutants include: 1) Dichlorodiphenyltrichloroethane (DDT), 2) polychlorinated biphenyls (PCBs), 3) Polycyclic aromatic hydrocarbons (PAHs), 4) chlordane, 5) tributyltin (TBT), 6) cadmium, 7) chromium, 8) copper, 9) lead, 10) nickel, 11) silver, 12) zinc, 13) pathogens, 14) total suspended solids, 15) nutrients, 16) trash and debris, 17) chlorine, 18) biological oxygen demand, and 19) oil and grease. Of particular interest, permitted effluent limits and performance goals address levels of metals such as copper and nutrients such as ammonia in the effluent that may lead to toxic exposures for varying types of marine life, as well as POPs such as PCBs, that may lead to significant problems as they accumulate over time. It is important to note that minimum dilution ratios are used to calculate effluent limitations for nonconventional and toxic pollutants for discharges from Hyperion. These dilution ratios

[5] West Basin is contractually entitled to receive up to 70 MGD of effluent from Hyperion. West Basin's discharge of brine is permitted under a separate NPDES permit, CA0063401.

[6] Performance goals are based upon actual performance data for the Hyperion Treatment Plant and are specified only as an indication of the treatment efficiency of the plant. They are not considered enforceable effluent limitations or standards for the plant.

assume that some minimum level of effluent dilution occurs immediately upon discharge that buffers the exposure of receiving waters to effluent, and that permitted levels of constituents in the effluent reflect the levels of constituents that would be encountered in the environment immediately surrounding the discharge. At the 5-mile outfall, a dilution ration of 84:1 (parts seawater/parts effluent) is used for all pollutants except ammonia and chronic toxicity, where a dilution ratio of 96:1 is used. The minimum dilution ratio used to calculate effluent limitations for nonconventional and toxic pollutants for the 1-mile outfall is 13:1, reflecting the difference in discharge at the 1-mile outfall without use of a diffuser system.

1.3.3 Monitoring Program

The proposed permit prescribes a detailed monitoring program to demonstrate compliance with the effluent limitations, discharge specifications, and other requirements in the proposed permit (detailed in Attachment E of the proposed permit). In summary, the data collected from the required monitoring program covers 152 square miles for sampling of the water column and 69 square miles for sediment and benthic monitoring. Under the proposed permit, more than 5,000 samples receiving water, sediments, fish, and invertebrates are collected and analyzed each year from this area and provide useful data to assess potential impacts to the Bay.[7] The monitoring program consists of the following elements: influent and effluent monitoring stations; receiving water monitoring stations (inshore and offshore); benthic infauna and sediment chemistry sampling; trawl monitoring stations; and local bioaccumulation sampling.

The monitoring of effluent that is required occurs at varying intervals depending on the constituent being monitored; ranging from daily, weekly, monthly, quarterly, semiannually, to annually. Inshore receiving water monitoring occurs at least annually (summer) to determine if objectives for bacteria are being met. Offshore receiving water monitoring occurs at least quarterly to provide the information necessary to demonstrate compliance with the water quality standards. In addition, these data collected contribute to the Central Bight Cooperative Water Quality Survey. In addition, whenever there is effluent discharge at the 1-mile outfall, additional offshore sampling for bacteria is required. Benthic infauna and sediment chemistry sampling is conducted at least annually for regular assessment of trends in sediment contamination and biological response along a fixed grid of sites within the influence of the discharge. As part of this sampling, acute sediment toxicity monitoring using one of the three amphipod species (*Eohaustorius estuarius*, *Leptocheirus plumulosus*, and *Rhepoxynius abronius*) is required. Trawl monitoring occurs at least annually to look at the health of demersal fish and epibenthic invertebrate communities in the vicinity of the discharge and are used for regular assessment of temporal trends in community structure along an array of sites within the influence of the discharge. Bioaccumulation sampling occurs at least annually to determine if fish tissue contamination levels in the vicinity of the outfall are changing over time. Hornyhead turbot (*Pleuronichthys verticalisis*) is the preferred species; however if the required numbers and sizes of hornyhead turbot are not available, English sole (*Parophrys vetulus*) may be used as a substitute.

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[7] Hyperion's monitoring program is modelled off of the principles, framework, and recommended design for effluent and receiving water monitoring elements in SCCWRP's Technical Report #357. This framework was also adopted in the California Ocean Plan, Appendices. (Schiff et al. 2001)

Monthly chronic toxicity testing using whole effluent for both the 5-mile and 1-mile outfalls is required under the proposed permit. Whole effluent toxicity tests employ the use of standardized, surrogate freshwater or marine (depending upon the mixture of effluent and receiving water) plants, invertebrates, and vertebrates. Different test species can exhibit different sensitivities to toxicants. However, EPA considers standard test species to represent the sensitive range of all ecosystems analyzed. Under the proposed permit, topsmelt (*Atherinops affinis*), red abalone *(Haliotis rufescens)*, and giant kelp (*Macrocystis pyrifera*) are standard test species for toxicity. EPA has noted that the results of whole effluent tests in the past using five marine/estuarine short-term chronic test methods also indicate that no species or test method is always the most sensitive. Therefore, the 2017 permit requires re-screening of the standard test species every two years to ensure the most sensitive test species is used in evaluating the toxicity of the effluent. In the past, Hyperion has been using red abalone as the test species for chronic toxicity tests and topsmelt for acute toxicity tests. However, the 2017 draft permit no longer requires acute toxicity testing as EPA has determined that chronic toxicity testing is more sensitive than acute toxicity testing, and more protective as a result (EPA 2017).

1.3.4 Special Studies

The proposed permit requires a number of special studies to be performed during the permit period to address questions of interest or concern regard the operation or impact of Hyperion's wastewater treatment and discharge. Two special studies required in the proposed permit are highlighted below:

CEC Monitoring Special Study

Under the proposed action, Hyperion is required by EPA to develop a special study that evaluates flame retardants and hormone concentrations in the effluent and mass loadings to the receiving water. Hyperion is required to submit a Special Study Work Plan for approval by the Regional Water Board Executive Officer and the USEPA Water Division Director within one year of the effective date of the proposed permit and submit the special study report no later than two years before the permit expires.

Ammonia and Water Conservation Special Study

In coordination with the West Basin Municipal Water District, Hyperion is required by EPA to propose a special study that evaluates the projected effects of water conservation and planned recycling on effluent acute toxicity and ammonia, including a mass balance of nitrogen species through the treatment plant and an assessment of operational alternatives (e.g. treatment optimization, additional treatment, additional dilution credits) to address projected compliance with acute toxicity and ammonia water quality objectives. Hyperion is required to submit a Special Study Work Plan, including a proposed schedule, for approval by the Regional Water Board Executive Officer and the EPA Water Division Director no later than one year from the effective date of the proposed permit. The special study report shall be submitted no later than two years before the permit expires.

1.3.5 Proposed Action Area

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

In the BE, EPA determined that the entire Santa Monica Bay, 209 square miles, was the action area where direct and indirect effects are foreseeable and are reasonably certain to occur given that constituents of the plume from wastewater discharge from Hyperion have been detected at times throughout a large area of the Bay (Figure 1). This area also would include movement of persistent pollutants discharged from Hyperion. The action area includes an inner, mid, and outer shelf. The bay is a plateau between two canyons: Redondo Canyon (a known source of upwelling in the bay) and Santa Monica Canyon (about 3.5 miles offshore and heads at a depth of 180 feet. At times, the plume moves away from the outfall over tens of miles, but other times, the plume folds back on itself due to eddy current reversals.[8] The 5-mile outfall was designed so effluent discharged from the diffuser would be trapped below the thermocline where temperature and density differences are the greatest, acting as a barrier to most vertical water movements in most calendar months (i.e. spring, summer, and autumn), to prevent nearshore transport (City of LA 2015). The effluent plume has been detected moving in variable directions, reflecting the erratic nature of local currents and eddies. Normally, the plume is submerged between 65 to 100 feet from the surface due to density stratification. During winter conditions, stratification decreases, and the effluent plume may reach the surface. However, even under these winter conditions, the plume from the 5-mile outfall does not reach the shore (Figure 1). From 2011 to 2014, the plume was typically localized within 4.35 miles northwest to southeast of the outfall, irrespective of sea conditions (EPA 2017).

When effluent is discharged from the diffuser ports at the 5-mile outfall, there is an initial and rapid mixing of the effluent with ambient seawater until a point of neutral buoyancy is reached referred to as initial dilution. This plume of mixed effluent and ambient seawater moves away from the discharge point and becomes more diluted as distance increases from the outfall. The effluent rapidly mixes, either trapping below the surface or reaching a boundary, such as the surface or ocean bottom. Initial dilution in this case is completed when the diluting wastewater ceases to rise in the water column and first begins to spread horizontally. The region surrounding the diffuser where initial dilution occurs is generally referred to as the zone of initial dilution, or ZID. The process of initial dilution is rapid and energetic, with timescales of seconds to minutes. Following initial dilution, passive diffusion becomes the dominant physical process that results in further dilution of the effluent with seawater.

The ZID is defined by critical conditions. Critical conditions are those under which the initial dilution will be the lowest (and the physical mixing zone the largest). To define critical conditions, the plume characteristic and initial dilution must be evaluated for a range of effluent and ambient receiving water conditions. Critical conditions generally described by the highest effluent flow, the minimum and maximum ambient currents, and the density structure of the effluent and receiving water that result in the lowest initial dilution. Based on a 2015 analysis by

[8] During 2011 and 2012, the plume predominantly traveled south to south easterly. During 2013, the plume was detected throughout the Bay in the summer, and in 2014, the plume stayed clustered around the 5-mile outfall running northwest to southeast (EPA 2017).

Hyperion, the ZID under critical conditions for the 5-mile outfall was estimated to extend 65.6 feet on either side of the diffuser legs, and 130 feet vertically up from the diffuser (EPA 2017). The ZID essentially represents the boundary where the end-of-pipe effluent limits that are prescribed by the proposed permit are expected to meet the standards and objectives for water quality on which the proposed permit is based.

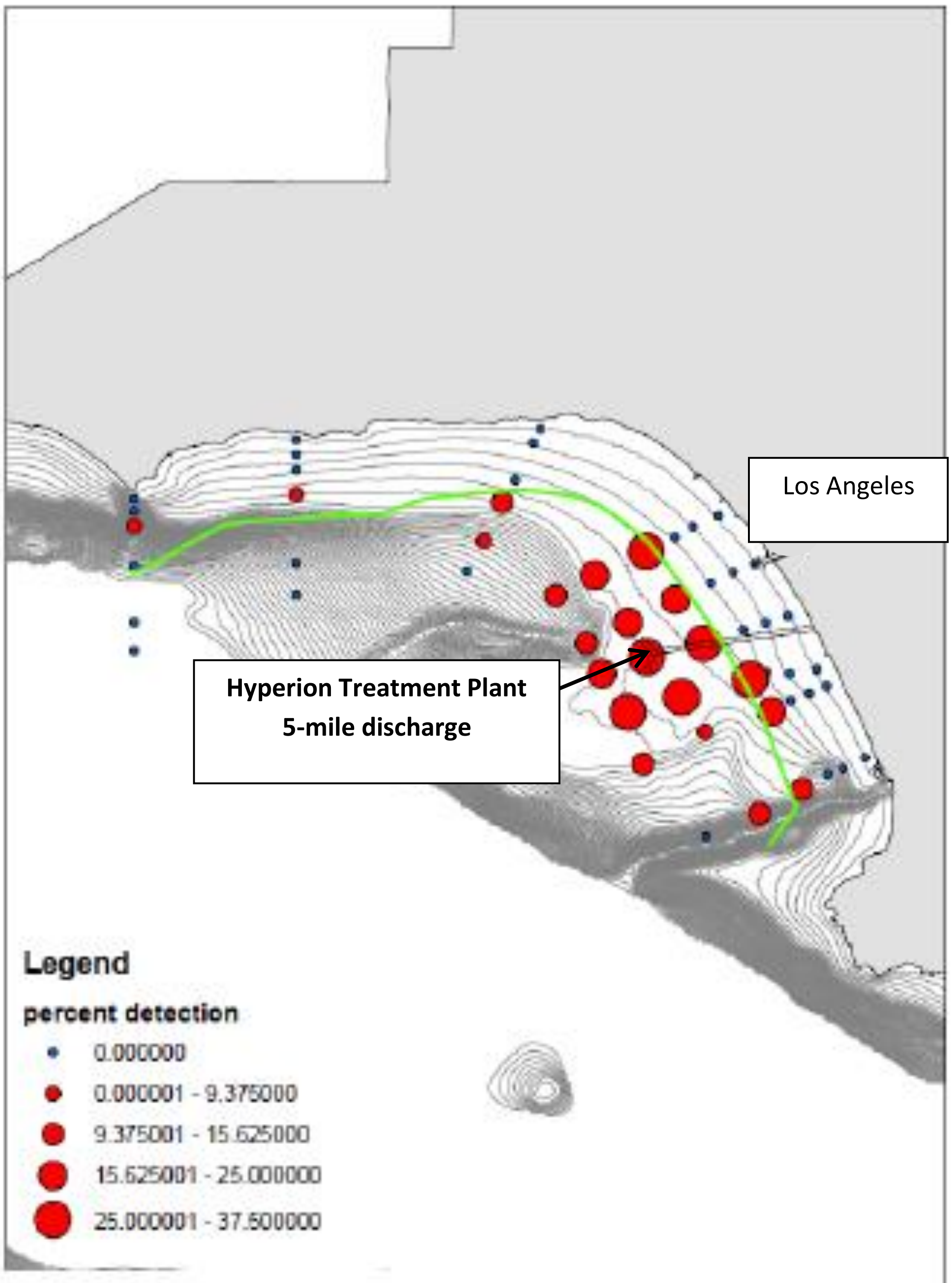

Figure 1. Location of Hyperion discharge into Santa Monica Bay, adjacent to Los Angeles, California. Figure 1 illustrates the plume probability for 2013 and 2014 by sampling location in terms of the percent detection of the wastewater field during all sampling. The green line denotes

the 3 nautical mile mark (From EPA 2017).

1.3.6 Proposed Action Area

"Interrelated actions" are those that are part of a larger action and depend on the larger action for their justification. "Interdependent actions" are those that have no independent utility apart from the action under consideration (50 CFR 402.02). No interrelated/interdependent actions were identified by EPA or NMFS during this consultation.

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

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

EPA has determined the proposed action is not likely to adversely affect: Southern California steelhead; North American green sturgeon, Southern DPS; scalloped hammerhead shark, Eastern Pacific DPS; blue whale; fin whale; humpback whale, Mexico DPS and Central America DPS; gray whale, Western North Pacific population; green sea turtle; leatherback sea turtle; loggerhead sea turtle; olive ridley sea turtle; and white abalone. No potential effects to ESA-designated critical habitat were identified by EPA.

As described above in section 1.2 *Consultation History*, we indicated that we could not concur with all of these determinations made by EPA after reviewing the available information. In this biological opinion, we analyze the likely adverse effects resulting from the proposed action on the following species that were identified by EPA as not likely to be adversely affected: blue whale; fin whale; humpback whale, Mexico DPS and Central America DPS; gray whale, Western North Pacific population; green sea turtle; leatherback sea turtle; loggerhead sea turtle; olive ridley sea turtle; white abalone, and black abalone. In addition, during consultation we determined that Guadalupe fur seals were likely to be adversely affected by this proposed action. We also determined that designated critical habitat for black abalone may be affected, but is not likely to be adversely affected (see section 2.11 *"Not Likely to Adversely Affect" Determinations*). Our concurrence with EPA's "Not Likely to Adversely Affect" determinations is documented in section 2.11 *"Not Likely to Adversely Affect" Determinations* for the following species: Southern California steelhead; North American green sturgeon, Southern DPS; and scalloped hammerhead shark, Eastern Pacific DPS.

2.1 Analytical Approach

This biological opinion includes both a jeopardy analysis and/or an adverse modification analysis. The jeopardy analysis relies upon the regulatory definition of "to jeopardize the continued existence of" a listed species, which is "to engage in an action that 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 relies on the definition of "destruction or adverse modification," which "means a direct or indirect alteration that appreciably diminishes the value of critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features" (81 FR 7214).

The designation(s) of critical habitat for (species) use(s) the term primary constituent element (PCE) or essential features. The new critical habitat regulations (81 FR 7414) replace this term with physical or biological features (PBFs). The shift in terminology does not change the approach used in conducting a ''destruction or adverse modification'' analysis, which is the same regardless of whether the original designation identified PCEs, PBFs, or essential features. In this biological opinion, we use the term PBF to mean PCE or essential feature, as appropriate for the specific critical habitat.

We use the following approach to determine whether a proposed action is likely to jeopardize listed species or destroy or adversely modify critical habitat:

- Identify the rangewide status of the species and critical habitat expected to be adversely affected by the proposed action.
- Describe the environmental baseline in the action area.
- Analyze the effects of the proposed action on both species and their habitat using an "exposure-response-risk" approach.
- Describe any cumulative effects in the action area.
- Integrate and synthesize the above factors by: (1) Reviewing the status of the species and critical habitat; and (2) adding the effects of the action, the environmental baseline, and cumulative effects to assess the risk that the proposed action poses to species and critical habitat.
- Reach a conclusion about whether species are jeopardized or critical habitat is adversely modified.
- If necessary, suggest a reasonable and prudent alternative (RPA) to the proposed action.

For this proposed action, we examine the available information regarding the constitution and extent of the wastewater effluent that is being discharged from Hyperion, as well as the expected and/or potential environmental impacts of that discharge with regard to effects to ESA-listed species and designated critical habitat. We also consider the framework that is established for monitoring the constitution and extent of the wastewater effluent that is established under the proposed action.

2.2 Rangewide Status of the Species

This opinion examines the status of each ESA-listed species that would be adversely affected by the proposed action. The status is determined by the level of extinction risk that the 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' current "reproduction, numbers, or distribution" as described in 50 CFR 402.02.

One factor affecting the range-wide status of ESA-listed species and aquatic habitat at large is climate change. Climate change has received considerable attention in recent years, with growing concerns about global warming and the recognition of natural climatic oscillations on varying time scales, such as long term shifts like the Pacific Decadal Oscillation or short term shifts, like El Niño or La Niña. Evidence suggests that the productivity in the North Pacific (Mackas 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 new use of previously unutilized or previously not existing habitats may be a necessity for displaced individuals. Changes to climate and oceanographic processes may also lead to decreased productivity in different patterns of prey distribution and availability. Such changes could affect individuals that are dependent on those affected prey.

The potential impacts of climate and oceanographic change on whales and other marine mammals will likely affect habitat availability and food availability. Site selection for migration, feeding, and breeding may be influenced by factors such as ocean currents and water temperature. For 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. Research on copepods has shown their distribution may be shifting in the North Atlantic due to climate changes (Hays et al. 2005). 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 (Learmouth et al. 2007).

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 (Kaska et al. 2006; Chan and Liew 1995). Rising sea surface temperatures and sea levels may affect available nesting beach areas as well as ocean productivity. 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). However, the existing data and current scientific methods and analysis are not able to make precise predictions about the future effects of climate change on this species or allow us to quantify this threat to the species (Hawkes et al. 2009).

Climate change impacts that could affect abalone and its habitat include sea level rise, ocean acidification, and elevated water temperatures. Rising sea levels may shift the distribution of rocky intertidal habitat along the coast, although this is expected to occur over very long time periods over which black abalone may be able to adapt and shift their range. Ocean acidification could result in water quality conditions that reduce larval survival and shell growth and increase shell abnormalities (Crim et al. 2011). However, 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; Feely et al. 2008; Feely et al. 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) and 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 1989; Tegner et al. 2001) and increase susceptibility to withering syndrome (Ben-Horin et al. 2013). At the same time, warmer water temperatures may benefit some abalone species 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 impacts to species resulting from proposed permit action could potentially influence the resiliency or adaptability of those species to deal with climate change that we believe is 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 being linked to the patterns of aggregated prey. Like other baleen whales, the seasonal and inter-annual distribution of blue whales is strongly associated with both the static and dynamic oceanographic features such as upwelling zones that aggregate krill (*Euphausia pacifica*; see Croll et al. 2005 for a recent review).

Blue whales are currently separated into three subspecies in the North Pacific, North Atlantic, and Southern Hemisphere. Their population structure has been studied through photo identification, acoustic, and genetic analyses showing both geographic isolation and overlap of some subpopulations. The Marine Mammal Protection Act (MMPA) identifies geographic stocks of marine mammals and requires the monitoring and management of marine mammals on a stock-by-stock basis rather than entire species, populations, or distinct population segments. The blue whales most likely to be observed within the proposed action area are identified as part of the Eastern North Pacific (ENP) stock. Tagging and photo identification studies have shown that the feeding population off southern California also migrates as far south as the equator to feed in the eastern tropical Pacific (Mate et al. 1999). These findings have been confirmed through vocal analyses, where the same call type representing the ENP stock have been recorded in the Gulf of Alaska south to the Costa Rica Dome (Stafford et al. 2001; Calambokidis et al. 2009). Recently, Irvine et al. (2014) documented the multi-year satellite track of a blue whale first tagged off of 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 of California to the ENP blue whale stock. Blue whales are commonly seen migrating near the action area between May and October, and are more common during summer months when krill are abundant (EPA 2017).

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 (Branch et al. 2007; Calambokidis and Barlow 2004). Although there is insufficient data available to well assess the present status in most parts of the North Pacific, there is evidence of a population increase rate of approximately 3% for the ENP stock based on mark-recapture estimates from the U.S. West Coast and Baja California, Mexico (Calambokidis et al. 2010), but it is not known if that corresponds to the maximum growth rate of this stock. Abundance estimates from summer/autumn research vessel surveys in the California Current ranged between approximately 400 and 800 animals from 2001 to 2008 (Barlow and Forney 2007; Barlow 2010). New photographic mark-recapture estimates of abundance for the period 2005 to 2011 presented by Calambokidis (2013) range from approximately 1,000 to 2,300 animals, with the most recent best estimate of blue whale abundance at 1,647 whales (Carretta et al. 2016a).

Threats: Blue whales experienced intensive whaling throughout the 20th century. Vessel interactions and fishery interactions, in addition to reduced prey abundance due to overfishing or other factors (including climate change), habitat degradation, and disturbance from low-frequency noise, constitute the most obvious threats to blue whales identified in the blue whale recovery plan (NMFS 1998). Because large whales that may become entangled in fishing gear such may often die later and drift far enough not to strand on land after such incidents, it is difficult to estimate the numbers of blue whales possibly killed and injured by fishing gear. Ship strikes are also a threat to all large whales, including blue whales, although reports of ship struck whales 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 (0.9/year) from ship strikes is less than the calculated potential biological removal[9] (PBR; 2.3) for this stock, but 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 ship strikes of blue whales (Carretta et al. 2016a). The number of blue whales struck by ships in the California Current likely exceeds the PBR for this stock (Redfern et al. 2013). To date, no blue whale mortality has been associated with U.S. west coast fisheries; therefore, total fishery mortality is approaching a zero mortality and serious injury rate (a standard under the Marine Mammal Protection Act; Carretta et al. 2016a). However, in 2015 and 2016, NMFS received the first confirmed reports of entangled blue whales along the U.S. west coast, although the ultimate fate of these animals is unknown and these events have not yet been evaluated for potential mortality and serious injury (NMFS WCR stranding data).

2.2.1.2 Fin Whale

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

Fin whales are distributed widely in the world's oceans and occur in both the Northern and Southern Hemispheres. In the northern hemisphere, they migrate from high Arctic feeding areas to low latitude breeding and calving areas. In the Atlantic Ocean, fin whales have an extensive distribution from the Gulf of Mexico and Mediterranean Sea northward to the arctic. The North Pacific population summers from the Chukchi Sea to California, and winters from California southward. Fin whales have also been observed in the waters around Hawaii. Fin whales can occur year-round off California, Oregon, and Washington (Carretta et al. 2016b), with 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). The fin whales most likely to be observed within the proposed action area are identified as part of the CA/OR/WA stock.

<u>Population Status and Trends:</u> 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

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[9] PBR is defined by the MMPA as the maximum number of animals, not including natural mortalities, that may be removed from a marine mammal stock while allowing that stock to reach or maintain its optimum sustainable population.

in terms of present population size relative to "initial" (pre-whaling, or carrying capacity) level is uncertain. Fin whales in the entire North Pacific are estimated to be less than 38 percent of historic carrying capacity of the region (Mizroch et al. 1984). The best estimate of fin whale abundance in California, Oregon, and Washington waters out to 300 nautical miles is 9,029 whales, generated from a trend-model analysis of line-transect data from 1991 through 2014 (Nadeem et al. 2016). The new trend estimates are based on similar to methods to those first applied to this population by Moore and Barlow (2011). However, the new abundance estimates are substantially higher than earlier estimates because the new analysis incorporates lower estimates of detection probability (Barlow 2015). The trend-model analysis incorporates information from the entire 1991-2014 time series for each annual estimate of abundance, and given the strong evidence of an increasing abundance trend over that time (Moore and Barlow 2011; Nadeem et al. 2016), the best estimate of abundance is represented by the estimate for the most recent year, or 2014. This is probably an underestimate because it excludes some fin whales which that could not be identified in the field and which were recorded as "unidentified rorqual" or "unidentified large whale".

Threats: A comprehensive list of general threats to fin whales is detailed in the Recovery Plan (NMFS 2010). 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. 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 not to strand on land after such incidents, it is difficult to estimate the numbers of fin whales killed and injured by gear entanglements. Documented ship strike deaths and serious injuries are derived from actual counts of fin whale carcasses and should be considered minimum values. The threats to fin whales due to underwater noise, pollutants, marine debris, and habitat degradation, are difficult to quantify. However, there is a growing concern that the increasing levels of anthropogenic noise in the ocean may be a habitat concern for fin whales that use low frequency sound to communicate.

For the CA/OR/WA stock of fin whales, the total quantified documented incidental mortality and serious injury (2.0/yr) due to fisheries (0.2/yr) and ship strikes (1.8/yr) is less than the calculated PBR of 81 (Carretta et al. 2016b). Total fishery mortality is less than 10% of PBR and, therefore, may be approaching zero mortality and serious injury rate (Carretta et al. 2016b). However, in 2015 and 2016, there have been additional instances where fin whale whales 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 to divide the globally listed endangered humpback whale into 14 DPSs and place four DPSs as endangered and one as threatened (81 FR 62259). NMFS has identified three DPSs of humpback whales that may be found off the coasts of Washington, Oregon and California. These are the Hawaiian DPS (found predominately off Washington and southern British Columbia [SBC]) which is not listed under the ESA; the Mexico DPS (found all along the U.S. west coast) which is listed as threatened under the ESA; and the Central America DPS (found predominately off the coasts of Oregon and California) which is listed as endangered under the ESA. Humpback whales are found in all oceans of the world and migrate from high latitude feeding grounds to low latitude calving areas. Humpbacks primarily occur near the edge of the continental slope and deep submarine canyons, where upwelling concentrates zooplankton near the surface for feeding. Humpback whales feed on euphausiids and various schooling fishes, including herring, capelin, sand lance, and mackerel (Clapham 2009). As described in the BE, humpbacks are commonly sighted 8 – 20 miles from shore along the CA coast, although sightings of humpback whales can occur within sight from shore. Because the primary outfall is 5 miles offshore and the effluent plume has been detected 4.35 miles beyond the outfall (EPA 2017), it is likely the humpback whales migrating and feeding in the Southern California Bight (SCB) will occur in the action area.

Current MMPA SAR for humpback whales on the west coast of the United States do not reflect the new ESA listings, thus we will refer in part to the status of the populations that are found in the action area using the existing SARs. The CA/OR/WA stock spends the winter primarily in coastal waters of Mexico and Central America, and the summer along the West Coast from California to British Columbia. As a result, both the endangered Central America DPS and the threatened Mexico DPS both at times travel and feed off the U.S. west coast. The Central North Pacific stock primarily spends winters in Hawaii and summers in Alaska, and its distribution may partially overlap with that of the CA/OR/WA stock off the coast of Washington and British Columbia (Clapham 2009). There is some mixing between these populations, though they are still considered distinct stocks. In December, 2016, NMFS WCR released a memo outlining evaluation of the distribution and relative abundance of ESA-listed DPSs that occur in the waters off the United States West Coast (NMFS 2016a). In summary, the proportional approach breaks down as follows:

Table 1. Proportional estimates of each DPS that will be applied in waters off of California, Oregon, and Washington/SBC.

Feeding Areas	Central American DPS (E)	Mexico DPS (T)
California/Oregon	20%	90%
Washington/SBC)	15%	42%

Based on the December 2016 memo, this biological opinion evaluates impacts 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 above. To the extent that impacts are evaluated at an individual animal level, these proportions would be used as the likelihood that the affected animal is from either DPS.

Population Status and Trends: Current estimates of abundance for the Central America DPS range from approximately 400 to 600 individuals (Bettridge et al. 2015; Wade et al. 2016). The size of this population is relatively low compared to most other North Pacific breeding populations. The population trend for the Central America DPS is unknown (Bettridge et al. 2015). The Mexico DPS, which also occurs in the action areas, is estimated to be 6,000 to 7,000 from the SPLASH project (Calambokidis et al. 2008) and in the status review (Bettridge et al. 2015). The population growth of California/Oregon feeding population of the North Pacific humpback whales has been estimated as increasing about 8 percent annually (the population growth rate for the entire North Pacific population is approximately 4.9 percent) (Calambokidis et al. 2008). The estimate for the abundance of the CA/OR/WA stock, which combines members of several different humpback whale DPSs, is 1,918 animals (Carretta et al 2016a).

Threats: A comprehensive list of general threats to humpback whales is detailed in the Recovery Plan (NMFS 1991). 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 impact of fisheries on the CA/OR/WA humpback whale stock is likely underestimated, since 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 ship 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 impacts 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.

Along the U.S. west coast, the estimated annual mortality and serious injury of the CA/OR/WA stock of humpback whales due to commercial fishery entanglements (5.3/yr), and non-fishery entanglements (0.2/yr), other anthropogenic sources (zero), plus ship strikes (1.0/yr), equals 6.5 animals, which is less than the PBR allocation of 11 for U.S. waters (Carretta et al. 2016b). 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 impacts. 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 (5.3/yr) is greater than 10% of the PBR; therefore, total fishery mortality and serious injury is not approaching zero mortality and serious injury rate (Carretta et al. 2016b). In 2015 (34 entanglements) and 2016 (54 entanglements), humpback whales were observed and reported entangled at record levels that will receive additional evaluation in upcoming SARs (NMFS WCR stranding data).

2.2.1.4 Gray Whale, Western North Pacific Population

Western North Pacific (WNP) gray whales were originally listed as endangered under the Endangered Species Conservation Act in June 1970 (35 FR 18319). WNP gray whales remain listed as endangered under the ESA (35 FR 8491). Currently there is no recovery plan for this population. There are two recognized gray whale stocks in the North Pacific; the WNP and the eastern North Pacific (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 most commonly observed near the limits of the Bay (EPA 2017). 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 is suggesting more overlap exists between these two stocks with WNP gray whales migrating along the U.S. west coast along with ENP gray whales. During the summer and fall, the WNP stock of gray whales feeds in the Okhotsk Sea, Russia and off Kamchatka in the Bering Sea (Carretta et al. 2016a). Known wintering areas included waters off Korea, Japan, and China. However, recent tagging, photo-identification, and genetics studies found some WNP gray whales migrate to the eastern North Pacific in winter, including off Canada, the U.S., and Mexico (Lang et al. 2011, Mate et al. 2011, Weller et al. 2012, Urbán et al. 2013). Combined, these studies have identified 27 individual WNP gray whales in the Eastern North Pacific (Carretta et al. 2016a). As a result, a portion of the WNP gray whale population is assumed to have migrated, at least in some years, to the eastern North Pacific during the winter breeding season.

<u>Population Status and Trends</u>: Photo-identification data collected between 1994 and 2011 on the gray whale summer feeding ground off Sakhalin Island in the WNP were used to calculate an abundance estimate of 140 WNP gray whales in 2012 (Cooke et al. 2013). There are some additional individual gray whales sighted during the summer off southeastern Kamchatka that have not been sighted off Sakhalin Island, but it is not clear whether those whales are part of the WNP stock (IWC 2014). 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 most recent estimate of abundance for the ENP population is from the 2010/2011 southbound survey and is 20,990 whales (Durban et al. 2013). At any given time during the migration, WNP gray whales could be part of the approximately 20,000 gray whales migrating through the California Current Ecosystem. However, the probability that any gray whale observed along the U.S. west coast would be a WNP gray whale is extremely small - less than 1% even if the entire population of WNP gray whales were part of the annual gray whale migration in the eastern North Pacific.

<u>Threats</u>: The decline of gray whales in the WNP is attributable to commercial hunting off Korea and Japan between the 1890s and 1960s (Carretta et al. 2016a). 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 ship strikes as well as a general degradation of the habitat. 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, increased shipping traffic, 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 (see Weller et al. 2013). 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 ship strikes or entangled in fishing gear within U.S. waters (Carretta el al. 2016a). 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 2008). 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 CITES and Mexican law. Guadalupe Island was declared a pinniped sanctuary by the Mexican government in 1975. Critical habitat has not been designated for this species in the U.S. Recently, likely in part due to their increasing trend and lack of threats, the species was "up-listed from "threatened" to "least concern" under the criteria of the IUCN Redlist 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) and Carretta et al. (2016b), and is therefore 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-AFSC 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 ½ 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 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 5 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 2017).

In recent years, Guadalupe fur seals have been increasing in numbers in the Channel Islands and several strandings have been observed along central CA coast. In 2015, an Unusual Mortality Event (UME) was declared.[10] The event is ongoing, with a total of 98 fur seals stranding in 2015 and 76 in 2016. All seals that stranded during this time period were malnourished or emaciated, and many had verminous and/or bacterial pneumonia, inflammation caused by parasites and/or bacteria, and a few animals had seizures due to domoic acid toxicity or hypoglycemia. Most animals were young, around 1 year old, post-weaning (Norris et al. 2017). As described in the BE, Guadalupe fur seals are occasional visitors to the action area, particularly in the summer.

Population Status and Trends: Commercial sealing during the 19th century reduced the once-abundant Guadalupe fur seal to near extinction in 1894. The size of the population prior to the commercial harvests of the 19th century 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 was made at other times of the year. These data indicate that the population of Guadalupe fur seals is increasing exponentially at an average annual growth rate of 10.3 percent (Carretta et al. 2016b). Direct counts of animals at Guadalupe Island and the Benito Islands during 2010 resulted in a minimum population estimate in Mexico at 15,830 animals, with a potential biological removal level of 542 Guadalupe fur seals (Carretta et al. 2016b). In the United States, a few Guadalupe fur seals are known to inhabit California sea lion (*Zalophus californianus*) rookeries in the Channel Islands (San Nicholas Island and San Miguel Island) (Stewart et al. 1987; National Marine Mammal Lab, unpublished data). Strandings of Guadalupe fur seals have occurred along the entire U.S. west coast, suggesting that the seal may be expanding its range (Hanni et al. 1997; NMFS-West Coast Region-stranding program unpublished data).

Threats: Although the Guadalupe fur seal population is growing at over 10 percent per year, 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 that 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 of Guadalupe fur seals occur around the rookeries, and the lower part of the California Current, which is influenced by human population centers with contaminant runoff, extensive oil tanker traffic and offshore oil extraction activity in southern California, increasing the risk of an oil spill. Sealing during the 19th century nearly exterminated the species; but with full protection in Mexico and in the U.S. now, it is presumed that Guadalupe fur seals are not presently hunted although it is not known if Guadalupe fur seals are currently being illegally killed.

There appears to be some minimal conflicts with fisheries, with gillnet and set-net fisheries likely take some animals, particularly in areas near Guadalupe Island and San Benito Islands (Aurioles-Gamboa 2015). Juvenile female Guadalupe fur seals have also stranded in central and

[10] http://www.nmfs.noaa.gov/pr/health/mmume/guadalupefurseals2015.html

northern California with net abrasions around the neck, fish hooks and monofilament line, and polyfilament string (Hanni et al. 1997). In 2016, one Guadalupe fur seal was found hooked in the Hawaii-based shallow-set longline fishery, and in 2015 three unknown pinniped species (otariids) were also found hooked that could have been Guadalupe fur seals (PIRO 2016, 2017). During El Nino 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 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 (2010-2014), in addition to fisheries interactions mentioned above, 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. 2016b). In addition, Guadalupe fur seals are susceptible to domoic acid toxicity, bacterial pneumonia and other associated impacts from emaciation/malnourishment (Norris et al. 2017). Military activities in southern California could affect Guadalupe fur seals through behavioral and physiological impacts from mid-frequency active sonar, underwater detonations, and missile launches and from sonic booms felt on the Channel Islands following a rocket launch. Scientific research is conducted on Guadalupe fur seals, primarily animals 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 and could threaten this species, depending on the extent of the spill.

2.2.2 Sea Turtles

2.2.2.1 Green Turtle, East Pacific DPS

Green turtles are found throughout the world, occurring primarily in tropical, and to a lesser extent, subtropical waters. The species occurs in five major regions: the Pacific Ocean, Atlantic Ocean, Indian Ocean, Caribbean Sea, and Mediterranean Sea. 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 turtles 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. The 2015 biological status report that was used to support the recent listing activities (Seminoff et al. 2015) can be found at: http://www.nmfs.noaa.gov/pr/species/Status%20Reviews/green_turtle_sr_2015.pdf

Molecular genetic techniques have helped researchers gain insight into the distribution and ecology of migrating and nesting green turtles. Throughout the Pacific, nesting assemblages group into two distinct regional areas: 1) western Pacific and South Pacific islands, and 2) eastern Pacific and central Pacific, including the rookery at French Frigate Shoals, Hawaii. In the eastern Pacific, greens forage coastally from southern California in the north to Mejillones, Chile in the South. Based on mitochondrial DNA 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: NMFS and USFWS (2007d) provided population estimates and trend status for 46 green turtle nesting beaches around the world. Of these, twelve sites had increasing populations (based upon an increase in the number of nests over 20 or more years ago), four sites had decreasing populations, and ten sites were considered stable. For twenty sites there are insufficient data to make a trend determination or the most recently available information is too old (15 years or older). A complete review of the most current information on green sea turtles is available in the 2015 Status Review (Seminoff et al. 2015).

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 1998). 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 mitochondrial DNA 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 has been suggesting steady increasing 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 percent 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 (Wallace et al. 2010; NMFS and USFWS 2007a). 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 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 in the vicinity of Long Beach, California (Lawson et al. 2011). This group of turtles has only recently been identified and very little is known about their abundance, behavior patterns, or relationship with the population in San Diego Bay.

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 impacts to foraging habitat becomes a concern. Pollution run-off can degrade sea grass beds that are the primary forage of green turtles. The majority of 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 and collisions with boat traffic are known to cause significant numbers of mortality every year (NMFS and USFWS 2007a; Seminoff et al. 2015). Marine debris is also a source of concern for green sea turtles due to the same reasons described earlier for other sea turtle species. 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 nesting season in Michoacán (Clifton et al. 1982). Even though 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 water continues to happen. In some places throughout Mexico and the whole of the eastern Pacific, consumption of green sea turtles remain 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 hatchling and many rookeries are already showing a strong female bias as warmer temperatures in the nest chamber leads to more female hatchlings (Kaska et al. 2006; Chan and Liew 1995). 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 el al. 2005), can cause erosion which leads to high nest failure (VanHouten and Bass 2007). Green sea turtles feeding may also be affected by climate change. 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). Climate change could cause shifts in ocean productivity (Hayes et al. 2005), which may affect foraging behavior and reproductive capacity for green sea turtles (Solow et al. 2002) similar to what has been observed during El Niño events in the western Pacific (Chaloupka 2001).

2.2.2.1 Leatherback Turtles

The leatherback turtle is listed as endangered under the ESA throughout its global range. 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 and Mexico. 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 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). 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 1998, 1999; Benson et al. 2007a, 2011). Recent satellite telemetry studies have documented transoceanic migrations between nesting beaches and foraging areas in the Atlantic and Pacific Ocean basins (Ferraroli et al. 2004; Hays et al. 2004; James et al. 2005; Eckert 2006; Eckert et al. 2006; Benson et al. 2007a; Benson et al. 2011). In the Pacific, leatherbacks nesting in Central America and Mexico migrate thousands of miles into tropical and temperate waters of the South Pacific (Eckert and Sarti 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 eastern North Pacific, westward to the Sulawasi and Sulu and South China Seas, or northward to the Sea of Japan (Benson et al. 2007a; Benson et al. 2011).

<u>Population Status and Trends:</u> Leatherbacks are found throughout the world and populations and trends vary in different regions and nesting beaches. In 1980, the leatherback population was estimated at 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). A current global population estimate is not available at this time, but details on what is known of populations are provided below.

In the Pacific leatherback populations are declining at all major Pacific basin nesting beaches, particularly in the last two decades (Spotila et al. 1996; Spotila et al. 2000; NMFS and USFWS 2007b). In the eastern Pacific, nesting counts indicate that the population has continued to decline since the mid 1990's leading some researchers to conclude that this leatherback is on the verge of extirpation (Spotila et al. 1996; Spotila et al. 2000). Steep declines have been documented in Mexico and Costa Rica, the two major nesting sites for eastern Pacific leatherbacks. Recent estimates of the number of nesting females/year in Mexico and for Costa Rica is approximately 200 animals or less for each county per year (NMFS and USFWS 2013) Estimates presented at international conferences show the numbers declining even more in all of the major nesting sites in the eastern Pacific.

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 with approximately 2700–4500 breeding females (Dutton et al. 2007; Hitipeuw et al. 2007). The current overall estimate for Papua Barat (Indonesia), Papua New Guinea, and Solomon Islands is 5,000 to 10,000 nests per year (Nel 2012). Although there is generally insufficient long term data to calculate population trends, in all of these areas, the number of nesting females is substantially lower than historical records (Nel 2012). This metapopulation is made up of small nesting aggregations scattered throughout the region, with a dense focal point on the northwest coast of Papua Barat, Indonesia; this region is also known as the Bird's Head Peninsula where approximately 75 percent of regional nesting occurs (Hitipieuw et al. 2007). Genetic results to date have found that nesting aggregations that comprise the western Pacific population all belong to a single stock (Dutton et al. 2007). The Bird's Head region consists of four main beaches, three that make up the Jamursba-Medi (JM) beach complex, and a fourth which is Wermon beach (Dutton et al. 2007).

The most recently available information on nesting numbers in northwest Papua reflects a disturbing decline. Tapilatu et al. (2013) estimated that the annual number of nests at Jamursba-Medi has declined 78.2 percent over the past 27 years (5.5% annual rate of decline), from 14,522 in 1984 to 1,532 in 2011. The beach at Wermon has been consistently monitored since 2002 and has declined 62.8 percent from 2,944 nests in 2002 to 1,292 nests in 2011 (11.6% annual rate of decline). Collectively, Tapilatu et al. (2013) estimated that since 1984, these primary western Pacific beaches have experienced a long-term decline in nesting of 5.9 percent per year. With a mean clutch frequency of 5.5 ±1.6, approximately 489 females nested in 2011. The total number of adult females in the Bird's Head region is estimated to be 1,949 based on summer nests (April-September) (Talipatu et al. 2013; Van Houtan 2014). This represents about 75 percent of the nesting activity in the Western Pacific; therefore NMFS estimates that there are approximately 2,600 nesting females in the population (in NMFS 2014a). While these two nesting beaches have been monitored since this most recent long term assessment was published, this is currently the best available information as data from 2012 through the present has not been analyzed (M. Tiwari, NMFS-SWFSC, personal communication, 2017). Recently, previously unknown by international researchers, leatherback nesting activity has been documented in a province south of Papua Barat, in northern Maluku, where 149 leatherback nests were discovered during a 3-4 month period (December, 2016 through February/March, 2017; J. Wang, NMFS-SWFSC, personal communication, 2017).

Migratory routes of leatherback turtles originating from eastern and western Pacific nesting beaches are not entirely known for the entire Pacific population; however, satellite tracking of post-nesting females and foraging males and females, as well as genetic analyses of leatherback turtles caught in U.S. Pacific fisheries or stranded on the West Coast of the U.S. indicate that the leatherbacks found off the U.S. West Coast are from the western Pacific nesting populations, specifically boreal summer nesters. Given the relative size of the nesting populations, it is likely that the animals will be from the Jamursba-Medi nesting beaches, although some may come from the comparatively small number of summer nesters at Wermon in Papua Barat, Indonesia. As mentioned earlier, one female has been tracked traveling from foraging areas on the U.S. West Coast to the Solomon Islands. The Papua Barat, Jamursba-Medi nesting population generally exhibits site fidelity to the central California foraging area (Benson et al. 2011; Seminoff et al. 2012).

Threats: The primary threats identified for leatherbacks are fishery bycatch and impacts at nesting beaches, including nesting habitat, direct harvest and predation. Leatherback 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. On the high seas, bycatch in longline fisheries is considered a major threat to leatherbacks (Lewison et al. 2004). At or adjacent to nesting sites, population declines are primarily the result of a wide variety of human activities, including legal harvests and illegal poaching of adults, immature animals, and eggs; incidental capture in coastal fisheries; and loss and degradation of nesting and foraging habitat as a result of coastal development, including predation by domestic dogs and feral pigs foraging on nesting beaches associated with human settlement and commercial development of coastal areas. 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). Marine debris is also a source of mortality to all species of sea turtles because small debris can be ingested and larger debris can entangle animals, leading to death.

2.2.2.3 Loggerhead Turtle, North Pacific DPS

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. On September 22, 2011, the U.S. Fish and Wildlife Service (USFWS) and NMFS published a final rule listing nine distinct population segments (DPS) of loggerhead sea turtles (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.

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 several years before returning to their neritic foraging habitats (Pitman 1990; Bowen et al. 1995; Musick and Limpus 1997). Adults may also periodically move between neritic and oceanic zones (Harrison and Bjorndal 2006). In the western Pacific, the only major nesting beaches are in the southern part of Japan (Dodd 1988). In Japan, loggerheads nest on beaches across 13 degrees of latitude (24°N to 37°N), from the mainland island of Honshu south to the Yaeyama Islands, which appear to be the southernmost extent of loggerhead nesting in the western North Pacific. 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; Kobayashi et al. 2008; Howell et al. 2008). Other important juvenile turtle foraging areas have been identified off the coast of Baja California Sur, Mexico (Peckham and Nichols 2006; Peckham et al. 2007; Conant et al. 2009). 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; Conant et al. 2009; Hatase et al. 2002).

Loggerheads that have been 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 have been found in great numbers, attracting top predators such as tunas, whales and sea turtles, particularly loggerheads (Pitman 1990; Wingfield et al. 2011). Pitman (1990) found loggerhead distribution off Baja to be strongly associated with the red crab, which often occurred in such numbers as to “turn the ocean red.” Considerable efforts have been spent studying the movements and relationships of juvenile loggerheads in the central Pacific and off Baja and the west coast of the U.S. to understand migrations and/or developmental patterns across the North Pacific (see Nichols et al. 2000; Polovina et al. 2003; Polovina et al. 2004; Polovina et al. 2006; Kobayashi et al. 2008; Howell et al. 2010; Peckham et al. 2011; Allen et al. 2013), but the ecology of juvenile loggerheads in the eastern Pacific is still not well understood.

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). Nesting beach monitoring in Japan began in the 1950s on some beaches, and grew to encompass all known nesting beaches starting in 1990 (Kamezaki et al. 2003). 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 40 percent of nesting occurs (Kamezaki et al. 2003). Census data from 12 of these 15 beaches provide composite information on longer-term trends in the Japanese nesting assemblage. As a result, Kamezaki et al. (2003) concluded a substantial decline (50–90%) in the size of the annual loggerhead nesting population in Japan 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. During the past decade, nesting increased gradually to 5,167 nests in 2005 (Conant et al. 2009), declined and then rose again to a record high of 11,082 nests in 2008, and then 7,495 and 10,121 nests in 2009 and 2010, respectively (STAJ 2008, 2009, 2010). At the November 2011 Sea Turtle Association of Japan annual sea turtle symposium, the 2011 nesting numbers were reported to be slightly lower at 9,011 (NMFS 2012a - Asuka Ishizaki, pers. comm. November 2011). The total number of adult females in the population was estimated at 7,138 for the period 2008-2010 by Van Houtan (2011). A more recent abundance estimate was conducted by Casale and Matsuzawa (2015) as part of an IUCN Red List assessment and, assuming a 2.7 year remigration and three nests per female (Conant et al. 2009), resulted in an estimated 8,100 nesting females in the population (data available through 2013). In recent years and through 2015, the population is generally increasing at the primary nesting beaches, with an overall (i.e., all Japanese nesting beaches) increase of 9% clutches per year over the most recent 12-13 years (through 2015; Y. Matsuzawa, Sea Turtle Association of Japan, personal communication, 2017). Therefore, the number of nesting females associated with the north Pacific loggerhead DPS is currently likely to be considerably higher than the Casale and Matsuzawa (2015)

Threats: A detailed account of threats of loggerhead sea turtles around the world is provided in recent status reviews (NMFS and USFWS 2007c; Conant et al. 2009). 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 are occurring throughout the species' range, especially coastal development, beach armoring, beachfront lighting, and vehicular/ pedestrian traffic. Coastal development includes roads, buildings, seawalls, etc., all of which reduce suitability of nesting beaches for nesting by reducing beach size and restricting beach migration in response to environmental variability. In Japan, many nesting beaches are lined with concrete armoring to reduce or prevent beach erosion, causing turtles to nest below the high tide line where most eggs are washed away unless they are moved to higher ground (Matsuzawa 2006). Coastal development also increases artificial lighting, which may disorient emerging hatchlings, causing them to crawl inland towards the lights instead of seaward. Overall, the Services 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).

For both juvenile and adult individuals in the ocean, bycatch in commercial fisheries, both coastal and pelagic fisheries (including longline, drift gillnet, set-net, bottom trawling, dredge, and pound net) throughout the species' range is a major threat (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 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; Ishihara et al. 2009; Conant et al. 2009). Additional fishery interactions in domestic and international pelagic fisheries in the North Pacific are also known to exist (Lewison et al. 2004; NMFS 2012a). As mentioned in the leatherback threats section, marine debris, including debris resulting from the 2011 earthquake and tsunami that took place off Japan, threatens the North Pacific DPS of loggerheads through ingestion and entanglement.

2.2.2.4 Olive Ridley Turtle

A 5-year status review of olive ridley sea turtles was completed in 2014.[11] Although the olive ridley 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. Olive ridley 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 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). A more complete review of current information can be found in the 5-year status review document published in 2014 by the US Fish and Wildlife Service and NMFS (NMFS and USFWS 2014).

<u>Population Status and Trends:</u> Olive ridleys are the most abundant sea turtle, but population structure and genetics are poorly understood for this species. It is estimated that there are over 1 million females nesting annually (NMFS and USFWS 2014). Unlike other sea turtle species, most female olive ridleys nest annually. According to the Marine Turtle Specialist Group of the

[11] http://www.nmfs.noaa.gov/pr/pdfs/species/oliveridleyturtle_5yearreview2014.pdf

IUCN, there has been a 50 percent decline in olive ridleys worldwide since the 1960s, although there have recently been substantial increases at some nesting sites (NMFS and USFWS 2007d). 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 is most likely to occur closer to eastern Pacific nesting and foraging sites, we assume that this population would be more likely to be affected by the proposed action. The eastern Pacific population is thought to be increasing, while there is inadequate information to suggest trends for other populations. Eastern Pacific olive ridleys nest primarily in large *arribadas* on the west coasts of Mexico and Costa Rica. Since reduction or cessation of egg and turtle harvest in both countries in the early 1990s, annual nest totals have increased substantially. On the Mexican coast alone, in 2004-2006, the annual total was estimated at 1,021,500 – 1,206,000 nests annually (NMFS and USFWS 2007c). Eguchi et al. (2007) analyzed sightings of olive ridleys at sea, leading to an estimate of 1,150,000 – 1,620,000 turtles in the eastern tropical Pacific in 1998-2006. In contrast, there are no known *arribadas* of any size in the western Pacific, and apparently only a few hundred nests scattered across Indonesia, Thailand and Australia (Limpus and Miller 2008).

<u>Threats:</u> 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. There has been historical and current direct harvest of olive ridleys. In the 1950's through the 1970's, 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 has been 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 UWFWS 2014).

Olive ridleys have been observed caught 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 2007d*)*. Based upon available information, it is likely that olive ridley sea turtles are being affected by climate change. Similar to other sea turtle species, olive ridleys are likely to be affected by rising temperatures that may affect nesting success and skew sex ratios and rising sea surface temperatures that may affect available nesting beach areas as well as ocean productivity. As mentioned in the leatherback threats section, marine debris, including debris resulting from the 2011 earthquake and tsunami that took place off Japan, threatens olive ridleys through ingestion and entanglement.

2.2.3. Marine Invertebrates

2.2.3.1 White Abalone

White abalone occur on the North American West Coast along offshore islands and banks (particularly Santa Catalina and San Clemente Islands) and along the mainland coast from Point

Conception, California, south to Punta Abreojos, Baja California, Mexico (Bartsch 1940; Cox 1960, 1962; Leighton 1972). NMFS published a final rule listing the white abalone as an endangered species on May 29, 2001 (66 FR 29046). On October 2008, NMFS published a final recovery plan for the white abalone (73 FR 62257), and recently NMFS also released a Species in the Spotlight 5-year Action Plan for white abalone to promote recovery actions and highlight the high level of extinction risk that currently exists for this species.

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). White abalone are the deepest living abalone species on the North American West Coast, occupying depths from 5-60m (Cox 1960), although current remnant populations are most common between 30-60 m depth and recent surveys by Butler et al. (2006) and Stierhoff et al. (2012) found the highest densities at depths of 40-50 m.

Abalone are broadcast spawners (i.e., individuals release their gametes into the water column and rely on external fertilization) and females and males must be in close proximity to one another to successfully reproduce. Spawning is highly synchronous (i.e., gametes are released at the same time), believed to occur once a year from February to April (Tutschulte and Connell 1981), and potentially triggered by chemical cues (bioactive triggers) and/or physical cues (abrupt temperature changes, tidal rhythm, lunar periodicity) (Giese and Pearse 1977; Leighton 2000) and the presence of the opposite sex (Hooker and Morse 1985; McCormick 2000). Females can release millions of eggs during a single spawn (Tutschulte and Connell 1981). About 24 hours after fertilization, the free-swimming larvae emerge from the embryo and swim in the plankton (Leighton 1989). This stage does not actively feed, but instead survives on its own yolk sac. The larval stage lasts about 3-10 days before the animals settle and metamorphose (McShane 1992). A chemical cue produced by crustose coralline algae induces abalone larvae to settle and metamorphose (Morse et al. 1979). Other environmental cues may also play a role in selection of a settlement site (Shepherd and Turner 1985; Slattery 1992; Daume et al. 1999).

Small juveniles feed on benthic diatoms, bacterial films, and other benthic microflora (Cox 1962). Juveniles occupy cryptic habitat (e.g., rock crevices, under rocks), move more frequently and over larger distances, and are difficult to see until they reach a size of about 75 to 100 mm (Cox 1962; Shepherd 1973; Tutschulte 1976). White abalone become sexually mature at approximately four to six years of age (about 88 to 134 mm SL; Tutschulte and Connell 1981). Abalone greater than 100 mm are considered "emergent" as they leave sheltered habitats and move to more open habitat to forage on attached or drifting macroalgae (Tutschulte 1976). Adults appear to have limited movements as they grow larger, remaining on homesites (Tutschulte and Connell 1988).

<u>Population Status and Trends</u>: Low population densities resulting from historical overfishing has been identified as the primary threat to white abalone in California. 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). However, 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).

<u>Threats:</u> White abalone face a high risk of extinction. 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.

2.2.3.2 Black Abalone

In contrast to white abalone, black abalone occupy rocky intertidal habitats from the upper intertidal to 6 meters depth. Historically, black abalone occurred from Crescent City (Del Norte County, California) to southern Baja California (Geiger 2004), but the current range is from

Point Arena, California, to Bahia Tortugas, Mexico, including offshore islands (74 FR 1937). On January 14, 2009, the species was listed as endangered under the ESA (74 FR 1937). Critical habitat was designated on October 27, 2011 (76 FR 66806).

Black abalone are most commonly observed in the middle and lower intertidal, in habitats with complex surfaces and deep crevices that provide shelter for juvenile recruitment and adult survival (see Leighton 2005 for review). They are able to withstand extreme variations in temperature, salinity, moisture, and wave action, and are usually strongly aggregated, with some individuals stacking two or three on top of each other (Cox 1960; Leighton 2005). Genetic studies indicate limited larval dispersal, with populations composed predominately of individuals spawned locally (Hamm and Burton 2000; Chambers et al. 2006; Gruenthal and Burton 2008).

As broadcast spawners, black abalone must be in close enough proximity to one another to successfully reproduce. They also have a short planktonic larval stage (about 3-10 days) before settlement and metamorphosis (McShane 1992). Larval black abalone are believed to settle on rocky substrate with crustose coralline algae, which serves as a food source for post-metamorphic juveniles, along with microbial and diatom films (Leighton 1959; Leighton and Boolootian 1963; Bergen 1971). Spawning has not been observed in the wild, but likely occurs from spring to early autumn (Leighton 1959; Leighton and Boolootian 1963; Webber and Giese 1969; Leighton 2005).

Population Status and Trends: Black abalone are believed to be naturally rare at the northern and southern extremes of their range, (Morris et al. 1980; P. Raimondi, pers. comm., cited in VanBlaricom et al. 2009). The highest abundances occurred south of Monterey, particularly at the Channel Islands off southern California (Cox 1960; Karpov et al. 2000). Rogers-Bennett et al. (2002) estimated a baseline abundance of 3.54 million black abalone in California based on landings data from the peak of the commercial and recreational fisheries (1972-1981). We note, however, that black abalone abundances in the 1970s to early 1980s had reached extraordinarily high levels, particularly at the Channel Islands, possibly in response to the elimination of subsistence harvests by indigenous peoples and large reductions in sea otter populations. Thus, our understanding of black abalone abundance and distribution for this time period may not accurately represent conditions prior to commercial and recreational harvest of black abalone in California.

Beginning in the mid-1980s, black abalone populations began to decline dramatically due to the spread of withering syndrome (Tissot 1995), a disease caused by a Rickettsiales-like organism (WS-RLO) that affects the animal’s digestion and causes starvation, leading to foot muscle atrophy, lethargy, and death (Friedman et al. 2003; Braid et al. 2005). The first recorded mass mortality associated with the disease was observed at Santa Cruz Island in 1985 (Lafferty and Kuris 1993). Researchers have since recorded mass mortalities at sites throughout the Channel Islands and along the California mainland as far north as Cayucos (San Luis Obispo County) by 1998-1999 (Altstatt et al. 1996, Raimondi et al. 2002). Withering syndrome was also observed in central Baja California around Bahia Tortugas during El Niño events in the late 1980’s and 1990s (Altstatt et al. 1996; Pedro Sierra-Rodriquez, pers. comm., cited in VanBlaricom et al. 2009) and may be linked to declines in the abalone fishery there in the 1990s.

Overall, populations throughout southern California and as far north as Cayucos have declined in abundance by more than 80%; populations south of Point Conception have declined by more than 90% (Neuman et al. 2010). Historical abalone harvest contributed to some degree, but the primary cause of these declines has been withering syndrome. Populations north of Cayucos have not yet exhibited signs of the disease, but all are likely infected by the WS-RLO pathogen. Disease transmission and manifestation is intensified when local sea surface temperatures increase by as little as 2.5 ºC above ambient levels and remain elevated over a prolonged period of time (i.e., a few months or more) (Friedman et al. 1997; Raimondi et al. 2002; Harley and Rogers-Bennett 2004; Vilchis et al. 2005). Thus, the northward progression of the disease appears to be associated with increasing coastal warming and El Niño events (Tissot 1995; Altstatt et al. 1996; Raimondi et al. 2002), and poses a continuing threat to the remaining healthy populations.

Most black abalone populations affected by withering syndrome remain at low densities, below the estimated levels needed to support successful reproduction and recruitment (0.34 abalone per m^2; Neuman et al. 2010). Data for 2002-2006 (Neuman et al. 2010) indicate that population densities exceed this threshold value in areas not yet affected by the disease (north of Cayucos; densities range from 1.1 to 10.5 abalone per m^2), whereas population densities fall below this threshold value, many significantly so, in areas affected by the disease (south of Cayucos; densities range from 0 to 0.5 abalone per m^2). Despite these low densities, however, researchers have observed evidence of recent recruitment and increases in abundance at several locations throughout southern California, including the Palos Verdes Peninsula, Laguna Beach, Santa Cruz Island, San Miguel Island, and San Nicolas Island (Richards and Whitaker 2012; Eckdahl 2015; unpublished data by Glenn VanBlaricom, U.S. Geological Survey).

Threats: Black abalone populations throughout California face high risk in each of four demographic risk categories: abundance, growth rate and productivity, spatial structure and connectivity, and diversity (VanBlaricom et al. 2009). Long-term monitoring data in California indicates that disease-impacted populations remain at low abundance and density, and the disease continues to progress northward along the coast with warming events, threatening the remaining healthy populations (Raimondi et al. 2002). The declines in abundance have potentially resulted in a loss of genetic diversity, though this needs to be evaluated. Although some sites in southern California have shown evidence of recruitment, natural recovery of severely-reduced abalone populations will likely be a slow process. Recovering the species will involve protecting the remaining healthy populations to the north that have not yet been affected by the disease, and increasing the abundance and density of populations that have already been affected by the disease.

2.3 Environmental Baseline

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 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (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. Although the action area for this proposed action (Santa Monica Bay) is a relatively small and confined area compared to the relatively large ranges of most of these species that are highly mobile, many of these same threats are present for animals when they do occur in the Bay. Given the large human population and high level of human activity in and around the coastal waters of Los Angeles, many of the threats including vessel strikes, disturbance, or habitat degradation, are especially prominent in this 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 impacting these species in the action area. Also, given the need to consider the potential impact of the proposed action introducing constituents into the environment that may affect ESA-listed species and the quality of the habitat, we review the current state of knowledge surrounding the health 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.

2.3.1 Habitat and Environment Health

Over 400 square miles of land area drains to Santa Monica Bay. This area is known as the Santa Monica Bay Watershed. There are 28 separate subwatersheds within the Santa Monica Bay Watershed, with the 2 largest being Ballona Creek and Malibu Creek watersheds (EPA 2017). About 16 MGD of runoff flows into the Bay from Ballona Creek during dry weather and 10 times higher or more during larger storms. Pollutants of concern from Ballona creek include heavy metals, trash/debris, pathogens, oil and grease, PAHs, and chlordane. Malibu Creek discharges nutrients, sediments, pathogens, total suspended solids, trash/debris, and oil into the Bay. While stormwater and urban runoff are significant sources of pollutants in the nearshore environment, EPA identified 7 major NPDES permittees in the action area that are significant sources of pollutants in the offshore environment. All of these facilities have NPDES permits and applicable effluent limits. These include an oil refinery, 3 electricity generating stations (Scattergood, El Segundo, and AES generating station), recycled water from the West Basin Edward C Little Water Recycling Plant (i.e. brine discharges via the 5-mile outfall), and two wastewater treatment discharges – Hyperion and Los Angeles County Sanitation Districts' Joint Water Pollution Control Plant (JWPCP). In 2011, the California State Water Resources Control Board identified the following permitted discharges into the Santa Monica Bay Watershed:

- 193 traditional NPDES discharges
- 18 minor NPDES discharges covered under individual permits
- 87 industrial stormwater NPDES discharges
- 401 construction stormwater NPDES discharges
- 175 discharges covered under other NPDES general permits

Over half of these discharges are related to stormwater and are discharged into the Bay through more than 200 outlets. Each year, an average of 30 billion gallons of stormwater and urban runoff can flow through the storm drain system (SWRCB 2011). According to the 2015 State of the Bay Report, SMBRC points out that most habitats in most areas of the Bay and its watersheds are degraded to some degree due to human activities. The report did highlight some of the recent successes with respect to habitat quality, such as soft-bottom habitat improvement with no dead zones, primarily due to reductions in DDT, PCB and mercury concentrations in the sediment, coupled with considerable reduction in suspended solids in wastewater treatment effluent. Specifically, the hypoxic conditions in the Bay were assessed as good and improving, with a moderate confidence level. The report also demonstrated a lack of sediment toxicity in most areas (SMBRC 2015).

There has been over 100 years of human waste (largely untreated or low/reduced/primary treatment) discharged into the Bay. As described in the BE, Hyperion has been discharging into the bay since 1894. The one mile outfall was constructed in 1925 and the 5 mile discharge has been operating since 1959. Full secondary treatment occurred in 1998. The effluent quality improved (in terms of turbidity, total suspended solids, and biological oxygen demand) subsequent to when Hyperion went from partial secondary to full secondary (City of LA 2014). In general, current levels of sediment contamination in the action area largely have resulted from the historical deposition (EPA 2017). For example, PCBs and DDT continue to have widespread contamination in the Bay following their ban decades ago. This emphasizes the highly persistent nature of these pollutants that continue to be measured in the sediment from Hyperion's historical discharge and from other sources, such as the Palos Verdes shelf.

2.3.1.1 Water Quality in the Action Area

As described above in section 1.3.2 of the *Proposed Action*, SMBRC identified 19 pollutants of concern for the Bay. The sources for these pollutants are varied, as the Bay receives pollutants from two marinas, seven major point sources described above, and over 160 smaller commercial and industrial facilities (SMBRC 2013). In general, contaminants enter marine waters and sediments from numerous point sources (including wastewater treatment plants, WWTPs) and non-point sources such as from atmospheric transport and deposition, ocean current transport, and terrestrial runoff (Iwata et al. 1993; Hartwell 2004; Hartwell 2008; EPA 2017).

Contaminant levels are typically concentrated near populated areas of high human activity and industrialization. For example, Santa Monica Bay is listed as a section 303(d) impaired water body under the Clean Water Act, largely due to sediment contamination (i.e. sediment toxicity) resulting from the historic discharge of primary treated wastewater and sludge. As a result, a fish consumption advisory exists for the Bay. Also, three new Total Maximum Daily Load standards (TMDLs) were enacted recently to address the impacts of marine debris, DDT/PCBs, and bacteria to the Bay. Higher bacteria levels occur near shore within the action area. As described in the BE, the 44 beaches in the Bay that were listed as impaired due to bacteria were because of the total and/or fecal coliform water quality standards were exceeded or because there were only one or more beach closures during the period assessed.

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

Metals and Ammonia

Metals are naturally found in the environment and some are essential to an animals' nutrition. However, human activities can increase the concentrations and metals can become toxic at certain exposure levels. Most metals settle to the ocean floor where they can accumulate in sediment. In the BE, EPA describes the levels of metals observed during sediment testing in the Bay, including levels of arsenic, cadmium, chromium, copper, lead, nickel, silver, mercury, and zinc. EPA reports that all metal concentrations around the 5-mile outfall during 2013 and 2014 were lower than historical averages and consistent with decreasing concentrations seen over time.

Ammonia is one of several forms of nitrogen existing in aquatic environments and is toxic to aquatic life at certain concentrations. When ammonia is present in water at high enough levels, aquatic organisms have difficulty excreting the toxicant. This leads to a toxic buildup in internal tissues and blood, and the buildup can cause death. Similarly, excess nutrients can increase plant and algal growth leading to eutrophication (EPA 2017). In the BE, EPA describes how ammonia concentrations in Hyperion's effluent have been increasing over the last 9 years, largely due to increased urbanization of the service area and the use of a thermophilic digester process. Specifically, the ammonia effluent concentration increased by 9% since the City began producing Class A biosolids. The most recent increases in ammonia effluent concentrations are also influenced by water conservation and drought conditions in the area. The City of Los Angeles found the highest ammonia concentrations in the receiving water located deeper than 66 feet at sites nearest the outfalls during 1994 to 2011 (EPA 2017). This observation of high ammonia concentration at this depth is also supported by receiving water monitoring data during 2013 to 2014 (City of LA 2015). In the BE, EPA reports that the receiving water monitoring data shows ammonia concentrations to be well below that discharged and below concentrations required by the ammonia water quality objectives.

Persistent Organic Pollutants

Persistent organic pollutants (such as PCBs, DDT, and PBDEs) can biomagnify, or accumulate up the food chain to a degree where levels in upper trophic-level species can have significantly higher concentrations than that found in the water column or in lower trophic-level species. PCBs were designed for chemical stability and were historically used in paints and sealants, industrial lubricants and coolants in electrical transformers and capacitors, and flame-retardants. There are potentially 209 congeners, or forms, and the chemical structure will influence the volatility, persistence, and toxicity. For example, the more chlorinated PCB congeners are more persistent in the environment than the less chlorinated congeners (Grant and Ross 2002). PCB congeners that are similar in structure to dioxin are highly toxic and can cause cancer, disruption to the immune system, reproductive impairment, endocrine disruption, and developmental problems (WHO 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 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 et al. 1988; Lieberg-Clark et al. 1995; Calambokidis et al. 2001; Rigét et al. 2010; Sericano et al. 2014), these compounds continue to be measured in marine biota around the world. Data from the Bay indicate that PCB and DDT levels in fish tissue have decreased over time, but still remain above the levels of concern. Unlike other areas in the SCB, the Bay has unpredictable currents. In the nearshore area of the Bay the prevailing current is toward the equator, whereas in the outside portion of the Bay the currents are poleward (SMBRC 2015). Consequently, the Palos Verdes shelf, located 2km offshore and south of the action area, is a major source of DDT and PCBs to the action area and to the surrounding area as well. This contamination of the shelf resulted from wastewater discharge from the Montrose Chemical Corporation manufacturing plant. The long-term significant discharges of DDTs from this manufacturing plant (Eganhouse et al. 2000, Bay et al. 2003) and the historically heavy agricultural use in the area lead to these higher levels of DDTs in California creating what has been called the “California signature”. In general, levels of DDTs are higher than PCB concentrations in sediments and marine biota from central and southern California (Jarvis et al. 2007, Blasius and Goodmanlowe 2008, Kimbrough et al. 2008).

PCB contamination is more widespread than DDT, likely due to the multiple sources including the inactive 7-mile outfall (EPA 2017). As described in the BE, near the 7-mile outfall, the highest DDT concentrations have been measured, consistent with the northward transport of DDTs from the Palos Verdes shelf. However, PCBs were found in highest concentrations in fish tissue near the 5-mile outfall. The PCB sources in the fish tissues are likely from the Palos Verdes shelf, Hyperion effluent, and stormwater (EPA 2017). Soft bottom habitats with high DDT, PCB, and mercury are reducing in size; however, the concentrations in the action area continue to be higher compared to the rest of the Southern California Bight (SMBRC 2015).

Recent decades have brought rising concern over a list of the so-called “emerging” contaminants and other pollutants, such as the PBDEs. PBDEs have been used as additive flame-retardants in many products including electronics, textiles and plastics. Additive flame-retardants can readily dissociate from the products they are added to and discharge into the environment. PBDEs have been identified as a growing concern and have a ubiquitous distribution with increasing levels found in various matrices including surface water, sewage sludge, sediment, air, and biota (Hale et al. 2003; Hites 2004). PBDEs are structurally comparable to PCBs and share some similar toxicological properties (Hooper and McDonald 2000).

Several studies have found higher PBDE concentrations in the sediments near wastewater outfalls (e.g., Gevao et al. 2006; Law et al. 2006; Samara et al. 2006; Johannessen et al. 2008; Grant et al. 2011). For example, measured PBDE concentrations in sediment immediately adjacent to the Iona Island wastewater outfall pipe (~12,700 pg/g) in British Columbia, Canada were 7 to almost 50 times greater than that measured elsewhere (Johannessen et al. 2008). In 2013, PBDE congeners (BDE-47 and BDE-99) were detected in Hyperion's effluent. In addition, other flame retardants (TCEP, TCPP, and TDCPP) were also consistently detected (EPA 2017). PBDEs were also documented in sediment and in fish tissues near the outfall. Although specific regional data is limited for PBDE levels, the environmental levels of a few PBDE congeners appear to have surpassed PCBs in some areas in North America (Hale et al. 2003; Ross et al. 2009). Several marine mammal species have recently experienced an almost exponential increase in PBDE concentrations (e.g., Ikonomou et al. 2002; Lebeuf et al. 2004).

TBT is a persistent pollutant and has been used as an antifoulant on ships, buoys, nets and piers to restrict or retard growth of fouling organisms. Although it may pose a toxic threat to species, bioaccumulation appears to be less than other persistent pollutants (e.g., PCBs, DDTs, and PBDEs). Butyltins have been measured in nearshore and in the deeper basins in southern California, including Santa Monica Bay (Venkatesan et al. 1998). TBT and other butyltins are found at relatively low levels in the basin sediments compared to other coastal sediments.

Contaminants of Emerging Concern (CECs)

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

- POPs such as PBDEs 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 promotersand 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.

As described in the BE, Hyperion's effluent is approximately 80% domestic wastewater, and many of the CECs found in higher concentrations include the PPCPs. Between 2012 and 2014, Hyperion's effluent was analyzed for CECs. Nonylphenol, TCPP, amoxicillin, azithromycin, gemfibrozil, galaxolide, caffeine, and iopromide were the highest CECs (above 1 micgrogram/L) measured in Hyperion's effluent (EPA 2017). Studies suggest that certain pharmaceuticals and personal care products (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 (Fair et al. 2009; Kannan et al. 2005; Nakata 2005; Nakata et al. 2007). 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. 2016). Recent evidence reveals one way microplastics are entering the marine food web is through zooplankton mistaking it for food (Wright et al. 2013; Desforges et al. 2015). Although the impact of microplastics in the food web is largely unknown, chemicals have been found to adsorb to these microplastics, including PCBs and DDT. In fact, some contaminants sorb 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. 2016).

2.3.1.2 Harmful Algal Blooms

In Pacific Ocean eastern boundary current upwelling systems such as that found in the SCB and Santa Monica Bay, significant amounts of nutrients are seasonally upwelled into shallower coastal waters. In the SCB, there is also a continuous source of nutrients discharged from several WWTPs such as the Hyperion facility (Howard et al. 2014) which is permitted to discharge up to 450 million gallons per day of secondary treated effluent during dry weather flows. Nitrogen is the primary nutrient limiting phytoplankoton production in coastal waters (Booth 2015) and additions of nitrogen cause phytoplankton production to increase, potentially reaching levels so high that they become harmful algal blooms (HABs). HABs in Santa Monica Bay are most commonly composed of diatoms or dinoflagellates, or a combination of several of these species and the zooplankton which graze upon them (Trainer et al. 2010). There are many known species in the California Current which may develop into HAB levels, but the most prevalent in Santa Monica Bay seem to be two diatom groups, the *Pseudo-nitzschia delicatissima* group and the *P. seriata* group, and dinoflagellates such as , *Akashiwo saguinea, Prorocentrum spp.,* and/or *Lingulodinium polyedrum*. The *Alexandrium tamarense* complex (*A. catenella* being most prominent) is present, but not common at high levels according to monitoring data generated by the Southern California Coastal Ocean Observing System (SCCOOS) at the Santa Monica Pier (http://sccoos.org/query/?project=Harmful%20Algal%20Blooms). The two diatom groups, *P. delicatissima* and *P. seriata* (*P. spp.* when referenced together) produce domoic acid which is responsible for well documented toxic events to marine mammals and birds in the SCB and amnesiac shellfish poisoning in humans. The *A. tamarense* complex can produce saxitoxin, which is responsible for Paralytic Shellfish Poisoning (PSP) and fish kill determinations (Backer and Miller 2016; Gosselin et al. 1989; Kudela et al. 2010; Lefebvre et al. 2004; Trainer et al. 2010) while *L. polyedrum* produces a yessotoxin, a large family of toxins whose potential impacts are being researched. Additional information regarding HABs and potential impacts to habitat and marine life can be found in section 3.2 *Essential Fish Habitat Effects Analysis*.

HAB occurrences appear to be increasing in frequency, duration, size, and severity throughout the SCB and the world in the last 10-15 years (Booth 2015; Howard et al. 2012; Nezlin et al. 2012). Anderson et al. (2012) notes that there are multiple reasons for this increasing bloom trend, including: natural dispersion of algal species, dispersal via human activities such as ballast water, improved detection of HABs and their toxins, increased aquaculture operations, and stimulation due to cultural eutrophication and climate change.

There is a compelling weight of evidence that nutrients are affecting algal dynamics in the SCB with chronic HAB outbreaks in areas that receive anthropogenic nutrient inputs (Booth 2015; Howard et al. 2014, 2012). 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 anthropological nitrogen sources. While this is true over the entirety of the SCB at peak upwelling, recent studies and resulting literature challenge this assumption in an important way. Nitrogen inputs from anthropological sources, particularly WWTPs which are continuous, are approximately equal to nitrogen inputs from upwelling at the spatial scales relevant to the formation of HABs (Booth 2015; Pondella et al. 2016; Howard et al. 2017, 2012; Corcoran and Shipe 2011) and the largest four WWTPs in the SCB account for 90% of the total WWTP discharges (Howard et al. 2014). In Howard et al. (2014) it was determined that Santa Monica Bay received equivalent nitrogen contributions from upwelling and wastewater at 47% of the total nitrogen flux for each, which suggests wastewater discharge roughly doubles the amount of nitrogen in the Bay.

2.3.2 Marine Mammals and Sea Turtles

2.3.2.1 Strandings

Strandings of ESA-listed marine mammals and turtles have been documented within the action area. Since 2012, the following total number of strandings has been documented within the action area (and immediate surrounding area) for each ESA-listed marine mammal species: fin whales (2); humpback whale (1); gray whales[12] (5); and Guadalupe fur seal (10). The cause of many of these strandings is unknown. It is known that one fin whale appeared to be a victim of a ship strike, and that all 10 Guadalupe fur seals appeared to be suffering from illness and/or malnutrition. Since 2012, the following total number of strandings has been documented within the action area (and immediate surrounding area) for each ESA-listed sea turtle species: green turtle (6); loggerhead (1); and olive ridley (1). It appears that four of the green turtles interacted with recreational fishing gear, and that two were entrained in local utility systems (1 power plant and 1 storm water retention pond). The one loggerhead turtle was entangled in marine debris, and the one olive ridley appeared to be possibly suffering from hypothermia.

[12] It is unknown if any of these gray whales were WNP gray whales.

2.3.2.2 Health and Contamination

Persistent organic pollutants 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. Persistent pollutants 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 POPs levels in ESA-listed marine mammals and sea turtles.

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; Robinson et al. 2013; Trumble 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 making direct comparisons difficult. Using different sampling techniques, such as biopsy or strandings, will yield different results (Krahn et al. 2001). Lastly, 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 et al. 2013).

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. Recently, 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 the two California feeding regions, the whales feeding in the southern region had levels more than 6 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 from 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 the lower end of that found in humpbacks from Alaska (Bachman et al. 2014). Furthermore, Domeles 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 POPs concentrations among these gray whales, which is likely due to the fact gray whales share common migration routes and foraging areas across the population. Between 1996 and 1998, 38 gray whales were biopsy sampled in the coastal waters of Washington (Krahn et al. 2001). Unlike in other species, these whales had higher mean PCBs levels compared to their DDT body burdens (2,100 ng/g lipid and 1,200 ng/g lipid, respectively). When comparing POPs in gray whales and humpback whales off Washington, gray whales had substantially higher PCBs than humpback whales, but slightly less DDTs. Humpback whales in southern California had substantially higher DDT concentrations (4,900 ng/g lipid) than that measured in gray whales (Krahn et al. 2001; Elfes et al. 2010).

Very little data are available for fin and blue whales. Blubber of fin whales off Iceland were measured for PCBs and DDTs (Borrell 1993). Similar to humpback whales, fin whales had relatively low POP levels compared to the toothed whales largely reflecting their lower trophic level status. POPs can also be measured in other matrices besides blubber. Earwax accumulates in some whale species throughout their lives and can be used to measure POPs. For example, POPs were measured in the earwax of a blue whale (Robinson et al. 2013; Trumble et al. 2013). The blue whale earplug was harvested after a ship strike off California. Although we can't 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 has 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 of these lower reproductive rates (Steiger and Calambokidis 2000; Elfes et al. 2010).

Currently POP levels in Guadalupe fur seals off California are not known. California sea lions generally share some of the migration habits and patterns as Guadalupe fur seals, and California sea lions eat a variety of prey species similar to that of Guadalupe fur seals. For these reasons, we examine POP levels in California sea lions as a proxy for potential contamination in Guadalupe fur seals. As expected, levels of PCBs and DDTs in dead California sea lions sampled in 2000 were higher than that found in humpback whales and gray whales (Kannan et al. 1994). 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. 1994). 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.

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

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., van de Merwe et al. 2009; Swarthout et al. 2010; D'llio et al. 2011). Similar to that found in other species, maternal transfer of POPs was documented with significant correlations between green sea turtle maternal blood and eggs, eggs and hatchling blood, as well as between maternal blood and hatchling blood (van de Merwe et al. 2010). Green sea turtles, loggerhead sea turtles, and hawksbill sea turtles in Japan were measured for PBDEs, PCBs, DDTs, and other organochlorine compounds between 1998 and 2006 (Malarvannan et al. 2011). When comparing PBDE levels among the three turtle species, green turtles had the lowest POP levels and decreasing concentrations were associated with increasing carapace length. Because green sea turtles are omnivores, we would expect them to have lower contaminant loads than other species that feed higher on the food chain. Interestingly, this was not observed in green sea turtles off San Diego. Blood and tissue from green sea turtles in San Diego Bay were sampled for trace metals, mercury, and POPs (Komoroske et al. 2011). 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.

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 can highlight the importance that foraging locations can strongly influence 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) also suggests the higher concentrations in San Diego green sea turtles may also be attributed to potential increased foraging rates as a result from elevated temperatures from power plant discharge, ultimately increasing consumption rates and thus elevated exposure risk. Furthermore, the high DDE levels in San Diego green sea turtles may suggest immunological effects because the levels exceeded lymphocyte proliferation no-effect levels established for loggerheads (Keller et al. 2006). Less is known about olive ridley sea turtles.

2.3.3 Abalone

2.3.3.1 Past and Ongoing Monitoring of Wild Populations

White abalone

Comprehensive surveys for white abalone have not yet been conducted throughout Santa Monica Bay, and thus limited information is available on white abalone presence within the action area. Most survey efforts in recent years have concentrated on the area off the Palos Verdes Peninsula. Since 2010, several surveys have been conducted and have identified several white abalone in waters off Palos Verdes (Neuman et al. 2015). Between October 2016 and March 2017, four of these white abalone individuals have been collected and brought into captivity to serve as broodstock for the white abalone captive breeding program, under the Scientific Research and Enhancement Permit 14344-2R issued by NMFS to the Bodega Marine Laboratory (BML) under ESA Section 10(a)(1)(A). These animals were deemed as "singletons" with a low likelihood of reproducing in the wild given their distance (>10 m) from other white abalone (NMFS 2016b). Prior to issuing the ESA Permit, NMFS conducted a consultation and concluded that the collection of wild white abalone under the Permit was not likely to jeopardize the continued existence of white abalone (NMFS 2016b). Instead, the collections have the potential to enhance the genetic diversity and productivity of the captive breeding program, to support species recovery. In March 2017, one of the newly collected animals was successfully spawned and contributed to the 2017 cohort of captive-bred animals at BML (pers. comm. with Kristin Aquilino, BML, 2 Mar 2017). The animals produced from this and future spawnings will be used for field planting and captive research activities.

Black abalone

Long-term monitoring of black abalone has been conducted at sites throughout the California coast since the mid-1970s or earlier. In the 1950s, Leighton and Boolootian (1963) recorded black abalone at rocky intertidal reefs at Palos Verdes (Flat Rock) and at Point Dume. After the disease hit in the 1980s and 1990s, black abalone have been rare along the mainland in the SCB. Because of their rarity, long-term monitoring surveys have continued but have not focused on black abalone and thus may not adequately detect their presence within the action area. Most recently, Eckdahl (2015) conducted surveys in Malibu and Palos Verdes (in the region down the coast from the action area) and found black abalone in Palos Verdes but not in Malibu. Eckdahl (2015) noted that in both areas, good and moderate quality habitat was present. Focused black abalone surveys are needed to fully assess their presence and status within the action area.

2.3.3.2 Impacts of Past Discharges from Hyperion Wastewater Treatment Plant and Other Discharges

White abalone and black abalone in the Bay have been exposed to effluent from Hyperion and other discharges for many years. A few studies have evaluated potentially harmful contaminant levels in black abalone and red abalone off Palos Verdes and other areas along the California coast and indicate that abalone have been exposed to and have accumulated contaminants in their muscle tissue. Jan et al. (1977) analyzed contaminant concentrations in red abalone collected off the JWPCP and Orange County Sanitation District (OCSD) and found elevated levels of silver, chromium, nickel, and zinc compared to red abalone collected at control sites (Catalina Island, Point Dume, and La Jolla). Black abalone collected around the same time period (1975-1977) from the Palos Verdes shelf showed similar levels of heavy metals as the red abalone collected off the JWPCP and OCSD, as well as total DDT levels of 0.001 mg/kg wet weight and PCB 1254 levels of 0.006 mg/kg wet weight (Young et al. 1980). The concentrations of copper measured in black abalone and red abalone in these studies (about 3.4 to 3.9 mg/kg wet weight) were greater than the concentrations measured in red abalone during acute and chronic exposure studies, in which the abalone were exposed to copper concentrations ranging from 66 to 126 ug/L (Viant et al. 2001). It is possible that abalone in the wild were exposed to high copper levels at that time, because in the 1970s, Hyperion and other discharges were releasing a mix of primary and secondary treated effluent; full secondary treatment was implemented at Hyperion in 1998 and at the JWPCP in 2002 (City of LA 2014). In a more recent study, Kannan et al. (2004) analyzed red abalone collected from Monterey Harbor in 1999 and found detectable levels of organochlorines and butyltins in the muscle tissue.

These studies did not evaluate the health effects of these pollutants on the abalone. Schafer (1961) found differences in the free amino acid content of black abalone collected from polluted waters (off Whites Point in Palos Verdes) compared to non-polluted areas (Channel Islands), indicating physiological effects of exposure to pollutants. In addition, Martin et al. (1977) showed that exposure to high copper concentrations can kill black abalone and red abalone. Studies involving other species of abalone have shown that exposure to high concentrations of heavy metals can affect abalone growth, reproductive development, and survival; these studies are discussed in more detail in Section 2.4 *Effects of the Proposed Action*. At this time, we can conclude that black abalone and white abalone within the Bay have been exposed to pollutants in the effluent from Hyperion and other discharges, but we cannot evaluate to what extent these discharges have affected the historical and present status of the species in the Bay.

2.3.3.3 Impacts of Fisheries Harvest

White abalone

Limited information is available on the historical presence of white abalone within the action area (Santa Monica Bay) and the effects of past fisheries harvest on the white abalone population there. Commercial landings of white abalone (by weight in shell) in the region from Point Dume to Palos Verdes made up about 0.8% of the total landings in California for the period from 1955-1997; this may include abalone collected elsewhere and landed/unloaded at the port within this region (Hobday et al. 2001). Thus, 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.

Black abalone

CDFG (now CDFW) recorded commercial landings of black abalone from the Malibu coast during the period of 1950 to 1993 when the fishery was open (2005). The Malibu coast was not one of the areas with the highest harvest of black abalone; however, harvest likely contributed to the decline in black abalone in this area. CDFW has prohibited harvest of black abalone since 1993, but poaching remains a problem.

2.3.3.4 Impacts of Other Factors

Other factors affecting white abalone and black abalone within the action area include disease, spills and spill response activities, HABs, and climate change impacts. Withering syndrome is a disease caused by a Rickettsiales-like organism (WS-RLO) that infects the gut tissue of abalone and inhibits the animal's ability to digest food. As a result, the animal becomes lethargic, unable to hold onto the substrate as its foot muscle shrinks, and typically dies within a few weeks of exhibiting symptoms. White abalone are known to be susceptible to the disease based on laboratory studies and observations of captive animals (NMFS 2008). One of the animals recently collected from the wild had low levels of infection with the WS-RLO (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. Withering syndrome was identified as the primary threat causing the severe declines in black abalone populations in Southern California, and continues to threaten remaining populations throughout the coast.

Spills and spill response activities, particularly oil spills, pose a risk to abalone populations depending on the type and amount of material spilled, the location, local environmental conditions, and the status of impacted populations. Given their presence in intertidal and shallow nearshore reefs, black abalone are likely at greater risk of spills than white abalone. NMFS is currently developing guidance on appropriate spill response activities and post-monitoring efforts to minimize and monitor the effects on abalone. The recent oil spill at Refugio Beach in 2015 resulted in oiling of rocky intertidal habitat, including an area where black abalone were found along the Santa Barbara coast (pers. comm. with Jack Engle and Pete Raimondi, 6 June 2015). Efforts are ongoing to monitor the impacts to the abalone and their habitat.

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; DeWit et al. 2014). Mortality was observed at all depths surveyed (0 to 20 m), with greater rates of mortality at shallower sites (DeWit et al. 2014). We do not know of documented abalone mortality events within the Bay that have been linked to HABs; however, the potential frequency and extent of blooms may have affected abalone survival in the Bay historically.

2.4 Effects of the Action

Under the ESA, "effects of the action" means the direct and indirect effects of an action on the species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action, that will be added to the environmental baseline (50 CFR 402.02). Indirect effects are those that are caused by the proposed action and are later in time, but still are reasonably certain to occur.

For the *Effects Analysis,* we have identified the following potential effects associated with the discharge of wastewater at Hyperion:

- Toxicity associated with exposure to the discharge plume constituents such as metals and ammonia

- Accumulation of other contaminants that may persist, be potentially harmful in low amounts, or otherwise emerging as concerns for marine life
- Exposure to environmental conditions created by the discharge of nutrients, including increased instances of harmful algal blooms

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

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

Given the overall similarity in how some ESA-listed species are generally exposed to the proposed action at an individual and population level based on similar long lived and migratory life histories, such as marine mammals and sea turtles, or more sedentary life histories such as abalone, we describe most of the *Effects Analysis* in general across these ESA-listed species groups. Where appropriate and necessary after the general synthesis of our understanding of how the proposed action may affect ESA-listed species groups, we consider species-specific information to help describe the potential effects of the proposed action.

2.4.1 Exposure and Response to the Toxicity of Hyperion Effluent

2.4.1.1 Species Occurrence and Exposure

To evaluate the presence of ESA-listed species within Santa Monica Bay, 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. To evaluate the presence of white abalone and black abalone within the Bay, we consider the distribution of potential abalone habitat, using habitat as a proxy for the presence of the species where we lacked harvest and/or monitoring data.

2.4.1.1.1 Marine Mammals and Sea Turtles

Marine mammals

Blue, fin, humpback, and gray whales are all generally well-known to be regular visitors to the

SCB throughout their lifetimes (juveniles and adults), observed frequently transiting or foraging in areas that can occur very close to shore, including within easy sight from land and/or access by recreational boaters, paddlers, etc. Specifically, whale watching companies throughout the SCB are the beneficiaries of the large amount of whale activity occurring in nearshore coastal waters. Individuals of all these whale species are known to visit Santa Monica Bay on an annual basis during migrations, and published scientific estimates of cetacean densities on the U.S. west coast (Becker et al. 2012) suggest that the coastal area in California that includes the Bay is an area 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 the Bay, it is expected that these whales would engage in foraging activity in association with prey sources that are known to occur in the Bay, including forage fish such as sardines and anchovies, and krill, 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 be variable, but generally can be expected to be as little as an hour up to several days a time. Exposure will generally follow seasonal patterns surrounding large-scale migrations, and could occur once per year during a migration, or multiple times for individuals that may be utilizing Southern California waters more regularly or for extended foraging activities.

Gray whale occurrence in the Bay is typically associated with biannual migratory transits between summertime foraging grounds in Alaska and breeding grounds in Mexico, and 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 Bay 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; Sanchez et al. 2001). Although the ESA-listed WNP gray whale population are expected to constitute not more than a small fraction of all the gray whales that migrate past and through the action area during a year, the fact that all of those gray whales will pass close to or into the action area makes it highly likely that at least some WNP gray whales will visit the action area during the 5-year period of the proposed action. WNP gray whale exposure is expected to be minimal as the animals would only potentially pass through the action area twice during the biannual migrations for very limited durations lasting no more than a number of hours each time.

As mentioned in section 2.3 *Environmental Baseline*, strandings of humpback fin and gray whales have occurred in or very near the Bay in recent years. As a result, we conclude all of these whale species, specifically the ESA-listed populations of these species, are likely to be in the action area and susceptible to impacts associated with the proposed action. As described in section 2.2 *Rangewide Status*, both ESA-listed DPSs of humpback whales are known to be present in California coastal waters and could be expected to occur in the Bay occasionally. While we do not expect any individuals of these whale species to take up extended residence in the Bay based on the highly migratory nature of their ecology, we do expect that some individuals could make numerous or possibly frequent and extended visits to the Bay over the course of relatively long-lived lifetimes of extensive migrations that include the SCB. For example, it has been documented that humpback whales have strong site fidelity and individuals feeding in and around the action area will likely return in subsequent years (throughout the 5-year period of the propose 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 2.3 *Environmental Baseline*, Guadalupe fur seal strandings have been documented stranding in or very near the Bay, and there has been an increase in strandings of Guadalupe fur seals along coast of California recently. These strandings began in the beginning of 2015 and are concurrent with the 2015-2017 California sea lion 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 impacts 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 Bay, we do expect that individuals could make numerous or possibly frequent and extended visits to the Bay over the course of relatively long-lived lifetimes of extensive migrations or residence in the SCB. The duration of exposure to the proposed action generally can be expected to be as little as an hour up to several days a time and could multiple times for individuals that may be utilizing Southern California waters more regularly or for extended foraging activities.

Sea turtles

From stranding data, anecdotal sightings, and scientific studies, we know that juvenile and adult green, leatherback, and olive ridley sea turtles occur at least occasionally in the SCB, as do juvenile loggerhead sea turtles as well. As mentioned in section 2.3 *Environmental Baseline*, strandings of green, loggerhead and olive ridleys have occurred recently in or very near the Bay. Although the SCB is not known to be a persistent or primary foraging or nesting location for leatherbacks, loggerheads and olive ridley sea turtles, the pelagic ecology of these species occasionally does lead them to migrate through the SCB and potentially into the Bay. While we do not have any information that suggests any individuals from these species take up extended residence specifically within the Bay, we do expect that individuals could make numerous or possibly frequent and extended visits to the Bay over the course of relatively long-lived lifetimes of migrations in the SCB. As a result, the duration of exposure to the proposed action (for individuals of all species may be variable, but generally can be expected to be as little as an hour up to several days a time. Exposure will generally follow seasonal patterns surrounding large-scale migrations, and could occur once per year during a migration, or multiple times for individuals that may be utilizing Southern California waters more regularly or for extended foraging activities.

Green turtle strandings in the area are much more frequent than the other sea turtle species, and existence of a resident foraging population of juveniles and adults in neighboring nearshore and estuarine areas (Long Beach) has recently been established through multiple avenues of study (Lawson et al. 2011; Crear et al. 2016). Although persistent occurrence or residence in nearshore or estuarine areas within the Bay by green turtles has not yet been documented, it is possible that some individual green turtles may spend some extended periods of time foraging within the Bay, and as a result be exposed more frequently or persistently to the proposed action. Fidelity to foraging sites by green turtles has been well described, including foraging sites in Southern California (e.g., Crear et al. 2016). At a minimum, we do expect that individual green turtles could make numerous or possibly frequent and extended foraging visits to the Bay over the course of relatively long-lived lifetimes of extensive migrations or residence in the SCB, and that the duration of exposure could last up to many days, weeks, or even months at a time.

Overlap with effects of the discharge

Schaffner et al. (2011) predicts the effluent discharge plume from Hyperion to extend throughout Santa Monica Bay, with greater plume probabilities in the vicinity of the outfall and downcoast toward Palos Verdes and lower plume probabilities upcoast toward Malibu, particularly in nearshore areas. For all marine mammal and sea turtle species, potential occurrence and overlap with the effluent discharge can occur essentially anywhere through the Bay, as all these species are highly mobile and could occur anywhere in the Bay. As described in section 1.3.5 *Proposed Action Area*, the ZID that represents the boundary of where concentrated effluent mixes with receiving water and the place where permitted effluent limits and performance goals apply represents a relatively small volume of the entire Bay. For the primary discharge at the 5-mile outfall, this is estimated to extend up to 65.6 feet on either side of the diffuser legs, and 130 feet vertically up from the diffuser during critical conditions. No ZID estimates have been provided for the occasional discharge that may occur from the 1-mile outfall, but we assume that this volume would also be relatively very small compared to the entire Bay. Although the volume of the Bay that is expected to be exposed to concentrated effluent is small, it is possible for ESA-listed marine mammals and sea turtles, or their prey or forage, to be exposed to the concentrated effluent within the ZID.

2.4.1.1.2 Abalone

White abalone

The available information regarding the abundance and distribution of white abalone in the action area, historically or currently, is very limited. The available fishery landings data indicate white abalone were present and harvested from the area between Point Dume and Palos Verdes, 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 Palos Verdes, where several white abalone have been found since 2010 (see section 2.3.3 *Environmental Baseline*). Focused surveys have not been conducted within the rest of the action area, particularly at deeper rocky reefs (e.g., at depths of 30-60 m).

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 m (Hobday and Tegner 2000). Using data from the NOAA NMFS EFH mapper (http://www.habitat.noaa.gov/protection/efh/efhmapper/) and CSUMB Seafloor Mapping projects (http://seafloor.otterlabs.org/SFMLwebDATA_s.htm#SMB), we identified subtidal rocky reef habitat within Santa Monica Bay (Figure 2). Most of the subtidal rocky reef habitat exists off Palos Verdes and the nearshore area (primarily within the 10m depth contour) along the coast of Malibu. Most of the habitat between the 30 and 60 m depth contour is soft-bottom habitat, with a larger rocky reef on the plateau between Redondo Canyon and Santa Monica Canyon at about 60 m depth and several smaller rocky reefs scattered throughout. White abalone are potentially present on these subtidal rocky reefs. In summary, white abalone are confirmed to be present at rocky reefs off Palos Verdes and are potentially present at nearshore and offshore rocky reefs throughout Santa Monica Bay, based on the presence of potential white abalone habitat.

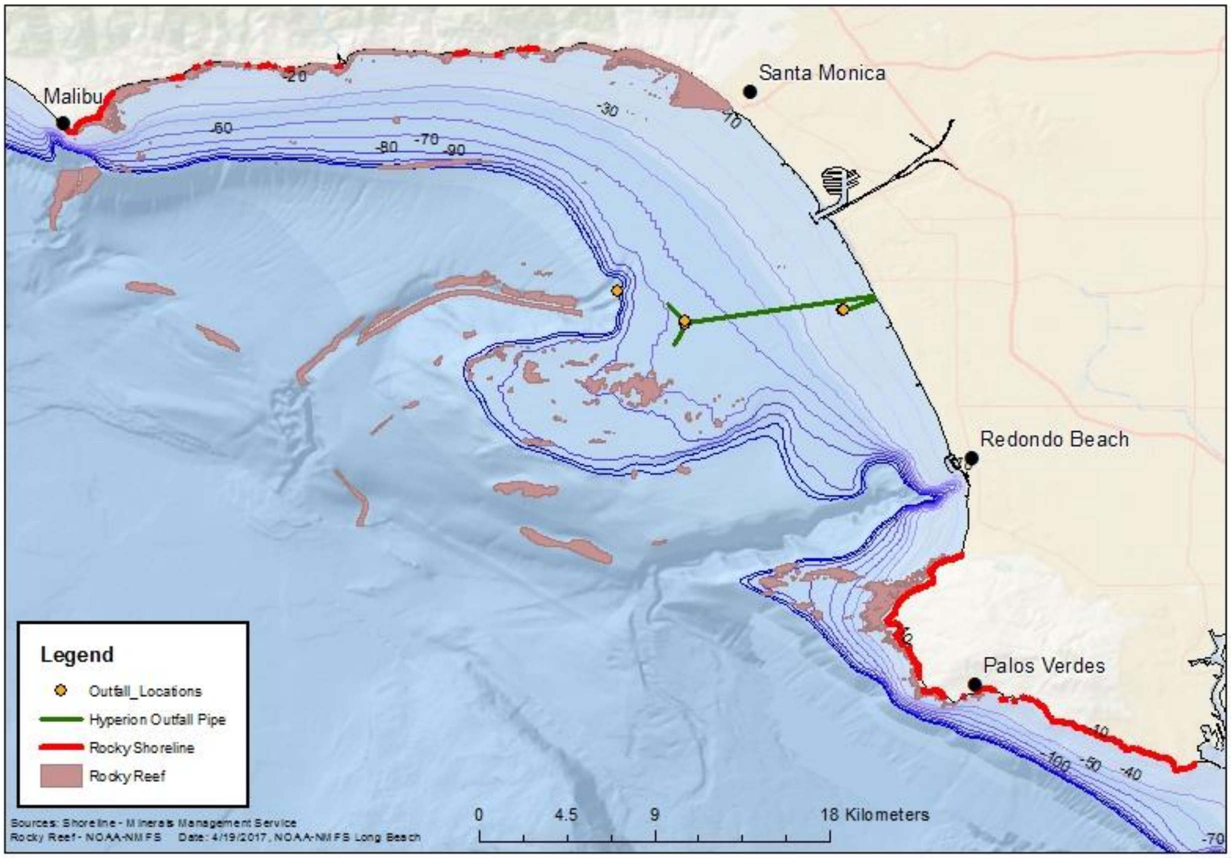

Figure 2. Rocky reef habitat in Santa Monica Bay.

Black abalone

Fisheries landings data and long-term monitoring data confirm the historical presence of black abalone in rocky intertidal habitats along the coasts of Palos Verdes and Malibu. However, information is lacking on the current presence of black abalone in these areas. Since the severe declines due to withering syndrome in the 1980s to 1990s, the numbers of black abalone along the Southern California mainland coast have remained low. As a result, long-term monitoring efforts along the Southern California mainland coast no longer specifically target black abalone, but focus on the intertidal community in general, noting black abalone when they are observed. Within the action area, the most recent black abalone focused surveys were conducted in 2012-2015, including 2 sites along the Malibu coast and 4 sites along the Palos Verdes coast (downcoast from Santa Monica Bay, outside of the action area; Eckdahl 2015). Several black abalone were found at the sites in Palos Verdes, but no black abalone were found at the sites in Malibu, although good and moderate quality habitat was present in both regions.

Where monitoring data are lacking, we rely on the presence of black abalone habitat as a proxy for the likely presence of black abalone. Rocky intertidal and shallow subtidal habitat occurs along the Palos Verdes and Malibu coasts. These habitats historically supported black abalone (prior to the disease-related declines) and habitat remains in good condition (Eckdahl 2015). That multiple size classes of black abalone were found at Palos Verdes, at sites just outside of the action area, also indicates the likelihood that black abalone occur within the intertidal and nearshore rocky reefs in the action area.

Overlap with effects of the discharge

As described above, the discharge plume extends throughout Santa Monica Bay, with greater plume probabilities in the vicinity of the outfall and downcoast toward Palos Verdes and lower plume probabilities upcoast toward Malibu, particularly in nearshore areas. Based on these modeled plume probabilities, the white abalone at rocky reefs off Palos Verdes and black abalone at intertidal and nearshore habitats along the Palos Verdes and Malibu coasts would be exposed to the discharge plume to some degree. In addition, white abalone habitat at offshore rocky reefs adjacent to the outfall, and any white abalone that may be present at those reefs, would be exposed to higher concentrations of the plume, whereas nearshore rocky reefs upcoast of the outfall would be less exposed to the plume. Thus, we conclude that the plume overlaps with nearshore and offshore rocky habitats where white abalone have been confirmed to occur or may occur, with higher plume concentrations affecting offshore areas such as the deep (60m) rocky reefs on the plateau adjacent to the 5-mile outfall. We also conclude that the plume overlaps with intertidal and nearshore rocky habitats where black abalone may occur, with higher plume concentrations affecting the reefs along the Palos Verdes coast.

The ZID occurs primarily within soft bottom habitat and does not overlap with potential white abalone habitat. However, the outfall structure itself may provide habitat for white abalone as white abalone could settle and survive on the outfall structure. However, the likelihood of white abalone occurring on the outfall structure within the ZID is low. Based on Schaffner et al.'s (2011) plume probabilities, any white abalone occurring on the outfall 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 within the vicinity of the ZID (EPA 2017), and 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. Under this Permit, white abalone found on the outfall structure may be collected, if they meet the collection criteria established under the Permit. Black abalone are not likely to occur in the ZID, because the ZID occurs outside of the species' depth range.

The ZID occurs primarily within soft bottom habitat and does not overlap with potential white abalone habitat. However, the outfall structure itself may provide habitat for white abalone. White abalone could settle and survive on the outfall structure; the probability of this increases with increasing distance from the ZID. Based on Schaffner et al.'s (2011) plume probabilities, any white abalone occurring on the outfall structure would be exposed to greater effluent concentrations, with the greatest exposure within the ZID. However, the likelihood of white abalone occurring on the outfall structure within the ZID is low. Surveys conducted to date have not observed any white abalone on the outfall structure within the vicinity of the ZID (EPA 2017), and 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. Under this Permit, white abalone found on the outfall structure may be collected, if they meet the collection criteria established under the Permit. Black abalone are not likely to occur in the ZID, because the ZID occurs outside of the species' depth range.

In summary, based on the best available information regarding white abalone and black abalone presence and the extent of the discharge plume and ZID within Santa Monica Bay, we conclude that: (a) white abalone off Palos Verdes may be exposed to constituents from the effluent discharge and their effects; (b) white abalone and black abalone habitat (and any individuals occurring in that habitat) may also exposed to the plume and its effects; and (c) white abalone may occur on the outfall structure itself and may be exposed to the plume and its effects, with greater levels of exposure in areas adjacent to the ZID; and (d) because of the sedentary life history of abalone, the risks for exposure are persistent across the entire year.

2.4.1.2 Constituents of Hyperion's Discharge

As described in the BA and in section 1.3.2 *Permitted Effluent Limits*, EPA evaluates 19 pollutants that are known to be present in quantifiable amounts in Hyperion's effluent. These pollutants include metals such as cadmium, copper, nickel, lead, silver, zinc. Elements and compounds such as nitrogen (ammonia), phosphorus, oil, and grease are also discharged. Other constituents of the discharge that are suspected or known to be present in wastewater discharge include: POPs such as PBDEs and TBT; PPCPs including prescribed and over-the-counter medications and numerous other products; and EDCs, including estrogen, androgens, and pesticides. In this *Effects Analysis*, we consider the potential effect of these pollutants on ESA-listed species.

2.4.1.3 Response to Exposure to Hyperion Effluent Plume

2.4.1.3.1 Effects on Marine Mammals and Sea Turtles

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 Hyperion does not appear to pose a threat to larger vertebrates that breathe air and have integumentary systems that limit direct uptake from the environment. The permitted effluent limits and performance goals for the proposed permit are set to meet the minimum standards of the California Ocean Plan (SWRCB 2015) that have been designed by the EPA and CA State Water Resources Control Board to protect marine organisms that likely are more immediately or directly sensitive to toxicity from wastewater effluent. Within the ZID, marine mammals and sea turtles could be exposed to relatively higher concentrations of various effluent constituents that are potentially toxic. We cannot precisely estimate exposure times within the ZID for ESA-listed marine mammals and sea turtles given their dynamic movements and occasional occurrence in the action area, but we anticipate exposure to concentrated effluent in the ZID will be relatively minimal. Several studies have recorded increased phytoplankton around the outfalls for multiple years (1957-1971; 1980) and more recent studies showed an increase in invertebrates around the 5-mile outfall (City of LA 1990; City of LA 2015). The increased productivity associated with effluent plume may attract all of these marine mammal and sea turtle species, which feed on forage fish and invertebrates, at least increasing the probability that the intake of food that may have been exposed to toxic pollutants from the effluent by ESA-listed marine mammals and sea turtles could occur in relative proximity to the outfall and the ZID.

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 Hyperion's effluent. These include ammonia, nickel, silver, and zinc. Some of these compounds are essential elements to nutrition (e.g., nickel and zinc; Das et al. 2003, Pugh and Becker 2001) 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 et al. 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 Hyperion's effluent that may cause adverse health effects but do not appear to biomagnify include: cadmium, chromium, copper, and lead. However, upper trophic-level predators can still accumulate metals even in the absence of biomagnification (Reinfelder et al. 1998). Low levels of arsenic, chromium, copper, and lead have been measured in marine mammal tissues and sea turtles (O'Shea 1999; Grant and Ross 2002; Das et al. 2003. Saeki et al. 2000; 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 Hyperion's effluent discharge as a result of occasional exposure to them when foraging in the Bay. See section 2.4.2 *Bioaccumulation of Pollutants* below for further analysis of potential effects associated with other more persistent and/or harmful constituents that may accumulate.

2.4.1.3.2 Abalone

To evaluate how white abalone and black abalone may respond to exposure to the discharge effluent, we consider the best information available regarding how different life stages of abalone may be affected by the potentially toxic pollutants and discharge effects to which they may be exposed. Because we lacked species-specific information for white abalone and black abalone, we use information from studies involving other abalone species to evaluate the potential effects on these species. We note that Section 2.4.2 *Bioaccumulation of Pollutants* addresses the potential effects associated with other more persistent and/or harmful constituents that may accumulate.

Effects on larval abalone

Studies have been conducted using other abalone species to evaluate the effect of different potentially toxic pollutants on larval stages, including some of the pollutants identified in Hyperion's discharge effluent. These primarily include heavy metals (e.g., copper, zinc, mercury, iron, lead, and cadmium; EPA 2017). Heavy metals can have adverse effects on larval shell development, depending on the concentrations to which larvae (fertilized egg to veliger stage) are exposed (e.g., Conroy et al. 1996; Gorski 2006; Gorski and Nugegoda 2006).

Copper, zinc, cadmium, and lead have been detected in Hyperion's effluent in quantifiable amounts (EPA 2017). For 2012-2016, the annual monthly average concentrations reported by EPA for cadmium and lead in Hyperion's discharge were below the values found to cause adverse effects on larval abalone; however, the values for copper and zinc were above the values found to cause adverse effects (EPA 2017). The reported values represent the concentrations within undiluted effluent. Thus, larvae would only be exposed to these concentrations within the ZID. Outside the ZID within the plume, concentrations would be diluted by a factor of 84:1 or more. Given this dilution factor, concentrations of copper, zinc, cadmium, and lead experienced in the plume outside of the ZID would be below those found to cause adverse effects on larval abalone. However, we note that the reported values are annual monthly averages; daily or instantaneous values may be higher, but sampling of the effluent for these constituents only occurs once a month or once a quarter, and raw sample data was not reported in the BE.

The City of Los Angeles uses red abalone as the test species for chronic toxicity tests in which larvae (fertilized egg to veliger stage) are exposed to different concentrations of the discharge effluent for 48 hours and then evaluated for reduced or abnormal shell growth (e.g., multiple indentations of the shell, lack of calcification, broken shells, shells separated from the rest of the animal). Studies comparing the effects of 48-hour exposure and 10-day exposure confirmed that this 48-hour test is an appropriate measure of adverse chronic effects on abalone larvae, including effects on survival past the planktonic stage and the ability to settle and metamorphose (Conroy et al. 1996).

During 2011-2014, the discharge from the 5-mile outfall and the 1-mile outfall did not exceed the chronic effluent limits, meaning that there was no observable effect of the discharge on the larvae (compared to the control) at the concentrations that larvae would be exposed to outside of the ZID (EPA 2017). These results indicate that exposure to the effluent concentrations found in the plume (outside of the ZID) would not be expected to reduce the survival and development of larval white abalone or black abalone.

Within the ZID, abalone larvae could be exposed to higher concentrations of the discharge effluent that can cause reduced or abnormal shell growth and development. The 2011-2014 chronic toxicity results indicate that abalone larvae can be exposed to higher concentrations of the discharge effluent (as high as 20% effluent) than is expected outside of the ZID (i.e., 1.19% effluent) and show no observable effect compared to the control; however, in most tests, observable effects on shell growth and development occurred at effluent concentrations of 2.1% or more (EPA 2017). However, these tests used an exposure time of 48 hours. Given the limited extent of the ZID and the planktonic nature of abalone larvae, we would expect that if abalone larvae were to pass through the ZID, the exposure time would be shorter than 48 hours. 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. The likelihood of occurrence within the ZID is lower for black abalone larvae than white abalone larvae, given the location of the ZID offshore and at deep depths (60m), much deeper than the depth range of black abalone (intertidal to 6 m depth).

In summary, studies show that the types of pollutants found within the discharge effluent can adversely affect larval development above certain concentrations. However, reported concentrations of the pollutants in the effluent plume (outside of the ZID) were below the values found to cause adverse effects on larvae, and chronic toxicity tests conducted by the City indicate that exposure of abalone larvae (early stages from the fertilized egg to veliger) to the effluent concentrations within the plume (outside the ZID) is not likely to result in an observable effect on larval development. Larvae may pass through the ZID and be exposed to higher effluent concentrations; however, the likely short duration of their exposure to the ZID indicates that effects on development would also be low. Based on this information, we would not expect reduced larval development or survival in abalone larvae exposed to the discharge plume. In addition, although larvae passing through the ZID would be exposed to higher effluent concentrations, we also do not expect this exposure to reduce larval development or survival given the likely short duration of the exposure compared to the 48-hour exposure times shown to cause adverse effects. White abalone larvae could settle on the outfall structure itself within the ZID; if so, they likely would not survive or would experience abnormal development due to high concentrations of copper and zinc.

Effects on juvenile and adult abalone

For juvenile and adult abalone, exposure to pollutants within the discharge effluent may occur through direct uptake from the water or from food (e.g., attached and drift macroalgae). We considered the information available regarding the effects of contaminants on juvenile and adult abalone, primarily using information from studies on other abalone species to infer potential effects on white abalone and black abalone.

The contaminants identified in the discharge effluent include heavy metals that have been found to have harmful effects on other species of abalone, and thus could have harmful effects on white abalone and black abalone. Several studies involving other abalone species have evaluated the effects of water borne and/or dietary exposure of juveniles and adults to copper, zinc, silver, and cadmium and found that abalone can accumulate these metals in their foot muscle, mantle tissues, and viscera, and experience adverse effects on growth, behavior, and survival (e.g., Martin et al. 1977; Liao et al. 2002; Gorski 2006; Chen et al. 2011; Huang et al. 2008, 2010). 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 ug/L for *H. rufescens* and 50 ug/L for *H. cracherodii*, compared to 87ug/L for *H. rubra* (Gorski 2006). Viant et al. (2001) 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 exposures 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 proposed discharge, we consider the concentration of these pollutants in the effluent and the concentrations to which white abalone and black abalone may be exposed. From 2012-2016, the annual monthly average concentrations for the 5-mile outfall effluent ranged from 9.19 to 26.07 µg/L for copper, and from 13.74 to 26.2 µg/L for zinc (EPA 2017). For silver and cadmium, annual average concentrations for the 5-mile outfall effluent were higher in the 1990s (about 3-5µg/L Ag and 1-2 µg/L Cd) but have been less than 1 µg/L from 2006/2007 through 2013/2014 (EPA 2017). Overall, the levels of copper, zinc, silver, and cadmium reported in the effluent are lower than the levels found to cause mortality or sublethal effects on abalone. Taking into account the dilution of effluent outside of the ZID (a dilution of at least 84:1), the levels of these metals in the plume are expected to be well below those documented to cause mortality or sublethal effects on abalone. Based on this, we would not expect exposure to the levels of these metals in the plume to inhibit growth, behavior, or survival of white abalone or black abalone. The level of risk would be lower for black abalone, given their location in nearshore habitats and likely exposure to lower concentrations of the effluent. We also would not expect exposure to the levels of these metals in the ZID to inhibit growth, behavior, or survival of white abalone juveniles and adults, should individuals be present within the ZID (i.e., on the outfall structure).

We note two sources of uncertainty. First, abalone may be able to develop increased tolerance to heavy metal exposure via chronic exposure to sublethal levels; this has been indicated in chronic exposure studies involving copper (Martin et al. 1977; Viant et al. 2001), silver, and cadmium (Huang et al. 2010). Increased tolerance would further reduce the effects of exposure on abalone, although further studies are needed to better understand how tolerance may develop and whether it compromises other aspects of the individual's health (e.g., growth, reproductive development, immunity). Second, little is known about the effects of exposure to multiple metals and other pollutants on abalone, or the cumulative levels to which abalone are exposed in the Bay. Synergistic effects could increase the potential for adverse effects on abalone. Unlike for larval stages, studies have not been conducted to directly evaluate the effects of Hyperion's discharge effluent on juvenile and adult abalone. As a result, we do not know how long-term exposure to the concentrations of copper, zinc, and other heavy metals in the effluent may affect juvenile and adult abalone growth, development, reproduction, and survival. Further studies to directly evaluate the effects of the effluent on juvenile and adult abalone could help to reduce this uncertainty.

2.4.2 Accumulation of Potentially Harmful Contaminants

2.4.2.1 POP Loading into the Action Area

Persistent organic pollutants are contaminants of concern for ESA-listed species and other marine life because they bioaccumulate, biomagnify, and can be toxic. The legacy organochlorines (e.g. PCBs and DDTs) and the more recent POPs of concern (e.g. PBDEs) have been well documented in the literature to pose a risk to many species. These pollutants are associated with reproductive impairment (Reijnders 1986; Subramanian et al. 1987; Reddy et al. 2001; Schwacke et al. 2002); immunotoxicity (de Swart et al. 1996; Fonnum et al. 2006); endocrine disruption (de Boer et al. 2000; Legler and Brouwer 2003; Darnerud 2008; Legler 2008); neurotoxicity (Darnerud 2003; Viberg et al. 2003; Viberg et al. 2006; Darnerud 2008); and cancer in humans and wildlife (Ylitalo et al. 2005; Bonefeld-Jørgensen et al. 2011).

Historically, the Hyperion effluent was one of the primary sources of PCBs and DDTs in the action area and these legacy pollutants continue to be measured today, although at relatively smaller concentrations compared to historical levels (EPA 2017). As described in the BE, the 2017 permit includes effluent limits and annual sediment monitoring for PCBs and DDT. Currently, the sediment concentrations of these two POPs across the action area exceed the TMDL targets (EPA 2017). The City of Los Angeles also monitors fish tissue (hornyhead turbot and other sportfish), and a fish consumption impairment exists for the entire action area. In fact, reduced thyroid production was found in horny-head turbot near the outfall and changes in gene expression when exposed to only 5% of Hyperion effluent (Bay et al. 2011; Maruya et al. 2011; Vidal-Dorsch et al. 2011). There are no temporal trends in the DDT and PCB data since 2002; however, there are spatial trends. PCBs in fish tissue were in highest concentration near the 5-mile outfall, whereas the highest DDT concentrations were found near the south end of the Bay. The potential for exposure to the legacy PCBs and DDT continue to be a concern for ESA-listed species. However, the majority of the exposure likely results from the historical contamination, and it is the persistence of these legacy pollutants that have caused the continuation of effects.

Although PBDEs have not been routinely monitored at the Hyperion WWTP, effluent from WWTPs in other regions has been identified as a major point source for PBDEs (North 2004; Ecology and King County 2011). Because PBDEs are currently found in significant and measurable amounts in wastewater effluent (EPA 2010), and ESA-listed species may receive the majority of PBDEs from their diet, it is important to understand how these pollutants from wastewater effluent move through the food web in order to estimate the potential for exposure of pollutants from the proposed action to ESA-listed species.

As an exercise to understand and isolate the effects of PBDE loading from the Hyperion WWTP, we estimated the cumulative difference in the total PBDE concentrations for the 5 years of discharged covered by this NPDES permit. However, total PBDE loads from Hyperion and other sources to Santa Monica Bay are currently unknown. It is also unknown to what extent PBDEs are distributed among the water column, food web, and sediment between the shelf and canyons in the action area. As described in the proposed action, to help fill this data gap, EPA and the CA State Water Resources Control Board agreed to add a special study to continue to examine PBDE concentrations in the effluent to evaluate the loadings in the Bay as part of the proposed action. In the meantime, to estimate the PBDEs in the Hyperion wastewater effluent, we identified three treatment plants (the Palo Alto WWTP, the Bremerton WWTP, and the Howard F. Curren Advanced Wastewater Treatment Plant [AWTP]) that have known PBDE levels in the effluent that we can use as surrogate data. North (2004) measured PBDEs in effluent from the Palo Alto WWTP, a source of PBDEs to the San Francisco Estuary. This plant is a tertiary WWTP much smaller in size than Hyperion, treating 25 MGD. The mean concentration of total PBDEs in the effluent was 29,023 pg/L, or an estimated total PBDEs mass loading of about 2 pounds (1 kg) per year in San Francisco Estuary (North 2004). Washington Department of Ecology and Herrera Environmental Consultants (2010) measured PBDEs in the effluent of the Bremerton WWTP, a source of PBDEs to Puget Sound, Washington. This plant has secondary treatment with an average maximum flow of 10 MGD and had a slightly less PBDE concentrations of 17,069 pg/L, or an estimate of 0.5 pounds (0.2 kg) per year. Following these studies, Siegel (2013) measured PBDEs at Howard F. Curran Advanced Wastewater Treatment Plant (AWTP) (also a tertiary plant) with 58 MGD. Siegel (2013) estimated approximately 0.3 pounds (0.1 kg) per year of PBDEs were loading into Hillsborough Bay, Florida from the Curran AWTP. This third study differed from previous studies in that only 7 PBDE congeners were analyzed as opposed to the 24-28 congeners analyzed at Palo Alto and Bremerton. Because this third study only analyzed 7 congeners, a lower estimate of PBDE mass loading was expected.

These three plants are substantially smaller in size and service than Hyperion. For example, the Palo Alto WWTP services approximately 220,000 residents in a low density residential housing area (Palo Alto Water Quality Control 2015). Because Hyperion is a large WWTP facility (servicing approximately 4 million residents), we assume that the PBDE concentrations and loadings are likely to be at least equivalent to what has been measured at these three plants. Therefore, we used the maximum concentration measurement (29,023 pg/L[13]) from these surrogates to estimate a range of the annual total PBDE concentration in the Hyperion effluent using the average design flow of 450 MGD and the peak hydraulic capacity of 720 MGD . As a result, we estimate a range approximately 40 – 62 pounds (18 – 28 kg) per year of total PBDEs may be loaded into Santa Monica Bay as a result of Hyperion's wastewater discharge. For the total 5 years of this proposed action, this equates to approximately 200 – 310 pounds (91 – 140 kg) for the permit cycle, a substantially larger total amount of PBDE loading than what is discharged from other smaller WWTPs. It is important to note here that actual wastewater flows have been substantially less than the average and maximum flows due primarily to drought conditions and conservation measures (EPA 2017). For example, Hyperion treated an average of 242 MGD between 2013 and 2014. Furthermore, it is unknown if the total PBDE loading into the marine environment is proportional to the treatment capacity (Gockel and Mongillo 2013). Therefore, we acknowledge using the hydraulic capacity of 720 MGD will likely result in a conservative estimate. To put this PBDE loading in context, relatively small concentrations of

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[13] pg (picogram) = 10^{-12} grams; or 1000000000000 pg to 1 gram

total PBDEs (ranging between 61 and 903 nanogram per gram of lipid[14]) in the blubber of grey seal (*Halichoerus grypus*) pups were significantly related to circulating thyroid hormone levels (Hall et al. 2003), suggesting that toxic effects may occur at concentrations in the body as low as one part per billion. This estimate of PBDE loading from the proposed action adds to the long-term accumulation the Bay has already experienced from historic discharges from past permits. Each additional permit following the current permit (which is reasonably likely to occur) will continue to add PBDEs into Santa Monica Bay and increase the cumulative difference in total PBDE loadings specific to Hyperion.

2.4.2.2 Adverse Health Effects from Exposure to Potentially Harmful Contaminants

2.4.2.2.1 Marine Mammals and Sea Turtles

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 their hydrophobic nature. In general, once the pollutants attach to these particles they may sink down in the water column and accumulate in the sediment, where the sediment acts as a sink and sequesters or buries contaminants rendering the POPs no longer readily available to organisms in the water column. However, the contaminated sediment can act as a source for benthic food webs and begin biomagnifying in the benthic food chain. Not all POPs accumulate in sediment, and some pollutants that enter the aquatic system may directly enter the pelagic food web. The proportional distribution of POPs in the local environment likely varies from site to site and is based on biotic and abiotic factors.

Recently, researchers from Washington Department of Fish and Wildlife have been tracking the movement of PCBs and other toxic chemicals in Puget Sound, WA and found comparatively lower levels of these POPs in the sediment, but higher levels in the resident pelagic species. For example, POPs in resident Pacific herring (*Clupea pallasii*) in Puget Sound can’t 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 suggests that many of the POPs in the water column don’t reach the benthos, but rather get picked up by bacteria or plankton which are then consumed by pelagic organisms, exposing the pelagic food web. This is likely an exposure route from POPs in Hyperion’s effluent to ESA-listed marine mammals and sea turtles, in addition to the deposition in sediments and the benthic food web.

Exposure to some of these contaminants does not need to occur in high concentration to be toxic, and has long been recognized as problematic (Carson 1962). There are currently no PBDE health-effects thresholds identified for marine mammals or sea turtles. However, relatively low PBDE concentrations have been associated with altered thyroid hormone levels in post-weaned and juvenile grey seals (Hall et al. 2003). Although it is important to keep in mind that the

[14] ng(nanogram) = 10^{-9} grams; or 1000000000 ng to 1 gram

effects due to PBDE exposure may potentially be species-specific, dose-dependent, and congener-specific, here we describe toxicology studies that examined effects to multiple species from PBDE exposure.

Similar to PCBs, PBDEs 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; Darnerud 2008; Legler 2008; Kodavanti et al. 2010). For example, some PBDE metabolites are structurally similar to thyroid hormones and these metabolites have disrupted the thyroid hormone homeostasis in laboratory species (Zhou et al. 2001; Zhou et al. 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).

Binding to thyroid receptors not only disrupts the transport of the hormones essential for brain development, but also transports PBDEs across the blood-brain and placental barriers in laboratory species (de Boer et al. 2000). PBDE exposure can also cause continuing behavioral alterations, and reduced learning and memory (Costa and Giordano 2007). Furthermore, young developing mammals may not be able to excrete PBDEs as efficiently as older individuals and may accumulate higher concentrations in the brain (Costa and Giordano 2007). The capacity for metabolic breakdown of PBDEs may also increase with age or with increasing concentrations (Weijs et al. 2009). It may also be possible that a contaminant-induced reduction of thyroid hormone levels has the potential to alter hearing and communication in mammals. For example, a 50-60 percent reduction of the thyroid hormone T4 in rats, which are commonly used as surrogates in medical health studies, during the postnatal period correlated with hearing loss in adults (Crofton 2004).

Endocrine disruptors can mimic or offset reproductive processes. Consequently, adverse reproductive effects have been associated with PBDE 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 this persistent pollutant and they caused permanent effects on reproductive processes.

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

While PBDEs 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 (AhR), generally described as "dioxin-like" effects. "Dioxin-like" contaminants are particularly effective at immunotoxicity across a range of species. PCBs, PBDEs, and DDTs have well documented effects on the immune system in a wide range of experimental animals (e.g., Thomas and Hinsdill 1978; Thomas and Hinsdill 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).

Less is known about early exposure of PBDEs to sea turtles. Recent studies have identified POPs transferred from nesting females to eggs and hatchlings likely have consequences on development. For example, POP concentrations in green sea turtles were significantly negatively correlated with body condition of hatchlings, an indication of effects on development (de Merwe et al. 2010). PBDEs 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, persistent pollutants (PCBs, PBDEs, and DDT) were associated with clinical health parameters (i.e., weight, carapace length, hematology, etc.; Keller et al. 2004; Komoroske et al. 2011; Swarthout et al. 2010; Camacho et al. 2013), fibropapilloma (Aguirre et al. 1994; Keller et al. 2014), hatchling mass and success (van de Merwe et al. 2010; De Andrés et al. 2016), lymphocyte proliferation (Keller et al. 2006), lysozyme activity (Keller et al. 2006), septicaemia (Orós et al. 2009), cachexia (Orós et al. 2009), and pansteatitis (Orós et al. 2013). Finlayson et al. (2016) also summarized the available *in vitro* and *in vivo* toxicity studies for sea turtles. Of the few available, they included alterations to immune response, alterations to sex determination processes and sex reversal, genotoxicity, endocrine disruption, metabolic disruption, and disruption of reproduction (see Finlayson et al.

2016 for a review). POPs may also have subtle effects on the development, size, and fitness of sea turtle eggs and hatchlings which is important for offshore dispersal, predator avoidance, and ultimately survival and population growth (van de Merwe et al. 2010; Keller 2013).

In addition to the legacy POPs (such as PCBs and DDT) and the relatively recent PBDEs, TBT also acts as an endocrine disruptor and has shown to competitively inhibit aromatase cytochrome P450 activity in humans (Heidrich et al. 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)

Mixture Effects and Non-Linear Dose-Response Curves.

Marine organisms are exposed to a number of toxic chemicals off California and the interactions of these chemicals have the potential to be additive (when the effects from two or more chemicals equal the sum of the effects of the isolated chemicals), synergistic (when the effects from the interaction is greater than the sum of the effects of the isolated chemicals), or antagonistic (when the effects from the interaction is less than the sum of the effects from the isolated chemicals). Although health risks are probably elevated as a result of interactions between toxic chemicals, and wildlife is rarely exposed to single compounds, the majority of studies have examined the effects of isolated chemicals. It has only been in more recent years that studies have examined health effects from exposure to mixtures of chemicals. For example, a few recent studies have highlighted the importance of evaluating mixture effects (Hallgren and Darnerud 2002; Crofton et al. 2005; Eriksson et al. 2006; Fischer 2008; He et al. 2009a, b, 2010). Mixture effects case studies that have examined effects from the interaction of PBDEs and PCBs (e.g. Eriksson et al. 2006; He et al. 2009 a,b; He et al. 2010) demonstrate that the interaction of these 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 such as PCBs and PBDEs, may underestimate risk (for a review, see Welshons et al. 2003) because some contaminants can interact at doses below the no observed effect concentrations (NOECs) and produce significant effects (Silva et al. 2002). For example, Crofton et al. (2005) tested the hypothesis that a mixture of thyroid hormone-disrupting chemicals has additive dose-response effects. They demonstrated that the effects from a mixture consisting of thyroid hormone disrupters can be additive at low doses and synergistic at high doses and more importantly, the highest mixture dose levels were at or below the NOECs of the chemicals. Endocrine disruptors, when isolated, have shown to produce nonlinear (e.g., U-shaped or J shaped) dose-response curves. For example, PBDE concentrations in the blubber of grey seals significantly contributed to circulating thyroid hormone concentrations (Hall et al. 2003). They found a positive association between PBDEs and circulating thyroid hormones, in contrast to several laboratory studies that have reported a negative correlation. Furthermore, the PBDE concentrations in the grey seals were at much lower doses than were used in laboratory studies, suggesting a hermetic dose-response (or an enhancement of the response at low doses and an inhibition at high doses). TBT can also act synergistically with a PCB congener (PCB-126) known to induce P450, and produce opposite effects than when the chemicals are isolated at higher doses. For example, female mice exposed to high doses of TBT combined with PCB-126 inhibited P450 activity, whereas low doses of TBT combined with the PCB congener enhanced the activity (DeLong and Rice 1997).

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

Summary

The Hyperion effluent contains potentially harmful contaminants that have been well established to adversely affect laboratory and wildlife species. PCBs and DDT have been measured in the action area and effluent and trends are declining from historical levels, however, the levels still pose a threat. TBT is considered less of a threat although it has been measured in the effluent but at lower levels. Less is known about PBDE levels in the action area and the effluent. Based on surrogate data from other WWTPs, we estimate approximately 40 pounds (18 kg) per year of total PBDEs may be loaded into Santa Monica Bay as a result of Hyperion's wastewater discharge. Over the course of the 5 years of this proposed action, this equates to approximately 200 pounds (91 kg) for the total permit cycle. Once in the aquatic system, these PBDEs attach to particles and become bioavailable to food webs.

ESA-listed marine mammals and sea turtles are affected by the proposed action indirectly by consuming prey that has accumulated POPs from Hyperion's effluent, which expedites the potential or timing for adverse health effects in ESA-listed marine mammals and sea turtles feeding in the action area to occur. Although baleen whales and sea turtles consume prey at lower trophic levels, and their total body burdens are relatively less than other species, endocrine disruptors do not necessarily need to be in high concentration to cause an effect. Furthermore, there may be synergistic effects between PBDE and PCB congeners likely increasing the health risk to the marine mammals and sea turtles. Thus, increasing PBDE levels in the ESA-listed species only further exacerbates their current susceptibility to adverse health effects including effects to an individual's reproductive, endocrine, and immune systems. As described above, we expect that all of the ESA-listed marine mammals and sea turtles species that may occur in the action area have individuals that may make numerous or possibly frequent and extended visits to the Bay 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 2.3 *Environmental Baseline*, there are numerous other potentially harmful contaminants for ESA-listed marine mammals and sea turtles, and many of those may also be present in Hyperion's discharge. However, there is limited knowledge and available information describing the levels of most of these contaminants in wastewater discharge and the extent of potential harmful effects. As described in section 1.3.4 *Special Studies*, EPA is requiring Hyperion to conduct a special study that includes developing monitoring programs for some of the CECs that may be harmful to ESA-listed species.

2.4.2.2.2 Abalone

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

Field studies in Japan have found that exposure of *H. madaka* and *H. gigantea* to the organotin compounds TBT and triphenyltin (TPhT) (found in anti-fouling paint) caused ovarian spermatogenesis (masculinization) and altered the timing of reproductive maturity in males and females (Horiguchi et al. 2001, 2005). Studies have also found adverse effects on larval abalone behavior and development from TBT (e.g., Horiguchi et al. 1998). 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). The concentrations of TBT in the muscle tissue were similar to the concentrations of butyltins (including mono, di, and tri-butyltin) measured in red abalone sampled in Monterey Harbor in 1999 (Kannan et al. 2004). Studies have also examined effects of potentially harmful contaminants on abalone 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.

These potentially harmful contaminants have been detected in Hyperion's effluent, but generally at lower levels than those described above (EPA 2017). In 2012-2016, TBT was detected, but not quantifiable in the effluent. In 2012-2014, measured levels of triclosan ranged from 200 to 380 ng/L (including non-detectable levels in 2014) and levels of bisphenol A ranged from 70 to 540 ng/L; TPhT and diallyl phthalate were not included in the list of CECs monitored. However, these represent only a handful of the more persistent and potentially harmful contaminants that can be found in Hyperion's discharge effluent. We do not know how many of these potentially harmful contaminants may affect abalone. In general, what these studies and their results show is that exposure to endocrine disruptors and other chemicals, including those found in Hyperion's discharge effluent, can have a harmful effect on abalone growth and reproductive development. Further studies are needed to evaluate the concentrations of potentially harmful contaminants in the effluent, the concentrations to which white abalone and black abalone may be exposed, and the effects of these potentially harmful contaminants on abalone at those concentrations. As described in section 1.3.4 *Special Studies*, EPA is requiring Hyperion to conduct a special study that includes developing monitoring programs for some of the CECs and other potentially harmful contaminants that may be harmful to ESA-listed species.

2.4.3 Harmful Algal Blooms

The discharge of effluent contributes additional nitrogen and other nutrients to Santa Monica Bay (e.g., Reifel et al. 2013; Howard et al. 2016; McLaughlin et al. 2016), promoting HABs that potentially pose a threat to ESA-listed species.

2.4.3.1 Effect of Hyperion's Discharge on HAB Occurrence

The Hyperion discharge may have the effect of fertilizing or kick-starting the spring time HABs by sustaining or even increasing populations of HAB species in the subsurface water (Cochlan et al. 2008; Kudela et al. 2010; Seeyave et al. 2009; Seegers et al. 2015; Trainer et al. 2007) and by providing nitrogen to the upper water column of Santa Monica Bay when stratification is weak or shallow. Concentrations of nitrogen (and phosphorus) in the effluent plumes are up to three orders of magnitude greater than maximal ambient concentrations and they also entrain deeper, nutrient laden water as they rise through the euphotic zone to the thermocline or to the surface when stratification is weak (Seegers et al. 2015; Reifel et al. 2013). These conditions can commonly occur during the winter months in the Bay. HAB species (*A. catenella*, *P. spp*.; Seegers et al. 2015; Trainer et al. 2010) 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 at the Hyperion 5-mile outfall as well as in the Santa Barbara Channel and elsewhere in the SCB with limited exchange of water.

The physical oceanography in the vicinity of the Hyperion discharge influences the fate and transport of the nutrients and any subsequent phytoplankton or zooplankton that utilize the nutrients to grow. Given that Santa Monica Bay experiences a frequent eddy pattern due to the CA Current and the Southern CA Counter Current found just offshore of the shelf area during times of the year (Howard et al. 2012), nutrients from Hyperion may remain within the Bay due to the gyre and become more concentrated over time. Nezlin et al. (2012) and Trainer et al. (2010) both identified the Bay as a hot spot area with longer residence time of its water and higher CHL-α levels. Additional nutrients may enter the Bay from the south due to the Southern CA Counter Current (Howard et al. 2012) and from winter runoff that is not captured and treated at Hyperion. While large winter storms likely serve to mix and exchange retained nutrients within the Bay, several studies also noted that water may travel north from the Bay into the Santa Barbara Channel when the Southern CA Counter Current is close enough to shore to drive this process (Anderson et al. 2006; Howard et al. 2012). The Santa Barbara Channel also experiences frequent algal blooms, and the nutrients and resulting phytoplankton species from Hyperion may be contributing to this phenomenon.

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. 2014, 2012). 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 (Booth 2015; Howard et al. 2012; Reifel et al. 2013; 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) (Schnetzer et al. 2013; Anderson et al. 2006). The discharge of large amounts of nitrogen in the effluent has the effect of unbalancing these ratios at the local level and may be partially responsible for the very potent HABs that have been occurring in the spring of many years in Santa Monica Bay. In 2014, Howard et al. estimated that Hyperion's nutrient loading of Santa Monica Bay increased total nitrogen (N) in the Bay by about 9,900 kg of N per km^2 per year. Given the area of Santa Monica Bay (1571 km^2), this amounts to about 15.6 million kg of N over the course of a year. As described in section 2.3 *Environmental Baseline*, this amount of nitrogen is roughly equivalent to the nitrogen brought into the Bay from coastal upwelling.

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. autralis* (Howard et al. 2007). Urea is a minor component in WWTP discharges in the SCB in contrast to riverine runoff where organic nitrogen forms are much more prevalent (35% of total nitrogen) (Howard et al. 2012). During wet years, urea entering the Bay from the land will be in the less dense, freshwater runoff and this becomes a reservoir of nutrients that influence HAB formation and toxicity when the subsurface species partially sustained by the Hyperion outfall are advected into the surface waters close to shore.

2.4.3.2 Potential Adverse Health Effects from Exposure to HABs

2.4.3.2.1 Marine Mammals and Sea Turtles

The potential for exposure of ESA-listed species to biotoxins such as those present in HABs is dependent on the co-occurrence of the harmful taxa present in the action area and ESA-listed species and/or their prey species. If a HAB occurs and exposes the food web (e.g., plankton, small fishes) in an area where ESA-listed species occur, then there is an increased likelihood for ESA-listed species to be exposed to any biotoxins produced. Because the majority of life in the Bay depends on phytoplankton, the risk of exposure of ESA-listed species and/or any prey species that may rely on phytoplankton (e.g., northern anchovies, pacific sardines, Pacific mackerel) to any HABs in the Bay is inherently present. As described above, it is likely Hyperion's effluent could function as a seed and kick starting HABs. The Southern California Coastal Ocean Observing System (SCCOOS) tracks HABs in and around the action area. Based on the frequent bloom events along the coast, it is likely that HABs could occur in and around the action area during the 5 years of the permit. Although uncertain what degree Hyperion's effluent would play in any HAB, NMFS anticipates the Hyperion effluent would provide conditions that help encourage a HAB during this 5 year proposed action and in future actions. Therefore, the foraging ESA-listed marine mammals and sea turtles nearby (both inshore and offshore species) could be at increased risk of exposure to biotoxins.

Four classes of marine algal toxins have been associated with marine mammal mortality and morbidity events. These include saxitoxin, brevetoxin, ciguatoxin, and domoic acid (Van Dolah et al. 2003). Between 1978 and 2006, there were 57 of these mortality events detected nationally by the NOAA Fisheries Stranding Network. Of those events, 29 were declared unusual mortality events (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 63 UMEs (Figure 3). Of the 63 UMEs, 19% have been caused by biotoxins from HABs (Figure 3). 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. Since 1996, UMEs associated with biotoxins have become more prevalent; the majority being attributed to toxicity from domoic acid and brevetoxin.

Of the four biotoxins, saxitoxin and domoic acid can occur in the action area because of the harmful taxa present. Recently, Corcoran and Shipe (2011) sampled plankton in multiple sites throughout the Bay. They found that at each site, potentially harmful plankton taxa were present. For example, the harmful diatom *Pseudo-nitzchia*, which can produce domoic acid, was observed at 7 out of the 9 sampling locations. During the last decade, the Los Angeles area has been identified as a hot spot for the domoic acid (Schnetzer et al. 2007). The dinoflagellates *Procentrum* and *Akashiwo sanguinea*, which can produce saxitoxin, are also present (Corcoran and Shipe 2011).

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

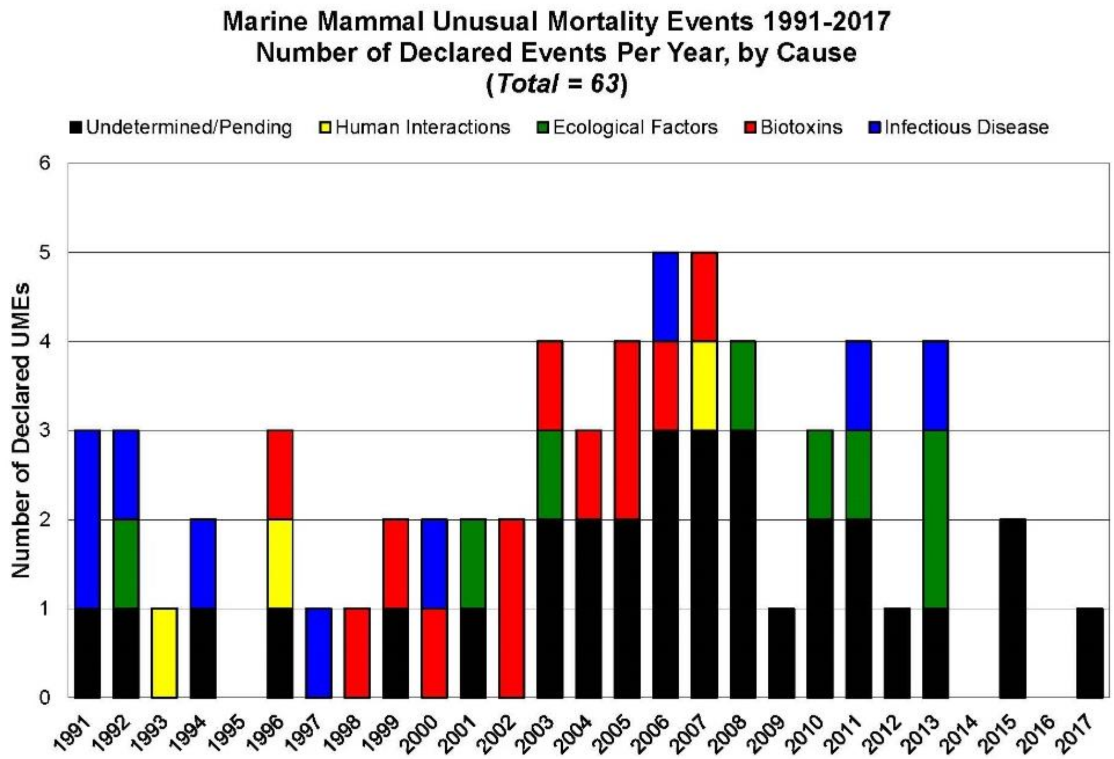

Figure 3. Number of unusual mortality events between 1991 and 2017 per year and by cause (figure reprinted from http://www.nmfs.noaa.gov/pr/health/mmume/events.html).

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

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

In general, large mammals are generally more sensitive to bioactive compounds so extrapolation from human studies is not appropriate for saxitoxins (Stoskopf et al. 2001). The increased vulnerability for humpback whales (and likely other large whales) that were exposed to saxitoxin off Massachusetts in 1987 and 1988 may be due to the fact that a larger proportion of their body weight is blubber (Geraci et al. 1989). Because saxiton 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 whales' 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). It was noted the sea lions were all 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 the 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, records have indicated gray whales have been observed feeding off California and Washington (Van Dolah et al. 2003). Krill were also collected offshore of Monterey Bay following the bloom and identified as a potential vector for domoic acid to higher trophic level species (Bargu et al. 2002). Based on the maximum domoic acid concentrations measured in krill, Bargu et al. (2002) estimated krill could transfer domoic acid levels up to 62 grams to a blue whale per day, or 0.62 mg per kg.

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

(*Grampus griseus*), Cuvier's beaked whale (*Ziphius cavirostris*), gray whale, and a humpback whale (Torres de la Riva et al. 2009).

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

HABs are also known to be capable of effecting sea turtles. Brevetoxins are produced by the dinoflagellate *Karenia brevis* and best known to be responsible for Florida red tides. Between 2005 and 2006, 318 documented sea turtle strandings off of Florida's Gulf of Mexico increased four-fold. 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 and it was not detected in feces at or above 500 parts per billion. They suggested jellyfish might not concentrate domoic acid as readily as forage fish. However, they did detect trace levels of domoic acid in the urine of a fresh dead leatherback that was struck by a propeller off the coast of California in 2008. Harris et al. (2011) emphasized that the potential for domoic acid exposure cannot be ruled out. In their study, they tested for domoic acid in plasma and feces, but suggested urine and stomach contents are likely better samples for evaluating exposure (Tor et al. 2003).

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

Summary

Similar to the analysis of potential effects of adding potentially harmful contaminants like PBDEs to the environment and increasing the accumulation of these contaminants by ESA-listed species, we conclude that the discharge of effluent by Hyperion can potentially increase the frequency and/or extent of HABs. At this time, we cannot predict the precise extent that

Hyperion's effluent discharge contributes to increased probabilities of HABs, or distinguish which HABs may be more or less associated or influenced by the additional nutrient input created by Hyperion's discharge. What is clear is that HABs pose a significant health risk for ESA-listed marine mammals and sea turtles; that increasing the probability of HAB occurrence further increases the likelihood of adverse effects from HABs that include impaired health and mortality, and that Hyperion's discharge increases the possibility of this occurrence. As described above, we expect that all of the ESA-listed marine mammal and sea turtle species that may occur in the action area have individuals that may make numerous or possibly frequent and extended visits to the Bay and be exposed to increased frequency or extent of HABs during those visits, increasing the risks of adverse effects that HABs are known to present.

2.4.3.2.2 Abalone

Documented abalone mortality events have been linked to HABs, including two recent events along the California coast. In 2007, a *Cochlodinium* bloom killed red abalone at the Monterey abalone farm by causing gill damage and reduced dissolved oxygen levels (Wilkins 2013; Howard et al. 2012). Also 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. 2014). In surveys conducted during the mortality event, an average of 25% of the red abalone observed were dead or dying (Rogers-Bennett et al. 2012). 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 20m), with the highest percent mortalities observed at 0 to 5m depth (approximately 20-75% mortality) and lower percent mortalities at 10 to 20 m depth (less than 10% to nearly 30%) across the four surveyed sites (DeWit et al. 2014). 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). *Cochlodinium* is a dinoflagellate that has been associated with Hyperion's discharge effluent (Howard et al. 2012; Reifel et al. 2013) and could affect white abalone and black abalone within Santa Monica Bay. Reifel et al. (2013) documented a bloom of *Cochlodinium* (the dinoflagellate linked to the 2007 abalone mortality event in Monterey) following discharge from the 1-mile outfall. Based on operations in the past 5 years, it is likely that over the duration of the 5-year permit, effluent will be discharged from the 1-mile outfall which could cause another bloom of *Cochlodinium*. As explained above, we conclude that the discharge of effluent by Hyperion can potentially increase the frequency and/or extent of HABs and the risk of adverse effects they pose for abalone. Because black abalone occur in nearshore, shallow waters, they may be relatively more at risk to the effects of HABs than white abalone. Other HAB related toxins, such as domoic acid and saxitoxins, have been detected in abalone tissues (Shumway 1995; Harwood et al. 2014; Malhi et al. 2014), but the effects on abalone health are not known.

2.4.4 Risks to Populations

2.4.4.1 Marine Mammals and Sea Turtles

In summary, the discharge effluent poses a risk to ESA-listed marine mammals and sea turtles via exposure of individuals to pollutants in the effluent and plume and/or to the increased frequency or extent of HABs promoted by the effluent. The concentrations of metals and most other potentially toxic constituents in the discharge effluent plume are expected to be lower than those typically expected to cause harmful effects for more sensitive species, and do not pose much of a threat for direct uptake from the water column or bioaccumulation through the food chain for ESA-listed marine mammals and sea turtles. On the other hand, studies confirm that marine mammals in particular, and likely sea turtles as well, are susceptible to endocrine disruption and harmful effects from POPs and other potentially harmful constituents that are known or expected to be found in the effluent (e.g., PBDEs), and the proposed action is likely to increase the body burdens of these contaminants and potentially expedite diminished health and fitness. However, further studies are needed to evaluate the levels of potentially harmful contaminants found in the effluent and their effects on ESA-listed marine mammals and sea turtles, as well as other marine species. Finally, HABs have been documented to cause mortality and other health issues in marine mammals along the California coast. The potential increase in frequency and/or extent of HABs due to the discharge effluent poses an increased risk of mortality for marine mammals especially, and possibly sea turtles as well. Further studies are needed to evaluate the composition, frequency, and extent of HABs that occur in the Bay, and better understand the association of these events with the discharge effluent in order to more completely assess the potential exposure of ESA-listed marine mammals and sea turtles to these blooms as a result of Hyperion's continued discharge. Based on our analysis, we conclude that exposure to the discharge effluent and potential environmental effects from it have the potential to reduce the fitness and survival of any ESA-listed marine mammals or sea turtles that may occur in the action area; an effect that we cannot discount or dismiss as insignificant.

However, it is difficult to assess how these potential impacts are expected to affect ESA-listed marine mammals and sea turtles at the population and species level given the available information. Long term effects for individuals, including diminished reproductive capacity and lower survival rates, could result from continued accumulation of potentially harmful contaminants, which is further accelerated by the proposed action. More acute effects, such as physical impairments, reduced foraging, disorientation, and even death are possible effects of exposure to HABs, which may be increased by the proposed action. These affects are likely to reduce the abundance of ESA-listed populations directly through removals, and indirectly through lost reproductive capacity or success. Although we expect that individuals from these ESA-listed species will be exposed to Hyperion's discharge given that we know individuals of these species visit the Bay, we generally are unable to describe the extent of exposure across the entire population that may be affected, or specifically track the level of exposure or response from any individual while it is within the action area or after departure. Given the transitory nature of most of these species, and the broad distribution of them in the Pacific, exposure to the proposed action in the Bay is likely somewhat limited at the population scale to relatively small segments of populations that may occasionally visit or favor the Bay as opposed to large proportions or entire populations. There may be some exception to this premise for a small population 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 Bay occasionally and be exposed. But with the available information, it isn't possible to further describe exactly how many individuals or what percentage of any of these populations will be exposed or potentially affected. In addition, as explained above, the extent of effects that can be expected at the individual level are also highly uncertain, making it difficult to anticipate what the population level effects may be.

2.4.4.1 Abalone

In summary, the discharge effluent poses a risk to white abalone and black abalone via exposure of individuals to potentially toxic pollutants in the effluent, accumulation of more persistent pollutants in the effluent, and/or exposure to increased frequency or extent of HABs promoted by the effluent. The concentrations of heavy metals in the effluent and plume are expected to be lower than those found to cause harmful effects on other abalone species; however, there is uncertainty regarding the species' ability to acclimatize to heavy metal exposure, as well as regarding the synergistic effects of exposure to multiple metals in the effluent and plume. In addition, studies confirm that abalone are susceptible to endocrine disruption and harmful effects from more persistent contaminants, such as those found in the effluent, although further studies are needed to evaluate the levels of these contaminants found in the effluent and their effects on abalone. Finally, HABs have been documented to cause abalone mortality along the California coast. The potential increase in frequency and/or extent of HABs due to the discharge effluent poses an increased risk of mortality for abalone. Further studies are needed to evaluate the composition, frequency, and extent of HABs associated with the discharge effluent and the potential exposure of white abalone and black abalone to these blooms. Based on our analysis, we conclude that exposure to the discharge effluent and its effects have the potential to reduce the fitness and survival of juvenile and adult white abalone and black abalone.

For both white abalone and black abalone, the major threat is the low density and spatial distribution of remaining animals in the wild. Any further reduction in reproductive capacity or survival of individuals would pose a threat to the population. Remaining animals may not be close enough in proximity to reproduce successfully or at levels needed to support recovery. The potential increased risk of mortality due to HABs could further reduce the number of individuals in the wild and the population's ability to reproduce. In addition, the potential masculinization of females and disruption to synchronous gonad maturation cycles would further reduce the population's ability to reproduce successfully or at levels needed to support recovery. We also note that effects could occur to white abalone within the ZID on the outfall structure (discussed section 2.4.1.2 above), but that the ZID consists primarily of soft-bottom habitat that would not be suitable for abalone but for the presence of the outfall. Thus, the settlement and survival of abalone within the ZID would not be considered a major component of the species survival or recovery.

For white abalone, the population(s) within the action area is important for the species survival and recovery. We do not have much information regarding white abalone within Santa Monica Bay, except for the area off Palos Verdes. The existing population off Palos Verdes represents one of the few known, remaining populations of white abalone along the Southern California coast. This population has provided several broodstock for the captive breeding program, inserting new genetic diversity into the captive population. In addition, this area is one of a few areas along the Southern California coast where outplanting of white abalone may be conducted, for the purpose of re-establishing and/or enhancing white abalone populations in the wild. Outplanting may involve larval and/or juvenile stages of white abalone. Palos Verdes is also one of the areas where adult broodstock may be reintroduced to the wild and aggregated to increase reproductive potential. Thus, the Palos Verdes area is important for the recovery of the species. Because this is a potential outplanting site, it will be important to understand how the discharge plume may be affecting water quality and white abalone in this area, to ensure the best conditions for the survival and growth of outplanted larval and juvenile abalone. Palos Verdes is located about 10 miles away from the 5-mile discharge; and although monitoring suggests that the discharge plume can be occasionally detected in that area, it is likely that concentrations of effluent constituents that could occur there would be at very low levels. However, as described above, low concentrations of some persistent constituents that can be accumulated may still present risks for white abalone in that area. Further studies are needed to evaluate the levels of CECs within the effluent and plume, the effects of these CECs on abalone at those concentrations, and how the effluent affects the frequency and extent of HABs within Santa Monica Bay.

For black abalone, the importance of the population(s) within the action area for the species survival and recovery is less clear. Compared to white abalone, much more information is available on black abalone population status throughout the California coast, due to long-term monitoring data that has been collected in some areas since the mid-1970s. The populations at Palos Verdes and Malibu represent a small portion of the populations along the Southern California mainland coast. Although recovery of the species at all historically occupied sites throughout its range would be ideal, this may not be a realistic goal, nor may it be needed to recover the species. Along the Malibu coast and Palos Verdes coast, black abalone have been observed at other sites upcoast and downcoast adjacent to the action area. Continued adverse impacts to the populations within Santa Monica Bay may not substantially hinder recovery of the species, although information is currently lacking on population connectivity. More information is needed on population connectivity along the coast to better evaluate the importance of these populations to the species as a whole.

In summary, the potential effects of the discharge plume on individual abalone growth, reproductive development, and survival would also affect the fitness of white abalone and black abalone populations. Because the main threat to the species is low reproductive potential due to low numbers and densities in the wild, the potential increase in mortality and reduction in reproductive capacity caused by the effects of the discharge pose a risk to the populations. For white abalone, the importance of the Palos Verdes population as a source of broodstock for the captive population and as a potential outplanting site means that substantial impacts to this population could affect the species' recovery. For black abalone, the lack of recovery of these populations in the Bay may not substantially affect species recovery, but more information is needed regarding population connectivity throughout the coast. We note that these effects at the population and species level are based on certainty of effects at the individual level; however, we have a high degree of uncertainty regarding effects at the individual level and this uncertainty is multiplied as we scale up to effects at the population and species levels. Overall, we cannot conclude that the effects of the discharge on white abalone and black abalone are discountable or insignificant. We can describe the potential effects on abalone at the individual level, but with a high degree of uncertainty about what the exposure and the extent of actual effects at the individual level will be making it difficult to anticipate what the population level effects may be. Further studies, as described above, are needed to address this uncertainty and better understand the effects at the individual level, and thus at the population and species levels.

2.5 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). 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 the overall environmental health and habitat quality within the action area. However, it is difficult if not impossible to distinguish between the action area's future environmental conditions that are properly part of the environmental baseline *vs.* cumulative effects. In section 2.3 *Environmental Baseline*, we described the current and ongoing impacts associated with other discharges and other activities that affect water quality in Santa Monica Bay. We are reasonably certain that these activities and impacts will continue to occur and remain in place during the full extent of this proposed action.

We did not identify additional state or private activities that are reasonably certain to occur within the action area, do not involve Federal activities, and could result in cumulative effects to ESA-listed species and designated critical habitat within the action area. Numerous restoration and monitoring activities that have been initiated through the SMBRC Bay Restoration Plan (BRP), that are designed to promote and improve water quality and habitat improvement within the Bay are expected continue in the future (SMBRC 2013). While the BRP spells out numerous goals and objectives that will guide future efforts to improve the health of Santa Monica Bay, it is difficult to pinpoint any specific activity or expectation for how the health of Santa Monica Bay will change over the course of the next 5 years as a result of the BRP and efforts of the SMBRC and affiliated institutions in a way that could influence any of the potential effects of this proposed action beyond what has already been considered in this opinion. Oil spills and the introduction of other pathogens and parasites could occur within the time frame of the permit and could affect ESA-listed species and designated critical habitat within the Bay. However, the potential effects are difficult to evaluate at this time, given the unpredictability and uncertainty in the timing, location, scope, and severity of such events. Spills can result in very different effects depending on many factors, including the type of oil, the amount, local conditions, and the location. In addition, although we have examples of how other pathogens have affected ESA-listed species (e.g., abalone) 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.

2.6 Integration and Synthesis

The Integration and Synthesis section is the final step in our assessment of the risk posed to species and critical habitat as a result of implementing the proposed action. In this section, we add the effects of the action (section 2.4) to the environmental baseline (section 2.3) and the cumulative effects (section 2.5), 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: (1) Reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) appreciably diminishes the value of designated or proposed critical habitat for the conservation of the species.

We aggregate the *Integration and Synthesis* across species groups (e.g., marine mammals, sea turtles, abalone) 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.

2.6.1 Marine Mammals and Sea Turtles

As described in section 2.4 *Effects Analysis*, 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 Hyperion's effluent discharge plume as a result of occasional exposure to them when foraging in the Bay, due to limited exposure to concentrated amounts of these constituents and/or minimal risks the exposure may pose to their health. However, as described in section 2.3 *Environmental Baseline* and section 2.4 *Effects Analysis*, 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, especially PBDEs, and individuals of these species may be already carrying loads of potentially harmful contaminants into the action area before exposure (or as a result of previous exposure) to the proposed action that could already be compromising overall health and fitness. As described in section 2.3 *Environmental Baseline* and section 2.4 *Effects Analysis*, we recognize that Hyperion's discharge may contain numerous other contaminants that could potentially harm ESA-listed species, but that the available information limits our ability to analyze those impacts further. As described in section 2.4 *Effects Analysis*, we expect the proposed action will increase the amount of PBDEs (and other potentially harmful contaminants) that are released into the environment, ultimately increasing or expediting the accumulation of these potentially harmful constituents within ESA-listed marine mammals and sea turtles feeding in the action area, increasing the potential for and pace where adverse health effects to these species can occur. As described in section 2.4 *Effects Analysis*, the occurrence and magnitude of exposure and adverse effects that we expect as a result of the discharge of potentially harmful contaminants is uncertain, given that levels of PBDEs and other potentially harmful contaminants in the effluent aren't currently monitored, and the variable potential exposure and response of individuals to the proposed action.

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

As described in section 2.3 *Environmental Baseline* and section 2.4 *Effects Analysis*, 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 the result of exposure to HABs, including HABs that may occur in the action area. Also, as described in section 2.4 *Effects Analysis*, the proposed action increases the probability of HABs occurring within the action area, increasing the probability that diminished health, reduced fitness, and even mortality, of ESA-listed marine mammals and sea turtles that occasionally occur within the action area can occur. As described in section 2.4 *Effects Analysis*, we don't have a precise understanding of how much Hyperion's discharge may increase the probability of HABs in the Bay, or a way to assess if particular blooms are associated with the proposed action and the nutrient input created by Hyperion's discharge. The proposed action includes initiation of a special study to better understand the nitrogen dynamics of Hyperion's discharge and the nitrogen loading that results from Hyperion's discharge into the Bay. This special study will improve our understanding of the proposed action's contribution to nutrient loading and HABs in the Bay. We also expect this study will help initiate efforts by EPA and the City of Los Angeles to investigate measures to minimize the discharge of nutrients that may increase the probability of HAB occurrence in the Bay during future permit actions.

In the meantime, as a result of uncertainty associated with these two potential avenues for adverse effects at an individual level at this time, we are also uncertain as to the relative occurrence and magnitude of the impact of these adverse effects on the total populations of the ESA-listed marine mammals and sea turtles that may be exposed to the proposed action. As described in section 2.4 *Effects Analysis*, 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; more likely to occur for individuals that may have some preference or site fidelity for the Bay. Although there is uncertainty in the extent of population level of exposure for some smaller populations, at this time we generally do not reasonably 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 2.3 *Environmental Baseline* and section 2.5 *Cumulative Effects*, we anticipate that most of the factors that have been affecting the quality and health of Santa Monica Bay's environment are likely to continue into the future over the duration of the proposed permit as potential threats to the health of ESA-listed marine mammals and sea turtles that may visit the Bay. Similarly, we expect the contributions of Hyperion's discharge to the overall health of the Bay, and to the health of ESA-listed marine mammals and sea turtles to persist as threats to the Bay as a whole and to ESA-listed species at an individual level. Climate change could influence migrations and distributions of prey and the relative exposure of various individuals and ESA-listed populations within the action area, as well as the probability or magnitude of HAB occurrence in the action area over time, but is unlikely to factor into the 5-year proposed action time frame considered in this opinion. There is substantial uncertainty in the specific occurrence and magnitude of expected impacts based on the available information. As a result, additional information is needed regarding the levels of PBDEs and other potentially harmful constituents in the effluent and their effects on ESA-listed marine mammals and sea turtles, and the effects of the effluent on the frequency and extent of HABs within the Bay that may harm ESA-listed marine mammals and sea turtles, to provide better understanding of these potential impacts and inform future analyses.

2.6.1.1 Blue Whale

Over the course of the proposed action, we anticipate that some individual blue whales may occasionally enter the Bay 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 possible mortality. Although the ENP stock of blue whales is relatively small (1,647 individuals), 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 blue whale species. At this time, the scientific information needed to more fully evaluate the anticipated outcomes from exposure of blue whales to Hyperion's discharge at an individual and population level is needed. EPA has proposed to require monitoring and studies under the NPDES permit to address key questions regarding the impact of the Hyperion's discharge on the Bay and any blue whales that may occur there will allow for improved evaluation of the impacts in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits. Based our current understanding of potential effects, as well as uncertainties in the possible magnitude and extent of those effects, and the measures that have been proposed to address these uncertainties and prospects for development of actions to minimize the extent of impacts in future consultations based on this information, we do not expect the potential effects of the proposed action to reduce the likelihood of survival and recovery of blue whales.

2.6.1.2 Fin Whale

Over the course of the proposed action, we anticipate that some individual fin whales may occasionally enter the Bay 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 possible mortality. The CA/OR/WA stock of fin whales is estimated to be 9,029 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, the scientific information needed to more fully evaluate the anticipated outcomes from exposure of fin whales to Hyperion's discharge at an individual and population level is needed. EPA has proposed to require monitoring and studies under the NPDES permit to address key questions regarding the impact of the Hyperion's discharge on the Bay and any fin whales that may occur there will allow for improved evaluation of the impacts in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits. Based our current understanding of potential effects, as well as uncertainties in the possible magnitude and extent of those effects, and the measures that have been proposed to address these uncertainties and prospects for development of actions to minimize the extent of impacts in future consultations based on this information, we do not expect the potential effects of the proposed action to reduce the likelihood of survival and recovery of fin whales.

2.6.1.3 Humpback Whale; Mexico DPS and Central America DPS

Over the course of the proposed action, we anticipate that some individual humpback whales may occasionally enter the Bay 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 Bay. These individuals will be at increased risk of diminished health and fitness, and even possible mortality. The Mexico DPS is estimated to be at least 6,000 individuals, and it is most likely (up to a 90% chance) that any individual present in the Bay belongs to the Mexican DPS. The Central America DPS is much smaller; estimated to be only at least 400 individuals. While it is less likely that any given individual that may be present in the Bay will be a Central America DPS whale (up to a 20% chance), most all Mexico DPS and Central America DPS humpback whales could occur in the action area given their general migratory movements along the U.S. west coast . Although we expect that the relative exposure of humpback whales to the proposed action may be limited to a relative small number of humpback whales, it is possible that a relative larger proportion of the Central America DPS could be affected.

At this time, the scientific information needed to more fully evaluate the anticipated outcomes from exposure of ESA-listed humpback whales to Hyperion's discharge at an individual and population level is needed. EPA has proposed to require monitoring and studies under the NPDES permit to address key questions regarding the impact of the Hyperion's discharge on the Bay and any humpback whales that may occur there will allow for improved evaluation of the impacts in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits. In addition, in the future NMFS will be developing further scientific information regarding the distribution of ESA-listed humpback whales, including any improved understanding about the potential exposure of Central America DPS humpback whales to actions throughout their range, including specifically their presence and abundance in the SCB.

Based our current understanding of potential effects, as well as uncertainties in the possible magnitude and extent of those effects, and the measures that have been proposed to address these uncertainties and prospects for development of actions to minimize the extent of impacts in future consultations based on this information, we do not expect the potential effects of the proposed action to reduce the likelihood of survival and recovery of the Mexico DPS or Central America DPS of humpback whales.

2.6.1.4 Gray Whales; WNP Population

Over the course of the proposed action, we anticipate that some individual gray whales may occasionally enter the Bay 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 Bay could belong to the WNP population of gray whales were, but that it is likely that at least one or more WNP gray whales would enter the Bay during the proposed action and be at risk of diminished health and fitness, and even possible mortality. The WNP population of gray whales is very small (~140 individuals), although exposure to the proposed action will likely be extremely limited given their migratory behavior through such a small action area and limited potential for foraging to occur, as well as the limited number of WNP gray whales that may occur in the action area. At this time, the scientific information needed to more fully evaluate the anticipated outcomes from exposure of WNP gray whales to Hyperion's discharge at an individual and population level is needed. EPA has proposed to require monitoring and studies under the NPDES permit to address key questions regarding the impact of the Hyperion's discharge on the Bay and any WNP gray whales that may occur there will allow for improved evaluation of the impacts in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits. Based on our current understanding of potential effects, as well as uncertainties in the possible magnitude and extent of those effects, and the measures that have been proposed to address these uncertainties and prospects for development of actions to minimize the extent of impacts in future consultations based on this information, we do not expect the potential effects of the proposed action to reduce the likelihood of survival and recovery of WNP gray whales.

2.6.1.5 Guadalupe Fur Seal

Over the course of the proposed action, we anticipate that some individual Guadalupe fur seals may occasionally enter the Bay 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 possible mortality. The Guadalupe fur seal population is estimated to be 15,830, although exposure to the proposed action will likely be relatively limited to a small number of individuals and a small portion of the population. At this time, the scientific information needed to more fully evaluate the anticipated outcomes from exposure of Guadalupe fur seals to Hyperion's discharge at an individual and population level is needed. EPA has proposed to require monitoring and studies under the NPDES permit to address key questions regarding the impact of the Hyperion's discharge on the Bay and any Guadalupe fur seals that may occur there will allow for improved evaluation of the impacts in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits. Based our current understanding of potential effects, as well as uncertainties in the possible magnitude and extent of those effects, and the measures that have been proposed to address these uncertainties and prospects for development of actions to minimize the extent of impacts in future consultations based on this information, we do not expect the potential effects of the proposed action to reduce the likelihood of survival and recovery of Guadalupe fur seals.

2.6.1.6 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 Bay and be harmed by the proposed action. As described above, it is possible that some individual green turtles that spend significant amounts of time foraging in the SCB could make frequent or extended visits, or even take up residence in the Bay. These individuals will be at increased risk of diminished health and fitness, and even possible mortality. While there are no estimates for the total abundance of East Pacific green sea turtle DPS, the number of nesting females in one of the primary nesting areas exceeds 10,000 individuals. Although we expect that the relative exposure will be limited to only a small number of individuals and small portion of the DPS, green turtles are likely at an increased risk of exposure to the proposed action compared to other ESA-listed sea turtles.

At this time, the scientific information needed to more fully evaluate the anticipated outcomes from exposure of East Pacific green sea turtles to Hyperion's discharge at an individual and population level is needed. EPA has proposed to require monitoring and studies under the NPDES permit to address key questions regarding the impact of the Hyperion's discharge on the Bay and any East Pacific green sea turtles that may occur there will allow for improved evaluation of the impacts in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits. Based our current understanding of potential effects, as well as uncertainties in the possible magnitude and extent of those effects, and the measures that have been proposed to address these uncertainties and prospects for development of actions to minimize the extent of impacts in future consultations based on this information, we do not expect the potential effects of the proposed action to reduce the likelihood of survival and recovery of East Pacific green sea turtles.

2.6.1.7 Leatherback Sea Turtle

Over the course of the proposed action, we anticipate that some individual leatherback sea turtles may occasionally visit the Bay and be harmed by the proposed action. These individuals will be at increased risk of diminished health and fitness, and even possible mortality. 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 nesting females in western Pacific has been recently estimated at 2,600. Although we expect that the relative exposure will be limited to only a small number of individuals, which constitute only a portion of population that may be affected and the globally-listed leatherback sea turtle species, there is concern that the western Pacific population is in a state of decline and at high risk of going extinction. However, the SCB is not a primary location of foraging or known site fidelity for this species, and the risks of exposure of the population to this proposed action are relatively low.

At this time, the scientific information needed to more fully evaluate the anticipated outcomes from exposure of leatherback sea turtles to Hyperion's discharge at an individual and population level is needed. EPA has proposed to require monitoring and studies under the NPDES permit to address key questions regarding the impact of the Hyperion's discharge on the Bay and any leatherback sea turtles that may occur there will allow for improved evaluation of the impacts in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits. Based our current understanding of potential effects, as well as uncertainties in the possible magnitude and extent of those effects, and the measures that have been proposed to address these uncertainties and prospects for development of actions to minimize the extent of impacts in future consultations based on this information, we do not expect the potential effects of the proposed action to reduce the likelihood of survival and recovery of leatherback sea turtles.

2.6.1.8 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 Bay and be harmed by the proposed action. These individuals will be at increased risk of diminished health and fitness, and even possible mortality. 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 7,138. Our expectation is that the relative exposure of this population will be limited to only a small number of individuals (juveniles) and small portion of the DPS. At this time, the scientific information needed to more fully evaluate the anticipated outcomes from exposure of North Pacific Ocean DPS loggerhead sea turtles to Hyperion's discharge at an individual and population level is needed. EPA has proposed to require monitoring and studies under the NPDES permit to address key questions regarding the impact of the Hyperion's discharge on the Bay and any North Pacific Ocean DPS loggerhead sea turtles that may occur there will allow for improved evaluation of the impacts in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits. Based our current understanding of potential effects, as well as uncertainties in the possible magnitude and extent of those effects, and the measures that have been proposed to address these uncertainties and prospects for development of actions to minimize the extent of impacts in future consultations based on this information, we do not expect the potential effects of the proposed action to reduce the likelihood of survival and recovery of North Pacific Ocean DPS loggerhead sea turtles.

2.6.1.9 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 Bay and be harmed by the proposed action. These individuals will be at increased risk of diminished health and fitness, and even possible mortality. 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. Our expectation that the relative exposure of this population will be limited to only a small number of individuals and small portion of the population. At this time, the scientific information needed to more fully evaluate the anticipated outcomes from exposure of olive ridley sea turtles to Hyperion's discharge at an individual and population level is needed. EPA has proposed to require monitoring and studies under the NPDES permit to address key questions regarding the impact of the Hyperion's discharge on the Bay and any olive ridley sea turtles that may occur there will allow for improved evaluation of the impacts in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits. Based our current understanding of potential effects, as well as uncertainties in the possible magnitude and extent of those effects, and the measures that have been proposed to address these uncertainties and prospects for development of actions to minimize the extent of impacts in future consultations based on this information, we do not expect the potential effects of the proposed action to reduce the likelihood of survival and recovery of olive ridley sea turtles, most likely from Mexican nesting beach origins.

2.6.2 Abalone

As described in section 2.4 *Effects Analysis* and in this section in general (see above), the proposed action would allow Hyperion to continue discharging treated wastewater into the Bay over the duration of the five-year permit, resulting in continued exposure of ESA-listed abalone to the effluent and potential accumulation of some potentially harmful contaminants, and to HABs that may occur more frequently or over a larger extent due to the discharge. Potential effects on abalone include reduced survival due to exposure to harmful algal blooms, as well as reduced growth, reproductive development, and survival due to exposure to POPs, CECs, and other pollutants in the effluent.

In general, for the heavy metals and other constituents that have been evaluated for impacts to abalone, the levels that have been reported in Hyperion's effluent over the past five years are lower than the levels found to significantly reduce survival, growth, and/or reproductive development in abalone. However, 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. Past studies using Hyperion's effluent have only evaluated the effects of different concentrations on larval abalone, but not juveniles or adults. We expect that the continued discharge of effluent under the proposed action is likely to increase the uptake of potentially harmful contaminants such as POPs, increasing the risks of some level of harm to abalone by reducing the growth and reproductive capacity of individuals. However, the level of these sublethal effects on individuals and the population as a whole is highly uncertain.

As described in section 2.3 *Environmental Baseline* and section 2.4 *Effects Analysis*, the available information does not indicate evidence of a HAB-related abalone mortality event in the Bay, despite decades of discharge into the Bay; however, given the low numbers of abalone within the Bay and limited monitoring, abalone mortality events may have occurred without being observed. As explained in section 2.4 *Effects Analysis* and in this section in general, we don't have a specific understanding of how much Hyperion's discharge may increase the probability of HABs in the Bay, or a way to assess if particular blooms are associated with the proposed action. But we do not necessarily expect that all abalone in the Bay will be exposed to all HABs that occur within the Bay. If oceanographic conditions expose abalone to a HAB, then there is a reasonable potential for some abalone to die. Unless a large number die, these mortalities would likely go unobserved. Based on the best available information on past effects and the distribution of abalone in the Bay, 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.

Any abalone that may be present in the Bay have already experienced years of exposure to discharges from wastewater treatment plants (including Hyperion), stormwater runoff, and adjacent rivers. As described in section 2.3 *Environmental Baseline* and section 2.5 *Cumulative Effects*, over the five-year period, other facilities, adjacent rivers, and stormwater runoff will continue to discharge into the Bay, adding to the pollutant load to which abalone would be exposed. Exposure to warming water temperatures and ocean acidification will also continue, with uncertain effects on abalone health and survival. They have also been exposed to harmful algal blooms in the Bay, including natural-occurring blooms and those enhanced by discharges. How this past exposure to discharges and harmful algal blooms may have affected the status and recovery of abalone in the Bay is not known, although conditions have improved with implementation of full secondary treatment at Hyperion since 1998-1999.

In summary, the proposed action may adversely affect survival, growth, and reproductive development of abalone within the Bay, further exacerbating the risks of low density and reduced reproductive capacity for this population and adding to the ongoing effects of other discharges into the Bay, warming water temperatures, and ocean acidification, along with other threats such as disease, and poaching. There is substantial uncertainty in the specific occurrence and magnitude of expected impacts based on the available information. As a result, additional information is needed regarding the abundance and distribution of abalone in the Bay, the levels of more persistent and potentially harmful contaminants in the effluent and their effects on abalone, and the effects of the effluent on the frequency and extent of HABs within the Bay that may harm abalone, to provide better understanding of these potential impacts and inform future analyses.

2.6.2.1 White 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 2.3 *Environmental Baseline*, little information is available regarding the abundance and distribution of white abalone in the Bay, except for the area off Palos Verdes. Thus, the status of white abalone throughout the Bay is not clear, but they are likely at low densities and reduced reproductive capacity. The population off Palos Verdes is one of a few known populations along the Southern California coast and therefore is an important population to study to learn about the species' status and habitat needs in the wild. In addition, this population serves as a source for wild broodstock for the captive propagation program and the rocky reefs off Palos Verdes are also a potential outplanting site for white abalone recovery. Although effluent plume concentrations at Palos Verdes are expected to be very low, there is a risk of some exposure to small concentration of some potentially harmful pollutants from Hyperion's discharge, and to HABs in the Bay, in that location.

Of concern are the potential for the discharge to (a) increase the frequency and/or extent of HABs and thus increase the risk of mortality for individuals; and (b) increase the loading of persistent pollutants in the Bay, which can accumulate in white abalone and reduce their reproductive capability. As described above, the proposed action may increase the risks of adverse effects on the survival, growth, and reproductive development of white abalone within the Bay, although the extent of exposure for the white abalone population is uncertain based on the limited knowledge of white abalone distribution in the Bay. The proposed action may affect individuals at Palos Verdes, although the extent of exposure and potential effects at the individual level at that location is uncertain. The presence of white abalone off Palos Verdes indicates that at least some abalone are able to survive some exposure within the Bay, and the levels of pollutants such as metals outside the ZID that have been measured are below concentrations that are expected to cause significant health affects for abalone. However, evidence shows that abalone may experience significant reductions in growth and/or reproductive development when exposed to other potentially harmful contaminants that may occur in small concentrations throughout the Bay, which could impact individuals at Palos Verdes. While there have not been any known instances of abalone mortality or impairment associated with HABs in the Bay to date, individual white abalone throughout the Bay, including Palos Verdes, are at risk if a HAB occurs within the Bay. Implementation of additional studies and monitoring is needed to reduce our uncertainty regarding the effects and inform future analyses. The proposed action includes initiation of special studies that we expect will begin to monitor and describe the discharge of these persistent pollutants and the contribution of the discharge to HABs in the Bay. As this information is collected in the future, we expect to be better able to assess the potential effects of Hyperion's discharge on white abalone survival and reproduction in the Bay.

At this time, the scientific information needed to more fully evaluate the anticipated outcomes from exposure of white abalone to Hyperion's discharge at an individual and population level is needed. EPA has proposed to require monitoring and studies under the NPDES permit to address key questions regarding the impact of the Hyperion's discharge on the Bay and white abalone that may occur there will allow for improved evaluation of the impacts in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits. Based our current understanding of potential effects, as well as uncertainties in the possible magnitude and extent of those effects, and the measures that have been proposed to address these uncertainties and prospects for development of actions to minimize the extent of impacts in future consultations based on this information, we do not expect the potential effects of the proposed action to reduce the likelihood of survival and recovery of white abalone.

2.6.2.2 Black Abalone

As described in section 2.2 *Rangewide Status of the Species and Critical Habitat*, black abalone have declined throughout a large portion of the species' range (south of Cayucos), in areas that once supported the majority of the adult abundance in California. Although fishery harvest contributed to these declines, the primary cause was the disease called withering syndrome. Most disease-impacted populations remain at low abundance/density and may be more vulnerable to other factors affecting the species. As described in section 2.3 *Environmental Baseline*, black abalone historically occupied rocky reefs along the coasts of Palos Verdes and Point Dume. Little information is available regarding the presence, abundance, and distribution of black abalone in the Bay in recent years, but a recent survey did find black abalone at Palos Verdes, at sites just downcoast of the Bay. These data, along with the presence of good and moderate quality habitat in the Bay, indicate black abalone are likely present in the Bay. Though their abundance, distribution, and status within the Bay are not known, black abalone are likely at low densities, similar to populations elsewhere in southern California.

The proposed action may increase the risks of adverse effects on the survival, growth, and reproductive development of black abalone within the Bay. Black abalone are likely exposed to lower plume concentrations due to their location in nearshore waters, which may reduce their exposure to the potential effects of the discharge. The populations in the Bay are a small portion of the many known populations of black abalone throughout the coast of California, including confirmed populations of black abalone just upcoast (in Santa Barbara) and downcoast (in Palos Verdes) of the Bay. The species' survival and recovery will likely not require restoration of black abalone in all historically occupied reefs. Thus, any continued declines in populations in the Bay may not have a substantive effect on the survival and recovery of the species as a whole, though more information is needed regarding population connectivity to better evaluate the importance of these populations to the species. Although there is substantial uncertainty in how Hyperion's discharge may affect black abalone individuals, we do not expect exposure to the discharge effluent to significantly reduce survival and recovery of black abalone in the Bay, based on the levels of pollutants that have been measured in the effluent and the lower effluent concentrations that black abalone are likely exposed to, given their location in nearshore waters. In addition, there have not been any known instances of abalone mortality or impairment associated with HABs in the Bay to date.

At this time, the scientific information needed to more fully evaluate the anticipated outcomes from exposure of black abalone to Hyperion's discharge at an individual and population level is needed. EPA has proposed to require monitoring and studies under the NPDES permit to address key questions regarding the impact of the Hyperion's discharge on the Bay and black abalone that may occur there will allow for improved evaluation of the impacts in future consultations on the proposed action, which is expected to continue into the foreseeable future beyond the current permit cycle. When that information becomes available, we anticipate that EPA and NMFS will be in a better position to assess potential measures to minimize effects under future NPDES permits. Based our current understanding of potential effects, as well as uncertainties in the possible magnitude and extent of those effects, and the measures that have been proposed to address these uncertainties and prospects for development of actions to minimize the extent of impacts in future consultations based on this information, we do not expect the potential effects of the proposed action to reduce the likelihood of survival and recovery of black abalone.

2.7 Conclusion

After reviewing and analyzing the current status of the listed species that may be affected by the proposed action, the environmental baseline within the action area, the effects of the proposed action, and cumulative effects, it is NMFS' biological opinion that the proposed action is not likely to jeopardize the continued existence of the following ESA-listed species: blue whales, fin whales, Mexico DPS and Central America DPS humpback whales, WNP gray whale population, Guadalupe fur seals, East Pacific DPS green sea turtles, leatherback sea turtles, North Pacific Ocean DPS loggerhead sea turtles, olive ridley sea turtles, white abalone, and black abalone.

2.8 Incidental Take Statement

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

2.8.1 Amount or Extent of Take

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

We anticipate that all individual ESA-listed marine mammals, sea turtles, and abalone residing or feeding in the action area would uptake and/or accumulate potentially harmful contaminants such as PBDEs, thus increasing their body burden of these contaminants and the risk of incurring adverse effects to their growth, reproduction, and overall health and survival over a shorter period of time than would otherwise occur absent the action. We expect all ESA-listed individuals that may enter or reside in the Bay will be at increased risks of increased body burdens associated with the proposed action, although we expect that adverse effects will generally be limited to relatively few individuals (adults or juveniles) from these populations.

Due to uncertainty in the number of individuals that may be subject to exposure, and the response and level of harm that will occur for individuals exposed from each ESA-listed species, 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 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 Bay by Hyperion. While there are many potentially harmful contaminants, much of our analysis focused on the threat associated with accumulation of PBDEs, given the prominent literature describing the potential harm PDBEs can have on numerous ESA-listed species, and its known association with wastewater discharge in general. Consequently, we elect to use the extent of PBDE 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 mass loading of PBDEs that we expect to be discharged. As we described in section 2.4 *Effects Analysis*, the levels of PBDEs that are discharged by Hyperion have not been well documented historically. Using available information from other WWTPs, we estimated that Hyperion could be discharging up to approximately 62 pounds (28 kg) per year of total PBDEs into Santa Monica Bay, which are released into the ecosystem and are potentially bioavailable for uptake into the food web and ESA listed species. For the total 5 years of this proposed action, the incidental take, therefore, equates to the discharge of up to approximately 310 pounds (140 kg) of PBDEs for the permit cycle.

The proposed action includes development of special studies to evaluate the levels of CECs, including specifically PBDEs, in the effluent and mass loadings to the receiving water. Through this special study, we expect EPA to be able to monitor the discharge of PBDEs relative to the amount of PBDE discharge that has been assumed and described above, through the requirements placed upon the permittee (City of Los Angeles).

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

At this time, we cannot predict the precise extent that Hyperion's effluent discharge contributes to increased probabilities of HABs, or distinguish which HABs may be more or less associated or influenced by the additional nutrient input created by Hyperion's discharge. Consequently, we cannot further enumerate the anticipated take of ESA-listed species from the proposed action. Instead, we can describe the extent of take associated with increased probabilities of harmful effects from exposure to HABs by relating the extent of increased probability of HABs to the amount of nutrients, specifically nitrogen, that are being released into Santa Monica Bay. Consequently, we elect to use the extent of total nitrogen discharged as a surrogate to describe the extent of take associated with risks of increased probability of HAB exposure for ESA-listed species as a result of the proposed action.

We have, therefore, quantified the potential incidental take of the proposed action in terms of total nitrogen that we expect to be discharged. As we described in section 2.4 *Effects Analysis*, Howard et al. estimated that Hyperion's nutrient loading of Santa Monica Bay increased total N in the Bay by about 9,900 kg of N per km^2 per year. Given the area of Santa Monica Bay, equates to about 15.6 million kg of N over the course of a year being released in to the Bay as a result of Hyperion's discharge.

As part of the proposed action, EPA requires Hyperion to monitor the effluent discharge for parameters that includes the several forms of nitrogen (e.g., ammonia). That data can be used to develop estimates of nitrogen loading resulting from Hyperion's discharge consistent with what has been done previously by Howard et al. (2014). As part of the special study required by EPA that includes analysis of the mass balance of nitrogen species being treated and discharged by Hyperion, we expect EPA to be able to monitor and estimate the total level of nitrogen loading of the Bay relative to the amount of nitrogen discharge that has been assumed and described above, through the requirements placed upon the permittee (City of Los Angeles).

2.8.2 Effect of the Take

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

2.8.3 Reasonable and Prudent Measures

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

1. EPA shall monitor, document, and report the extent of incidental take of ESA-listed species resulting from Hyperion's discharge consistent with the surrogates described in Section 2.8.1 biological opinion, through the requirements placed upon the permittee (City of Los Angeles).

2.8.4 Terms and Conditions

The terms and conditions described below are non-discretionary, and the EPA or any applicant must comply with them in order to implement the RPMs (50 CFR 402.14). The EPA or any applicant has a continuing duty to monitor the impacts of incidental take and must report the progress of the action and its impact on the species as specified in this ITS (50 CFR 402.14). If the entity to whom a term and condition is directed does not comply with the following terms and conditions, protective coverage for the proposed action would likely lapse.

1. The following terms and conditions implement reasonable and prudent measure 1:

   1a. EPA shall require that the City of Los Angeles implement the CEC Monitoring Special Study required by the permit in a manner using sampling and analysis protocols that are consistent and/or at least equivalent with studies that have been used to measure PDBE levels in effluent and loading of receiving waters by other wastewater dischargers referred to in this opinion.

   1b. EPA shall require that the City of Los Angeles collects the necessary data to support the ongoing monitoring of all nitrogen forms from Hyperion's discharge, and the estimation of total nitrogen discharge on an annual basis to the waters of Santa Monica Bay. In order to support this, EPA shall require Hyperion to increase the frequency of the nitrate and organic nitrogen sampling (from quarterly in Table E-7) to match the ammonia sampling (monthly in

Table E-7), and to add grab sampling testing for both constituents (nitrate and organic nitrogen), in order to produce a more consistent and robust dataset. This monitoring frequency increase may be achieved through development and implementation of the special study work plan required by the NPDES permit. NMFS expects this dataset will be valuable in efforts such as the Bight 2018 studies by the discharger and other organizations.

1c. As part of the assessment of operational alternatives required by the special study evaluating the projected effects of water conservation and planned recycling on effluent acute toxicity and ammonia, EPA shall require the City of Los Angeles to evaluate the potential for denitrification at the Hyperion Wastewater Treatment Plant, in addition to any initiatives that are contingent upon compliance with acute toxicity and ammonia water quality objectives.

2.9 Conservation Recommendations

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

Harmful Algal Blooms

The following conservation recommendations related to HABs in the action areas would provide information for future consultations and address questions related to the effects of Hyperion's discharge on the frequency and extent of harmful algal blooms in the Bay.

1. EPA should support additional data collection within Santa Monica Bay and the Southern California Bight to help understand the discharge's potential influence on harmful algal bloom dynamics from discharges at the 5 and 1-mile outfalls. This could include:

    a. Generation of nitrogen form, timing, and mass balance data from upwelling and stormwater runoff events in Santa Monica Bay and the Southern California Bight to couple with the required data generation of nitrogen data from Hyperion's discharge.

    b. Assess what HAB species are in Santa Monica Bay, and whether they are manifesting concurrently with P. spp. and high domoic acid levels or if P. spp. tends to bloom first and therefore reduce the prevalence of other HAB species.

    c. Incident-specific monitoring of phytoplankton communities in the Bay before, during, and after planned discharges from the 1-mile outfall, to evaluate the presence, composition, and extent of blooms related to the discharge in the nearshore area.

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

Abalone

The following conservation recommendations for white abalone, black abalone, and black abalone critical habitat would provide information for future consultations involving the continued discharge of treated wastewater from Hyperion into Santa Monica Bay:

1. EPA should conduct a literature review of wastewater effects on marine invertebrates, focusing on abalone and other molluscs closely related to abalone. This literature review should provide information on contaminants, including toxic pollutants and persistent organic pollutants, which have been found to cause adverse effects on marine invertebrate species. The concentrations at which adverse effects result should then be compared to the concentrations measured in Hyperion's effluent to evaluate potential effects on abalone species in the Bay and identify pollutants of concern, to inform future studies as needed.

2. EPA should support monitoring of all abalone species in the Bay to evaluate their presence, abundance, and distribution, as well as to monitor effects of Hyperion's discharge on other abalone species. Information on the effects of Hyperion's discharge on other abalone species would inform our understanding of the potential effects on ESA-listed white abalone and black abalone. Field observations of effects would likely be more feasible than for the ESA-listed abalone species, because the non-ESA-listed species are likely to be more abundant.

3. The EPA should support and/or continue to support efforts to monitor and restore kelp beds in the Bay. Kelp is an important food resource for black abalone and white abalone and one of the essential features of black abalone critical habitat.

2.10 Reinitiation of Consultation

This concludes formal consultation for re-issuance of a permit to the City of Los Angeles for wastewater discharge by the Hyperion Treatment Plant under NPDES.

As 50 CFR 402.16 states, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained or is authorized by law and if: (1) The amount or extent of incidental taking specified in the ITS is exceeded, (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion, (3) the agency action is subsequently modified in a manner that causes an effect on the listed species or critical habitat that was not considered in this opinion, or (4) a new species is listed or critical habitat designated that may be affected by the action.

In this biological opinion, we have described the extent of take of the proposed action as being related to the amount of these potentially harmful contaminants being discharged into the Bay by Hyperion, and specifically to PBDEs. We estimated that approximately 62 pounds (28 kg) of total PBDEs could be discharged into Santa Monica Bay each year. If it is determined through special studies required by EPA or other means that the amount of PBDEs being discharged each year, then we would determine that the extent of take of the proposed action that has been anticipated in this biological opinion has been exceeded. We have also described the extent of take of the proposed action as being related to the amount of nutrients discharged into the Bay by Hyperion, specifically nitrogen. We have anticipated that about 9,900 kg of N per km^2, or about 15.6 million kg of N, is discharged over the course of a year into Santa Monica Bay. If it is determined through special studies required by EPA or other means that the amount of nitrogen being discharged each year, then we would 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 described in this biological opinion, we have described numerous uncertainties regarding the exposure and effects of ESA-listed species to the proposed action. If an event(s) transpire such that HABs in the Bay are identified as causing significant harm and/or mortality to ESA-listed species, we may determine that the extent of take associated with Hyperion's potential contribution to HABs and resulting effects to ESA-listed species has been exceeded, pending available information about the HAB event(s). In addition, as the state of science develops around contaminants, HABs, wastewater discharge, and ESA-listed species, along with any 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 we may determine that the extent of take of the proposed action that has been anticipated in this biological opinion has been exceeded.

2.11 "Not Likely to Adversely Affect" Determinations

The following ESA-listed species are not expected to be adversely affected by the proposed action, for the reasons explained below.

2.11.1 Southern California steelhead

Status and Occurrence in the Action Area

The Southern California Steelhead Distinct Population Segment (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. Adult migration to freshwater depends on physical factors such as the magnitude and duration of instream flows and sand-bar breaching. Adults may migrate several miles, hundreds of miles in some watersheds, to reach their spawning grounds. Once they reach their spawning grounds, females will use their caudal fin to excavate a nest in streambed gravels where they deposit their eggs. Males will then fertilize the eggs and, afterwards, the females cover the nest with a layer of gravel, where the embryos then incubate within the gravel. After emerging from the gravel, 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.

Although we found no information specifically regarding steelhead presence in the Bay, we expect the presence of individual adult and juvenile steelhead migrating to/from known steelhead watersheds (Malibu Creek and Topanga Creek) tributary to Santa Monica Bay. Based on our understanding, their presence in the Bay is expected to be intermittent and short in duration (hours to a few days), occurring when they migrate to or from the freshwater environment. Juvenile steelhead rapidly migrate off-shore upon entering the ocean, swimming hundreds of miles from their natal river or stream (Light et al. 1989; Daly et al. 2014). Migration rates of juvenile steelhead in the marine environment, assuming constant movement in a straight line, have been reported to range from about 1 mile per day to as high as 26 miles per day (Daly et al. 2014). Two juvenile steelhead that migrated from a southern California river and detected (acoustic tag) offshore of northern California were estimated to migrate at a rate of 20 and 12 miles per day (Kelly 2012).

Additionally, adult and juvenile steelhead migrating through the action area in coastal waters are expected to primarily occupy the upper water column. Tagging studies on the vertical distribution of adult steelhead have shown that adult steelhead spend on average approximately 95 percent of the time within 20 feet of the ocean surface, and 72 percent 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).

To estimate the number of steelhead likely to occur in the action area, we examined adult and juvenile steelhead migrant data collected from tributary watersheds to the Bay that are known to support steelhead, Malibu Creek and Topanga Creek (Table 2). These two waterways are the principal sources of steelhead to the action area. These data indicate few steelhead are likely to occur in the action area over the duration of the proposed action (i.e., 5-year permit).

Table 2. Summary of migrating steelhead observed in Malibu Creek and Topanga Creek for years 2001 through 2011 (Dagit and Krug 2011). N/A indicates no data reported.

Year	Malibu Creek		Topanga Creek	
	Adult	**Smolt**	**Adult**	**Smolt**
2001	N/A	N/A	1	0
2002	N/A	N/A	2	0
2003	N/A	N/A	1	14
2004	0	N/A	0	0
2005	0	N/A	0	0
2006	0	N/A	1	9
2007	4	N/A	2	0
2008	2	N/A	1	1
2009	1	N/A	0	1
2010	3	N/A	1	28

2011	N/A	N/A	2	1

Effects of the Proposed Action

The effects of the proposed action on steelhead are expected to be discountable and insignificant. As explained more fully below, the available information indicates that steelhead are not expected to encounter the principal areas of discharge (i.e., 1-mile and 5-mile discharge points, and ZID, and should steelhead encounter the wastewater effluent plume outside of the ZID, the limited concentration and duration of exposure is not expected to result in lethal or sub-lethal effects. We used ammonia for assessing the effects of wastewater discharge because ammonia is known to be toxic to steelhead and provides the most extensive available data on wastewater effluent in the action area.

As a matter of background, the one-mile discharge outfall is used for preventative maintenance activities (up to 4 times per year) and when stormwater overwhelms capacity of the facility. As a result, this discharge point is used infrequently and for short-duration. For this reason, and because this discharge point is more than 8.5 miles from the expected steelhead migration pathway, we do not expect juvenile or adult steelhead to be exposed to wastewater effluent from this discharge point.

To assess the likelihood of adult or juvenile steelhead encountering the most concentrated portion of the wastewater-effluent plume (i.e., ZID) from the 5-mile discharge point, we delineated a migratory pathway presumed to occur within a 45-degree approach angle to and from the confluence of Malibu Creek and Topanga Creek (Figure 4). The closest point of the ZID to the migration pathway is about 4.5 miles, rendering the likelihood of steelhead encountering high effluent concentration extremely unlikely. Furthermore, the ZID is 65 feet or more below the water surface, well below the depth that juvenile and adult steelhead are reported to occupy.

Should steelhead encounter the wastewater effluent plume outside of the ZID, the limited concentration and duration of exposure is not expected to result in lethal or sub-lethal effects. In this analysis, we use potential exposure to ammonia for assessing the potential effects of wastewater discharge because ammonia is known to be toxic to steelhead and provides the most extensive available data on wastewater effluent in the action area. Regarding steelhead exposure to diluted concentration of wastewater effluent, the proposed action allows for an instantaneous ammonia concentration of 6 mg N per liter outside of the ZID, about half the LC_{50} (96-hour exposure) for rainbow trout (EPA 1999), which could potentially cause lethal or sub-lethal effects on steelhead depending on the duration of exposure. However, steelhead are not likely to encounter such concentrations because of additional dilution occurring between the ZID and the location of steelhead in the very upper part of the water column. Also, the potential exposure duration would most likely be on the order of minutes rather than days based on steelhead swimming speed and distribution of the effluent plume, further reducing the likelihood of lethal or sub-lethal effects on steelhead. Monitoring data collected quarterly from February 2011 to August 2016 under similar or same wastewater discharge as the proposed action indicate the highest ammonia concentration within the water column where steelhead may occur was 0.23 mg N per liter, with more than 90 percent of the samples detecting no ammonia (i.e., <0.02 mg N per liter). It is noteworthy that this detection was isolated to the upper water column at a single monitoring site (1 of 21 monitoring locations distributed throughout the action area) and no detectable concentration of ammonia observed that day at any of the other monitoring sites. Aside from the 0.23 mg N per liter detection, ammonia sampling resulted in values ≤0.06 mg N per liter, well below the concentration considered by researchers to be protective of steelhead (i.e., ≤0.67 mg N per liter) (Eddy 2005). Because concentrations of ammonia (and wastewater effluent in general) that steelhead may be exposed to during the course of the proposed action are expected to be similar to those detected during the quarterly monitoring of 2011 through 2016, neither lethal nor sub-lethal effects associated with exposure to potentially toxic constitutes in the plume are expected for steelhead to occur as a result of the proposed action.

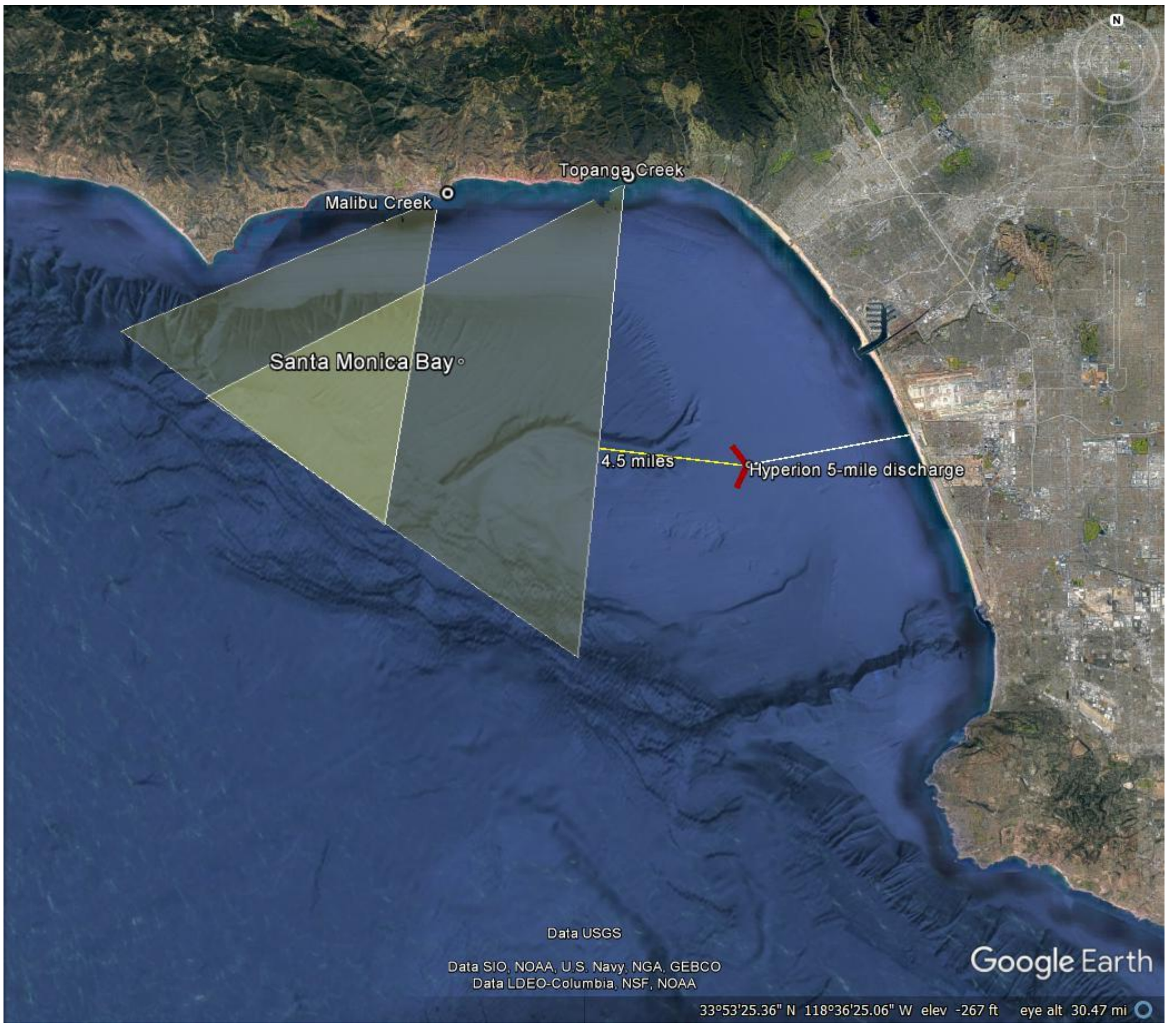

Figure 4. The shaded areas represent potential migration pathways using a 45-degree angle off-shore trajectory from steelhead-occupied watersheds. The red "Y" is the zone of initial dilution for wastewater treatment effluent has a volume of about 1,607 acre-feet (0.004% of the action area).

Wastewater discharges of persistent bio-accumulative constituents (e.g., DDT, PCBs and PBDEs) can potentially cause lethal or sub-lethal effects on steelhead, by inhibiting growth or increasing disease susceptibility. These effluent constituents are likely concentrated in sea-floor sediments in the action area near the wastewater discharge point, well below the depth that juvenile and adult steelhead are reported to occupy. Although these constituents may be introduced to the upper water column via the food web and consumed by steelhead, the brief exposure (hours to a few days) is not expected to facilitate accumulation of these constituents sufficient to cause lethal or sub-lethal effects on adult or juvenile steelhead. For instance, in a study of disease susceptibility owing to exposure to PBDEs (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 (i.e., control (no added PBDEs), environmentally relevant concentration, and 10 times environmentally relevant concentration) did not exceed 2 percent in any treatment group, and there were no significant differences in cumulative mortality among the treatment groups. Although not statistically significant, a slight downward trend in weights and condition factors was observed for fish fed PBDE diets relative to fish fed the control diet. Fourteen days after introducing the three treatment groups to a marine bacterial pathogen, cumulative mortality was 7 percent higher in the environmentally relevant concentration group relative to the control group (i.e., cumulative mortality of 33% and 26%, respectively). Interestingly, the treatment group fed the highest concentration of PBDEs had the lowest cumulative mortality (see section 2.4.2.2 *Adverse Health Effects from Exposure to Potentially Harmful Contaminants* for discussion of hermetic dose-responses). 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.

HABs may result from increased nutrient discharges owing to the proposed action causing potential indirect effects to steelhead. For instance, the neurotoxin domoic acid produced from *Pseudo-nitzschia* blooms can cause sub-lethal effects to death of seabirds and marine mammals that consume fish or marine invertebrates containing the toxin. However, research indicates that fish, including steelhead, are tolerant to domoic acid exposure, showing no neurological symptoms following oral exposure (circle, upside-down, and spiral swimming; Lefebvre et al. 2007). Another consequence that may result from HABs is reduced or depleted dissolve oxygen in the water column, causing lethal or sub-lethal effects to fish (e.g., hypoxia). However, the occurrence of reduced or depleted dissolved oxygen in the action area appears to occur in confined basins (e.g., reported fish kills in King Harbor) 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). Additionally, plume monitoring (Figure 1) indicates that the Hyperion discharge is confined to the southern portion of the Bay the vast majority of the time due to localized currents. Therefore, the nutrient contributions of the discharge and any related HAB activity occur away from areas that steelhead utilize migrating into and out of the Bay. Because juvenile and adult steelhead are reported to occupy the near-surface water column, exposure to reduced or depleted concentrations of dissolved oxygen is not expected to occur.

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

2.11.2 Green sturgeon; Southern DPS

Status and Occurrence in the Action Area

The green sturgeon is an anadromous and bottom-oriented (demersal) fish species in the family Acipenseridae. Green sturgeon are long-lived, with a maximum age of adults likely ranging from 60 to 70 years. They are also large fish, capable of exceeding 6.5 feet (2m) in length and 198 pounds (90kg) in weight. Subadult and adult green sturgeon have a marine and coastal range that extends from the Bering Sea, Alaska (Colway and Stevenson 2007) to El Socorro, Baja California, Mexico (Rosales-Casian and Almeda-Juaregui 2009). Adults spawn in the mainstem of large rivers during spring and summer months. After rearing in freshwater or the estuary of their natal river as juveniles, green sturgeon transition to the subadult stage and move from estuarine to coastal marine waters. They are one of the most marine-oriented and widely distributed of the sturgeon, spending much of their lives migrating between estuaries along the coast. Relatively little is known about how green sturgeon use coastal marine habitats. While in the ocean, they generally occur between 0 and 110 m, spending most of their time in areas between 20 to 80 m in depth (Erickson and Hightower 2007; Huff et al. 2011). While in marine waters, they may be feeding or simply migrating between estuaries.

Based on genetic analyses and spawning site fidelity (Adams et al. 2002; Israel et al. 2004), NMFS determined that the green sturgeon includes at least two DPSs: a Northern DPS consisting of populations originating from coastal watersheds northward of and including the Eel River (Northern DPS); and a Southern DPS consisting of populations originating from coastal watersheds south of the Eel River (Southern DPS). In 2006, NMFS listed the Southern DPS as threatened under the ESA, but determined that ESA listing was not warranted for the Northern DPS. The main threats to the Southern DPS are the loss of access to historical spawning habitat in the upper Sacramento and upper Feather Rivers due to impassable barriers (USFWS 1995; Mora et al. 2009); restriction of spawning to a portion of the mainstem Sacramento and lower Feather rivers; impaired spawning and rearing habitats in fresh and estuarine waters in the Central Valley, California; and historical and ongoing bycatch in fisheries.

Outside of their natal rivers and estuaries, the Northern DPS and Southern DPS co-occur throughout much of their range. Tagging studies have confirmed the presence of Southern DPS fish from as far north as Graves Harbor, Alaska, to as far south as Monterey Bay, California (Lindley et al. 2008; NMFS 2009; Lindley et al. 2011; Huff et al. 2012). While these studies recognize that telemetry studies to date have not focused on areas south of Monterey Bay, fisheries data further indicate that green sturgeon are rare in the region (Lee et al. 2017; 74 FR 52300). In NOAA observer records for the California halibut fishery operated along the California coast from 2002-2015, observed green sturgeon encounters occurred exclusively off San Francisco Bay (Lee et al. 2017). Fishery interaction records highlighted in the Southern DPS critical habitat designation (74 FR 52300) 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). We do not know of any records of green sturgeon in Santa Monica Bay, although one green sturgeon was observed in the California halibut set net fishery off San Pedro/Rancho Palos

Verdes in 1993 (unpublished data from Rand Rasmussen, 18 July 2006). These records indicate that green sturgeon could occur in the action area, but the reports are infrequent and speak to the rarity of the species in the geographic region. None of the green sturgeon observed south of Monterey Bay have been identified

Effects of the Proposed Action

Subadult and adult green sturgeon have a marine and coastal range that extends from the Bering Sea, Alaska (Colway and Stevenson 2007) to El Socorro, Baja California, Mexico (Rosales-Casian and Almeda-Juaregui 2009). Telemetry and modeling studies suggest that Southern DPS green sturgeon primarily occur from Graves Harbor, Alaska to Monterey Bay, California (Lindley et al. 2008, 2011, Huff et al. 2012). While these studies recognize that telemetry studies to date have not focused on areas south of Monterey Bay, fisheries data further indicate that green sturgeon are rare in the region (Lee et al. 2017; 74 FR 52300). In NOAA observer records for the California halibut fishery operated along the California coast from 2002-2015, observed green sturgeon encounters occurred exclusively off San Francisco Bay (Lee et al. 2017). Fishery interaction records highlighted in the Southern DPS critical habitat designation (74 FR 52300) 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). These records indicate that green sturgeon could occur in the action area, but the reports are infrequent and speak to the rarity of the species in the geographic region.

Based on the studies cited above, Southern DPS green sturgeon are likely extremely rare in Southern California, with a very low probability of occurrence in the Bay and exposure to Hyperion's discharge effluent. Furthermore, 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 low given the rarity of observations this far south in recent decades. As a result, we conclude that the effects of the proposed action on Southern DPS green sturgeon are discountable, because the likelihood that green sturgeon occur in the action area is extremely low based on the species distribution and habitat use. Therefore, the proposed action is not likely to adversely affect Southern DPS green sturgeon.

2.11.3 Scalloped hammerhead shark; Eastern Pacific DPS

Status and Occurrence in the Action Area

The scalloped hammerhead shark can be found in coastal warm temperate and tropical seas worldwide. The scalloped hammerhead shark occurs over continental and insular shelves, as well as adjacent deep waters, but is seldom found in waters cooler than 22° C (Compagno 1984). It ranges from the intertidal and surface to depths of up to 450–512 m (Klimley 1993), with occasional dives to even deeper waters (Jorgensen et al. 2009). It has also been documented entering enclosed bays and estuaries (Compagno 1984). Distribution in the eastern Pacific extends from the coast of Southern California, including the Gulf of California, to Ecuador and possibly Peru, to the offshore waters around Hawaii and Tahiti (Miller et al. 2014). The 2014 Status Review Report (Miller et al. 2014) identified 6 DPS 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 (NMFS 2014b). 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 (NMFS 2014b). Abundance data from the eastern Pacific are limited, but available information suggests that the Eastern Pacific DPS is declining (NMFS 2014b). Although precise population estimates are not available in the eastern Pacific, estimates based on assumptions related to genetic and demographic parameters have been made for populations in Baja and Pacific Panama, which suggest combined totals in these two populations is at least in the 10's of millions (Duncan et al. 2006; Miller et al 2014).

Effects of the Proposed Action

Even though the Bay 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 been rarely documented. To date no scalloped hammerheads have been documented as captured in fisheries along the U.S. west coast (NMFS 2015). Although the presence of scalloped hammerheads in the action area is possible, the possibility of such an occurrence during the course of the proposed action is extremely unlikely given that scalloped hammerheads sharks favor warmer waters more often located in lower latitudes. Consequently the risks of exposure to the proposed action are very low. As a result, we conclude the risks of adverse effects from the proposed action for the Eastern Pacific DPS of scalloped hammerhead sharks are discountable.

2.11.4 Black Abalone Critical Habitat

Status and Occurrence in the Action Area

NMFS designated critical habitat for black abalone in 2011 (76 FR 66806). The designation encompasses rocky intertidal and subtidal habitat (from the mean higher high water, MHHW, line to a depth of -6m relative to the mean lower low water, MLLW, line) within five segments of the California coast between Del Mar Landing Ecological Reserve to the Palos Verdes Peninsula, as well as on the offshore islands. Essential habitat features include rocky substrate (e.g., rocky benches formed from consolidated rock or large boulders that provide complex crevice habitat); food resources (e.g., macroalgae); juvenile settlement habitat (rocky substrates with crustose coralline algae and crevices or cryptic biogenic structures); suitable water quality (e.g., temperature, salinity, pH) for normal survival, settlement, growth, and behavior; and suitable nearshore circulation patterns to support successful fertilization and larval settlement within appropriate habitat. Threats to black abalone critical habitat include coastal development or in-water construction projects (e.g., coastal armoring, pier construction or repair); activities that can increase sedimentation (e.g., sand replenishment, beach nourishment, side-casting); oil or chemical spills and response activities; and vessel grounding and response activities. Operations that involve withdrawing water from and/or discharging water to marine coastal waters may also affect black abalone critical habitat by increasing local water temperatures (e.g., discharge of heated effluent), introducing elevated levels of metals or other contaminants into the water, or altering nearshore circulation patterns.

The rocky intertidal and shallow subtidal habitats surrounding the Palos Verdes Peninsula (from the Palos Verdes/Torrance border to Los Angeles Harbor in southwestern Los Angeles County) are designated as critical habitat for black abalone and overlap with the action area in Santa Monica Bay. Past long-term monitoring data (primarily at sites downcoast of the Bay) indicate that Palos Verdes supported dense black abalone populations. Populations have declined severely due to disease, but critical habitat remains in fair to excellent condition. In particular, the area continues to provide good to high quality rocky substrate and food resources and fair to good settlement habitat for black abalone (NMFS 2011).

Studies indicate past impacts of wastewater discharges on black abalone critical habitat at Palos Verdes. Wastewater discharges likely contributed to the decline of kelp beds along the Palos Verdes Peninsula in the 1940s through 1960s, by increasing siltation, reducing light levels, and reducing water quality (Wilson et al. 1980). Leighton and Boolootian (1963) noted that in the 1950s, black abalone populations at Flat Rock (a site at Palos Verdes within the Bay) appeared to be starving due to a lack of vegetation, which they attributed to contamination of the water due to wastewater discharges. In the mid-1970s, kelp beds along the Palos Verdes Peninsula began to recover, due to a variety of factors including improvements in wastewater treatment and kelp restoration efforts (Wilson et al. 1979). Although they have not yet recovered to their extent in the early 1900s, the most recent assessment rated the condition of the kelp canopy in the Bay as fair (Pondella 2015). Restoration efforts are ongoing and are expected to continue to improve the condition of kelp beds in the Bay.

Effects of the Proposed Action

The rocky intertidal and shallow subtidal habitats along the Palos Verdes Peninsula have been designated as black abalone critical habitat, including a portion of the Palos Verdes coast within the Bay. As described above, the discharge plume is expected to extend throughout the Bay, with greater plume probabilities downcoast toward Palos Verdes and lower plume probabilities upcoast toward Malibu. Based on these modeled plume probabilities, black abalone critical habitat along the Palos Verdes coast would be exposed to the discharge plume.

Exposure to relatively high concentrations of constituents from the effluent discharge plume is less likely in the nearshore environment where black abalone critical habitat has been designated given circulation patterns and the depth and distance of the outfall from the shore. If there is some exposure to the effluent plume, those constituents are not likely to affect the physical properties of the rocky substrate. However, given the effect of Hyperion's effluent discharge on water quality in the Bay, described in detail throughout section 2.4 *Effects Analysis* above, the plume has the potential to affect the growth of algae that serve as settlement habitat and food resources for black abalone. Crustose coralline algae is an important component of juvenile settlement habitat for black abalone. A few studies have examined how crustose coralline algae may be affected by exposure to wastewater effluent and show that the magnitude of effects depends on the distance from the outfall and the concentration of wastewater to which the reefs are exposed. May (1985) observed no significant change in the abundance of red encrusting algae due to exposure to secondary and tertiary treated wastewater in Australia, whereas Bjork et al. (1995) observed decreased cover of crustose coralline algae at sites closer to sewage outfalls in Zanzibar. Roberts et al. (1998) observed significantly reduced crustose coralline algal cover at subtidal rocky reefs within months after exposure to secondary treated wastewater off Australia. In that study, the reef was within 100 m of the wastewater outfall. Black abalone critical habitat is located about 10 miles from Hyperion's outfalls, in shallow nearshore waters where plume concentrations are expected to be relatively low (Schaffner et al. 2011), reducing the likelihood that the discharge will have significant effects on the growth of crustose coralline algae. Therefore, we expect the risks of effects from Hyperion's discharge on juvenile black abalone settlement habitat to be relatively low.

Historical wastewater discharges (prior to full secondary treatment) contributed to declines in water quality and kelp growth along the Palos Verdes Peninsula in the 1940s to 1960s, likely reducing this important food source for black abalone (Leighton 1959**;** Cox 1962**;** Leighton and Boolootian 1963**;** Wilson et al. 1979**;** Miller and Lawrenz-Miller 1993), potentially causing reduced growth and reproductive development in black abalone (Leighton and Boolootian 1963). Lab studies support this idea, showing that exposure of giant kelp (*Macrocystis pyrifera*) to low concentrations (1% or greater) of primary treated effluent significantly inhibited zoospore germination (Anderson and Hunt 1988). Improvements in wastewater treatment, particularly implementation of full-secondary treatment, have reduced the adverse effects of wastewater discharge on kelp growth and allowed recovery of kelp beds along the Palos Verdes Peninsula (Wilson et al. 1979). Chronic toxicity testing using Hyperion's discharge effluent have found no observable effects on giant kelp sporophytes when exposed to concentrations as high as 10% effluent (compared to concentrations of 1.19% effluent expected in the plume; City of LA 2015). Based on these results, we would not expect exposure to the discharge plume to reduce the growth of giant kelp and the availability of this food resource in black abalone critical habitat. Overall, the discharge likely continues to affect critical habitat, but at an insignificant level that is not likely to affect the quality of the habitat and its conservation value for black abalone. As a result, we conclude that the proposed action is not likely to adversely affect black abalone critical habitat.

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

Section 305(b) of the MSA directs Federal agencies to consult with NMFS on all actions or proposed actions that may adversely affect EFH. The MSA (section 3) defines EFH as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity." 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) also requires NMFS to recommend measures that can be taken by the action agency to conserve EFH.

This analysis is based, in part, on the EFH assessment provided by the EPA and descriptions of EFH for Pacific Coast groundfish (Pacific Fishery Management Council [PFMC] 2005), coastal pelagic species (CPS) (PFMC 1998), and highly migratory species (HMS) (PFMC (2007)**]** contained in the fishery management plans developed by the PFMC and approved by the Secretary of Commerce.

3.1 Essential Fish Habitat Affected by the Project

The proposed project occurs within EFH for various federally managed fish species within the Pacific Coast Groundfish, Coastal Pelagic Species, and Highly Migratory Species Fishery Management Plans (FMPs). In addition, the proposed project occurs within, or in the vicinity of, rocky reef and canopy kelp habitats, which are designated as habitat areas of particular concern (HAPC) for various federally managed fish species within the Pacific Coast Groundfish FMP. HAPC are described in the regulations as subsets of EFH which are rare, particularly susceptible to human-induced degradation, especially ecologically important, or located in an environmentally stressed area. Designated HAPC are not afforded any additional regulatory protection under MSA; however, federal projects with potential adverse impacts to HAPC will be more carefully scrutinized during the consultation process.

3.2 Adverse Effects on Essential Fish Habitat

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

As described above in section 1.3.2 *Effects Analysis*, the SMBRC identified 19 pollutants of concern for the bay. The EFH Assessment provided by EPA evaluated a subset of those 19 pollutants that are present, in quantifiable amounts, in the effluent. These pollutants include metals (cadmium, copper, nickel, lead, silver, zinc), nutrients (nitrogen, phosphorus) and ammonia, total suspended solids, biological oxygen demand, and oil and grease. CECs were also analyzed in the EFH Assessment. In the following sections, we evaluate the adverse effects to EFH from this subset of pollutants, including ongoing studies to better understand those effects, and efforts to address them, such as compliance with existing water quality standards. Much of the information used in this effects analysis was taken directly from the EFH Assessment provided by EPA.

Metals

Metals are known to bioaccumulate in marine organisms and can cause a variety of chronic health problems and physical anomalies at elevated concentrations. The concentrations of metals in the plant influent, except for copper and zinc, have declined significantly since the 1980s, largely due to source control programs. Copper, zinc, and silver were frequently detected in the effluent during the last 5 years, which is similar to results for the previous reporting period. Nickel, which is soluble in wastewater and thus has a lower removal efficiency, was also detected in the influent and effluent in 2013 and 2014. However, concentrations of all detected metals in the effluent, after applying the initial dilution factor as prescribed by the 2015 California Ocean Plan, are below water quality standards. Moreover, the 2017 permit includes performance goals for metals to ensure that treatment (i.e., removal efficiency) is maintained. Specifically, the City must investigate the cause if a performance goal is exceeded. If a performance goal is exceeded in three successive monitoring periods, the City of Los Angeles must submit a written report with corrective actions.

Toxicity and Ammonia

Various pollutants, including ammonia, pesticides, petroleum-based contaminants, and metals, can adversely affect EFH through acute (i.e., lethal) or chronic (i.e., sublethal) toxicity (Hanson et al. 2003). Initial dilution, the process which results in the rapid and irreversible turbulent mixing of wastewater with ocean water around the point of discharge, is rapid and energetic, with timescales of seconds to minutes. As a result, the EFH Assessment concludes that organisms temporarily entrained in or passing through the region surrounding the diffuser where initial dilution occurs, referred to as the ZID, are not present long enough to be exposed to chronic or lethal toxicity effects. As noted previously, the ZID under critical conditions for the 5-mile outfall was estimated to extend 65.6 feet on either side of the two diffuser legs, and 130 feet vertically up from the diffuser (EPA 2017).

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. The City has been using red abalone and topsmelt for acute and chronic toxicity tests, respectively. Because EPA has determined that no species or test method is always the most sensitive, the 2017 permit requires re-screening of the standard test species every two years to ensure the most sensitive test species is used in evaluating the toxicity of the effluent. Monthly monitoring is employed to ensure the chronic toxicity effluent limit is not exceeded. Effluent discharge did not exceed chronic effluent limits during 2011 to 2014 and no chronic toxicity exceeding the effluent limits were reported. However, limited and transient acute toxicity has been reported, most likely due to higher than normal ammonia concentrations.

Ammonia is one of several forms of nitrogen existing in aquatic environments and is toxic to aquatic life at certain concentrations. Environmental factors, such as pH and temperature, affect ammonia toxicity to aquatic organisms. Concentrations of ammonia in the effluent have been increasing over the last 9 years, largely due to increased urbanization of the service area and the use of a thermophilic digester process. Thermophilic digestion can produce Class A biosolids, which contain no detectible levels of pathogens, but also produce a higher concentration of ammonia compared to mesophilic digestion (Vindis et al. 2009). More recent increases in ammonia concentrations may also be due to water conservation and drought conditions. However, receiving water monitoring data show ammonia concentrations to be below those required by the California Ocean Plan water quality objectives. The 2017 permit includes an average monthly and daily maximum limit. It also requires monthly monitoring of the effluent (as well as the influent) to characterize effluent quality, detect noncompliance, and consider the need for data (used in reasonable potential analysis to establish effluent limits). In addition, as a result of informal consultation with NMFS, EPA and the Regional Board added a special study requirement to evaluate the projected effects of water conservation and planned recycling on effluent acute toxicity and ammonia. The special study will include a mass balance of nitrogen species through the treatment plant and an assessment of operational alternatives (e.g., treatment optimization, additional treatment, additional dilution credits) to address projected compliance with acute toxicity and ammonia water quality objectives.

Nutrients and HABs

As described above in section 2.3.1 of the *Environmental Baseline* and section 2.4.3 of the *Effects Analysis*, nutrient loading can cause increased plant and algal growth leading to eutrophication and increased instances of HABs. As noted previously, *P. spp*. are domoic acid producing diatoms, and are the most frequently noted HAB species in the action area. Domoic acid is a water soluble neurotoxin that accumulates in shellfish and planktivorous fish such as anchovy and sardine. Although the effects to piscivorous birds and marine mammals are well documented and wide spread (Schnetzer et al. 2013), impacts to other species are less certain. Impacts on schooling fish species are not believed to be as extensive, and laboratory work has shown that fish species ingesting domoic acid producing phytoplankton seem to be able to isolate and eventually depurate these compounds (Lefebvre et al. 2012, 2007). However, it is unknown if there is a metabolic cost to this process for the fish. Effects to zebrafish (*Danio rerio*) egg hatching and development were shown when domoic acid was microinjected into their eggs (Vasconcelos et al. 2010) but studies reporting effects to fish egg or larval development under realistic exposure scenarios were not readily found in a literature search. Vasconcelos et al. (2010) also reported that most studies showed little or no effects to survival or reproduction to several species of mollusks and crustaceans from exposure to domoic acid. However, Lui et al. (2007) found significantly compromised growth and survival of king scallop larvae at environmentally realistic exposures to domoic acid. Further research on the 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 (Backer and Miller 2016; Gosselin et al. 1989; Kudela et al. 2010; Lefebvre et al. 2004; Trainer et al. 2010). *A. cantenella* is the predominant PSP toxin producing species in the CA Current system and the State of WA has experienced numerous shellfish fishery closures due to the presence of saxitoxin in the environment (Trainer et al. 2010). Vasconcelos et al. (2010) reviews several studies that found effects of saxitoxin on crustacean larvae ranging from lethality to brine shrimp to sublethal effects on crab larvae. In addition to numerous study references indicating toxicity to winter flounder, red sea bream and Japanese anchovy, Gosselin et al. (1989) found heavy mortality to capelin and Atlantic herring larvae and juveniles exposed to environmentally realistic concentrations of *A. tamarense* complex species through both vectorial poisoning and direct intoxication. Lefebvre et al. (2004) conducted a dietary uptake experiment with zebrafish larvae and found that 24 hour exposures to saxitoxin induced paralysis, and high levels had morphological impacts. Reduced growth rates were also prevalent resulting in depressed cumulative survival compared to control fish (40% v. 80%). A follow-up study with dissolved saxitoxin by Lefebvre et al. (2005) was conducted on larval Pacific herring. At levels > 47 µg/L, the saxitoxin caused significant reductions in sensorimotor function within one hour. Interestingly, the effect was transient to a few hours indicating there may be significant variability in effects to different species. However, impacts to the early life stages of Pacific herring are still likely due to an inability to avoid predators or escape the hypoxic conditions caused by significant HABs. There is a paucity of data on extracellular saxitoxin concentrations (Lefebvre et al 2008) but the patchiness of HABs and the highly variable release of toxins when the cell lyse could produce high levels in the field.

Although *A. cantenella* is present along the entire outer open coast, incidents of saxitoxin effects to fish species or EFH specific to the SCB are not prevalent in the literature and monitoring by SCCOOS at the Santa Monica Pier infrequently detects *A. spp*. It may be that the spring time blooms of *P. spp*. typically appear earlier than *A. app.* complex, limiting its ability to develop into a full blown HAB in the SCB, or much of the monitoring effort may simply be focused on the marine mammals and birds impacted by domoic acid, and any saxitoxin toxicity to federally managed fisheries or their prey species is being missed. Both Lefebvre et al. (2004) and Gosselin et al. (1989) theorized that finfish stocks could be significantly impacted for species whose larval and early juvenile life stages coincided with saxitoxin generating HABs. More monitoring of algal species distribution and occurrence during HABs across the entirety of the Bay needs to occur to determine if multiple species and/or toxins routinely overlap in Santa Monica Bay and the SCB as a whole.

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, but it is unclear if this large family of marine toxins impacts fish and their habitat. However yessotoxin has been identified as the major causative agent in the largest invertebrate mass mortality event recorded in coastal Northern California, in Sonoma County in 2001, which impacted red abalone, sea urchins and crab species from Bodega Bay to Anchor Bay (De Wit et. al 2014). Yessotoxin impacts such as these represent an effect to EFH for rockfish species which consume crabs or other invertebrates.

One other group of dinoflagellates, Procentum species, has been recorded frequently and in large numbers at the Santa Monica Pier by SCCOOS. Similar to all dense HABs, its effect to EFH likely comes from impacts to dissolved oxygen levels at the scale of the algal bloom resulting in fish kills (Anderson et al. 2012; Backer and Miller 2016; Trainer et al. 2010) and presumably impacts to other species which cannot escape the HAB area. Algal masses are known to rapidly deplete available dissolved oxygen in the water column due to high respiration by the algae or increased respiration by bacteria during algal decay and this decrease can potentially lead to hypoxic levels for periods of time (Backer and Miller 2016; Booth et al. 2015, 2014). HAB biomass is believed to be contributing to the overall decline of dissolved oxygen levels in coastal waters (Booth et al. 2015, 2014; Capone et al. 2013; McLaughlin et al. 2017). The *P. spp.* are also known to flocculate and form masses large enough to sink to the ocean floor when they die, carrying domoic acid with them which may be ingested by benthic species spreading the toxin within the benthic food web (Schnetzer et al. 2013, 2007; Trainer et al. 2010). If levels are sufficient, this may also cause a depression in dissolved oxygen levels at the sea floor as the algae decays. Other studies of HABs have noted mechanical damage to fish gills and shading impacts from the massive blooms to other species of phytoplankton or even sea grass beds (Backer and Miller 2016; Anderson et al. 2012).

Information specific to the impacts on EFH by HABs in the Bay is lacking. Based upon available information from local monitoring at the Santa Monica Pier and scientific literature, impacts to CPS may be occurring when lifestages 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 Bay.

Dissolved Oxygen, Biological Oxygen Demand, and Total Suspended Solids

Aquatic organisms require sufficient levels of dissolved oxygen to breathe and grow. As a result, dissolved oxygen is an important measure of water quality and an indicator of a water body’s ability to support aquatic life. Biological oxygen demand and total suspended solids both affect the level of oxygen in a receiving water, either directly or indirectly. Oxygen is depleted more rapidly with higher biological oxygen demand. Although the impact mechanisms are more indirect with respect to total suspended solids (e.g., decreased photosynthesis resulting from reduced light; increased water temperature, which holds less oxygen; etc.), elevated total suspended solids can also reduce dissolved oxygen levels. The permit includes effluent limits for biological oxygen demand and total suspended solids, which have been met since full secondary treatment was implemented in 1998. Moreover, when compared to reference sites, out of range occurrences for dissolved oxygen, which were mainly attributable to entrainment (i.e., the plume pulling up colder deeper water), were still within the allowable variability per the California Ocean Plan objective.

Oil, Grease, and Trash

High density residential and commercial areas may contribute significant amounts of pollutants, including oil, grease, and trash to stormwater runoff. These pollutants may be introduced through littering or improper waste disposal and cleaning practices by restaurants. Simple and effective methods to address these pollutants are being implemented within the Santa Monica Bay Watershed, such as installation of catch basin screening and filtration devices, bird-proofing trashcan lids in parks, better placement of trash receptacles in high traffic areas, and public outreach. To reduce stormwater pollution generated by restaurant activities, the SMBRC and Bay

Foundation implement the Clean Bay Restaurant Certification Program (SMBRC 2013). To be certified under the program, restaurants must have a full-scale recycling program, properly divert storm water runoff, implement exterior dry sweeping, and follow additional city restrictions such as adhering to plastic bag bans or avoiding the use of Styrofoam. Hyperion's NPDES permit also includes effluent limits for pollutants with TMDLs and technology-based effluent limits, including oil and grease, settleable solids, turbidity, and temperature.

CECs

As described above in section 2.3.1.1 of the *Environmental Baseline*, the term CEC refers to several types of chemicals, including persistent organic pollutants, pharmaceuticals and personal care products, veterinary medicines, endocrine-disrupting chemicals, and nanomaterials. Because Hyperion's effluent is mostly domestic wastewater (~80%), many CECs such as pharmaceuticals and personal care products are expected to be found in higher concentrations in Hyperion's effluent as opposed to CECs associated with wasterwater discharge from industrial activities.

Wastewater effluent can be a major source of CECs, which can cause deleterious effects in aquatic life. For instance, PBDEs resist degradation in the environment and can bioaccumulate in adipose tissues. They are endocrine disruptors and neurotoxins that can negatively impact fish nervous systems, thyroid and hepatic (i.e., liver) functions, endocrine and reproductive systems (Siddiqi et al. 2003). Crane et al. (2006) found that pharmaceuticals have the potential to adversely affect aquatic organisms, and recommended additional testing to better understand their acute and chronic effects in the natural environment. A review of recent publications on environmental concentrations and aquatic toxicity of personal care products by Brausch and Rand (2011) noted that available information varied substantially depending upon the specific compound. According to their review, existing data indicate most personal care products are relatively non-toxic to aquatic organisms at anticipated environmental concentrations. However, many of these compounds are known endocrine disruptors that can have negative effects to fish, with some having a potential to cause estrogenic effects at relatively low concentrations (Brausch and Rand 2011). They also recommended additional studies be conducted on the potential toxicity of these substances to aquatic organisms, especially benthic invertebrates, algae and vascular plants. In addition, Vajda et al. (2008) identified increased gonadal intersex (i.e., the presence of both male and female characteristics within the same fish), altered sex ratios, and other reproductive abnormalities in fish downstream of wastewater effluent with elevated concentrations of endocrine-disrupting chemicals, while no evidence of reproductive disruption was observed upstream of the site. It's worth noting this study was conducted within a more confined river system, and evaluating similar effects in the marine environment is more difficult due to greater dispersion and dilution rates (Reyes et al. 2012).

As referenced above in section 2.3.1.1 of the *Environmental Baseline*, research conducted in Santa Monica Bay has found that male hornyhead turbot can exhibit levels of an active estrogen, 17β-estradiol (E2), comparable to those in reproductively active females (City of Los Angeles 2011). These elevated concentrations in hornyhead turbot were observed wherever samples were collected within the bay (i.e., at a Hyperion Treatment Plant outfall and a far-field reference site). However, E2 levels were substantially lower in male hornyhead turbot collected from offshore of Orange County (approximately 25 km south) and other flatfish species in the region, including English sole, which is managed under the Pacific Coast Groundfish FMP. 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. Although their results also showed elevated levels of E2 in males, their findings of apparently normal seasonality in androgen levels indicate that these E2 levels do not impair gonadal steroid production or its seasonality. They 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 (Reyes et al. 2012).

CECs currently have no Clean Water Act regulatory standard (e.g., no established water quality standards and/or notification levels). However, the California State Water Control Board and Regional Water Quality Control Boards have identified monitoring strategies and sampling plans for CECs. The most recent effort is a statewide pilot study monitoring plan to determine the occurrence and biological impacts of CECs (Tadesse 2016). The pilot study is designed to narrow the data gap among regions by producing comparable CEC data throughout the state. The results will help the State Water Board develop a monitoring strategy and control action. For marine outfalls in the SCB, at least two WWTP outfalls that discharge at mid-continental shelf depths will be monitored for estrone, bis(2-ethylhexyl) phthalate, butylbenzyl phthalate, p-nonylphenol, PBDE-47, PBDE-9934, and PFOS. These CECs will be monitored in effluent, ambient water, sediment, and where applicable, fish tissue.

In addition to the statewide pilot study, the Los Angeles Regional Board has required approximately two dozen publically owned treatment works (POTWs) to conduct special studies to evaluate effluent concentrations of target CECs in their discharges (including freshwater and ocean dischargers). Each facility is required to conduct annual monitoring for a minimum of two years for a suite of approximately 34 CECs. This special study requirement has been incorporated into NPDES permits as they are renewed, so not all dischargers have completed the special studies as of January 2017. The CEC special study requirement was incorporated into Hyperion's 2010 NPDES permit. Regional Board staff will evaluate the overall data set upon completion of the special studies to determine which CECs merit continued monitoring in the future, which CECs pose potential threats to water quality and beneficial uses throughout the Los Angeles Region, and whether there are significant differences in CEC loadings discharged by various POTWs.

In 2013, PBDEs were detected in Hyperion's effluent (BDE-47 and BDE-99; EPA 2017). Other fire retardants, such as TCEP, TCPP, and TDCPP were also consistently detected in the effluent when sampled. The Southern California Coastal Water Research Project (SCCWRP) also has documented PBDEs in sediment and fish tissue samples near Hyperion's outfall, reduced thyroid production in hornyhead turbot at sites near Hyperion's outfall, and changes in gene expression when exposed to 5% of Hyperion effluent in a lab setting (Bay et al. 2011; Maruya et al. 2011; Vidal-Dorsch et al. 2011). However, Bay et al. (2011) concluded that while chemical exposure at low level doses occurs, as shown by sediment and tissue analysis, the biological responses did not appear to be associated with reduced reproduction or survival. The City's receiving water monitoring program for fish abundance and fish community health also shows minimal impact to fish species, as numbers and diversity of species are greater than those in previous decades for Hyperion's outfall sampling sites. These results are also consistent with the 2008 regional Bight monitoring data indicating the condition of offshore fish communities throughout the bight is equivalent to that of reference areas (Bay et al. 2011). Regardless, the 2017 NPDES permit contains a requirement for the City to propose a special study regarding flame retardants and hormones concentrations. These monitoring requirements will provide useful information related to CECs in the SCB.

Sediment Contamination

Many heavy metals and persistent organic compounds, such as pesticides and PCBs, tend to adhere to solid particles discharged from outfalls. As the particles are deposited, these compounds or their degradation products (which may be equally or more toxic than the parent compounds) can enter the EFH foodchain by bioaccumulating in benthic and pelagic organisms at much higher concentrations than in the surrounding waters (Stein et al. 1995). Due to burrowing, diffusion, and other upward transport mechanisms that move buried contaminants to the surface layers and eventually to the water column, pelagic and nektonic biota may also be exposed to contaminated sediments through mobilization into the water column.

Areas of sediment contamination are present within the action area, much of which is a result of historical deposition and not associated with recent discharges from Hyperion. In particular, there is widespread contamination of DDT and PCBs in the bay, with the highest concentrations in deeper waters associated with larger particle sizes. Bioaccumulation of DDT in several species, including Dover sole, managed under the Pacific Coast Groundfish FMP, was well documented prior to the 1980s. High concentrations of DDT were found in the muscle tissues of these organisms and attributed to fin erosion and diseases. The concentrations of DDT and PCBs in the sediments have decreased substantially from those observed prior to the 1980s, primarily due to burial. Concentrations of DDT and PCBs in fish tissue have also decreased during that time but still remain above levels of concern. As a result, a TMDL for DDT and PCBs in the Bay was developed to address this legacy contaminant issue. Two additional TMDLs for marine debris and bacteria were enacted since the last permit issuance, due primarily to sediment contamination/toxicity resulting from historic discharge of primary treated wastewater and sludge. Despite these legacy contaminant issues, benthic communities on the Palos Verdes shelf have improved substantially. For instance, the 2013 Bight Study concluded that 68% of sediments in the Bight have minimal or low exposure to sediment contamination, while less than 1% of sediment has high exposure to contamination. In addition, the 2017 NPDES permit contains annual sediment monitoring requirements for, among others, acute toxicity, pesticides (i.e., demeton, guthion, malathion, diazinon, chlorpyrifos, and parathion), and chlorinated hydrocarbons (i.e., aldrin, dieldrin, endrin, chlordane, heptachlor, heptachlor epoxide, endosulfan I, endosulfan II, and endosulfan sulfate), which should inform our understanding of these compounds as a potential source of sediment contamination.

Impacts to Sensitive Habitats

Discharge sites may also impact sensitive habitats, such as kelp beds, if located improperly. For instance, high discharge velocities may cause scouring at the discharge point or entrain particulates and create turbidity plumes. These turbidity plumes of suspended particulates can reduce light penetration and lower the rate of photosynthesis and the primary productivity of an aquatic area while elevated turbidity persists. The contents of the suspended material can react with the dissolved oxygen in the water and result in oxygen depletion, or smother submerged aquatic vegetation sites including kelp beds and eelgrass. However, kelp beds in the bay are primarily limited to two areas (i.e., the Palos Verdes Shelf and the area from Malibu west to Point Dume), neither of which are in close proximity to the discharge point. The 5-mile outfall also discharges at a depth of 187 ft and was designed to prevent nearshore transport of the effluent. In addition, to assess their status, giant kelp beds are mapped annually throughout most of the southern California coast, including Los Angeles County, as part of the NPDES permit requirements for ocean dischargers in this region. Eelgrass habitat does exist within shallower regions of Santa Monica Bay but is not expected to be present within the action area.

Cumulative Impacts

Contaminants released into the Bay from approximately 874 total permitted discharges result in cumulative impacts to EFH. More than half of these discharges are related to stormwater, which can be a significant source of pollutants. Low flow diversions and treatment facilities, originally implemented to reduce beach closures resulting from stormwater discharges, have been effective at reducing bacteria and influent levels. When combined with other stormwater management practices, including the development of Watershed Management Program/Enhanced Watershed Management Programs to ensure compliance with TMDLs and stormwater NPDES permits, low-flow diversions will improve water quality within the bay. In addition, the City of Los Angeles's four WWTPs recycled 76.2 MGD of wastewater while LA County recycled 155 MGD of wastewater that would have been discharged into the Bay. Generally speaking, reduced flow, discharge prohibitions, and other NPDES permit requirements help improve water quality in the Bay, although the potential for increased concentration of contaminants that are not removed during treatment or recycling processes remains a concern in some situations.

Cumulative impacts associated with brine discharges from the West Basin Edward C. Little Water Recycling Plant (West Basin) also occur. The brine discharge is mixed with effluent from Hyperion and discharged via the 5-mile outfall. The main impact from the brine effluent is buoyancy, which drives initial dilution. Brine effluents are denser than freshwater effluents and may sink in the receiving water. However, because the brine effluent is such a small portion of the discharge (i.e., less than 2 percent), there is little to no impact to the discharge density. For example, even when future expansion of water recycling at West Basin to 80 MGD was accounted for in a dilution study, the resulting dilution factor of 147:1 was much lower than the dilution factor allowed under the NPDES permit (i.e., 96:1 and 84:1). Therefore, brine discharges are not expected to impact the available mixing in the receiving water. Ammonia, which can be toxic to marine organisms, is also commonly found in brine. However, all detected values are below water quality requirements within the California Ocean Plan. Moreover, the previously mentioned special study that was added to assess the projected effects of water conservation and planned recycling on effluent acute toxicity and ammonia as a result of informal consultation with NMFS should inform this issue.

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, Coastal Pelagic Species, and Highly Migratory Species FMPs due to impacts associated with the release of various contaminants into the Bay. Adverse effects to EFH for species managed under the Pacific Coast Groundfish, CPS, and HMS FMPs associated with the proposed project would be primarily limited to the ZID and to the influence of the discharge on HAB formation and prevalence. Due to the high site fidelity of many species managed under the Pacific Coast Groundfish FMP (e.g., rockfish), they may be at risk of greater localized impacts from wastewater discharges relative to other fish species with a more dispersed, pelagic distribution, such as those managed under the CPS and HMS FMPs. However, localized impacts from discharge via the 5-mile outfall have decreased, both in spatial extent and severity, over the past few decades as a result of implementing full secondary treatment and a decrease in effluent volume. Increases in invertebrate and fish species abundance and diversity suggest the conditions around the 5-mile outfall are progressing toward background conditions. Moreover, the proposed action includes measures to avoid, minimize, or otherwise offset many of these adverse effects, including source control programs for toxic constituents, compliance with discharge permit requirements and water quality standards, outfall design to prevent nearshore transport of the effluent, and effluent discharge via a multi-port diffuser to reduce discharge velocities and pollutant concentrations at the point of discharge. In addition, where data gaps exist (e.g., toxicity effects anticipated from increased water recycling, flame retardant and hormone concentrations in the effluent and loadings to the Bay), special studies have been proposed to increase the understanding of potential impacts associated with these constituents. Therefore as long as these measures are implemented, in addition to the measures spelled out in section 2.8.4 *Terms and Conditions*, we conclude there are no additional measures are needed to avoiding or minimize the adverse effects described in section 3.2 *Adverse Effects on Essential Fish Habitat*.

3.4 Supplemental Consultation

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

4. DATA QUALITY ACT DOCUMENTATION AND PRE-DISSEMINATION REVIEW

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

4.1 Utility

Utility principally refers to ensuring that the information contained in this consultation is helpful, serviceable, and beneficial to the intended users. The intended users of this opinion are the U.S. EPA and the California State Water Resources Board that jointly issues wastewater discharge permits in federal waters off the coast of California, including the permit subject to this proposed action, as well as issuing permits to all dischargers into California state coastal waters. Other interested users could include the City of Los Angeles, other wastewater treatment plants 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 EPA. This opinion will be posted on the Public Consultation Tracking System website (https://pcts.nmfs.noaa.gov/pcts-web/homepage.pcts). The format and naming adheres to conventional standards for style.

4.2 Integrity

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

4.3 Objectivity

Information Product Category: Natural Resource Plan

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

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

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

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

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